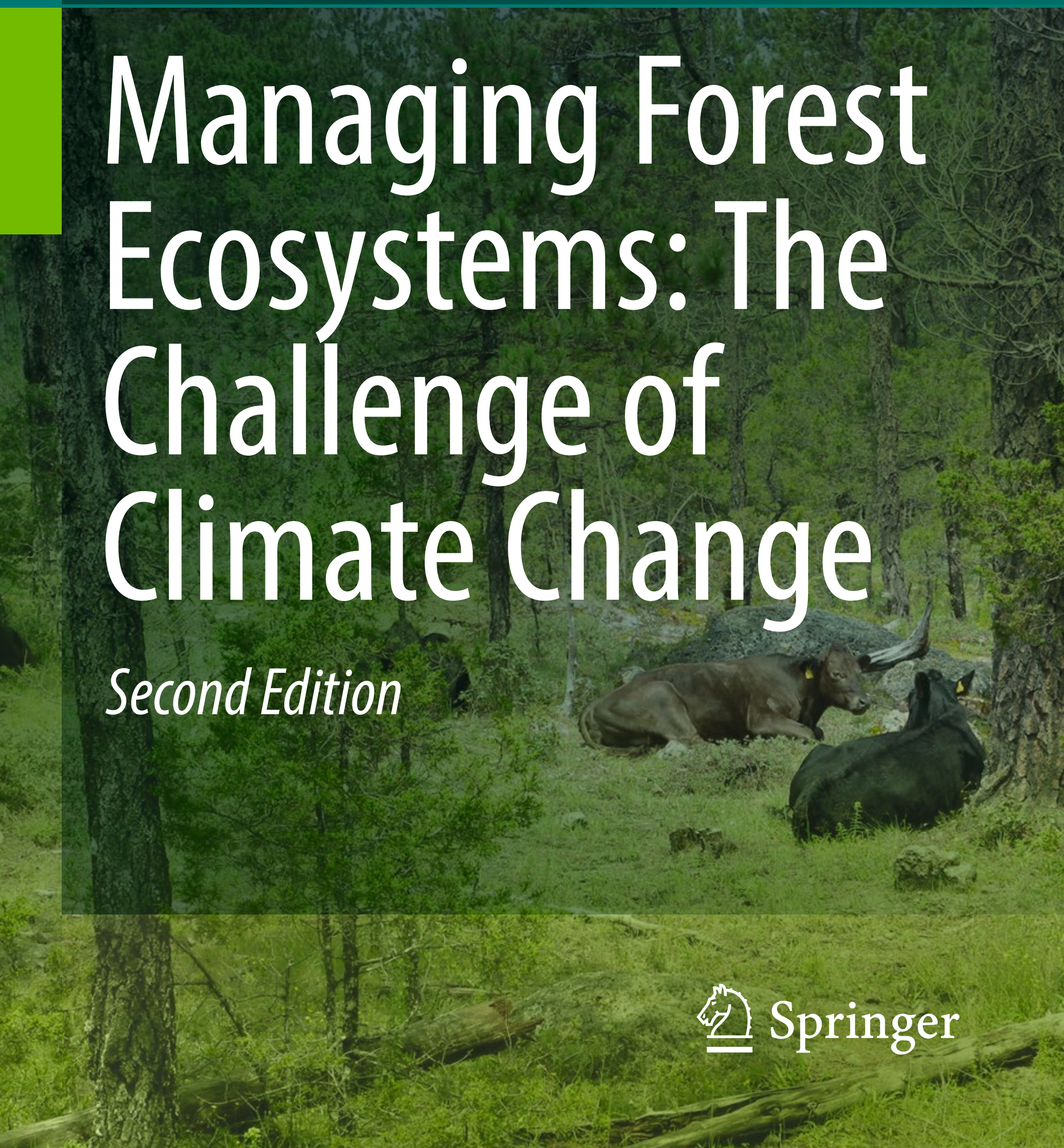
Managing Forest Ecosystems

Felipe Bravo
Valerie LeMay
Robert Jandl *Editors*

Managing Forest Ecosystems: The Challenge of Climate Change

Second Edition



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Managing Forest Ecosystems

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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.

More information about this series at http://www.springer.com/series/6247

Felipe Bravo • Valerie LeMay • Robert Jandl
Editors

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Editors
Felipe Bravo
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest Management Research Institute
Universidad de Valladolid - INIA
Palencia, Spain

Valerie LeMay
Forest Resources Management Department
University of British Columbia
Vancouver, BC, Canada

Robert Jandl
Austrian Research Centre for Forests (BFW)
Vienna, Austria

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Foreword

During the last decades, 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 or bark beetle in secondary spruce forests in Central Europe), drought episodes and intense forest fires in the Mediterranean countries and SW United States, and unusual storm activities in SE Asia. Climate changes are expected to impact vegetation manifesting as changes in vegetation extents, tree species compositions, growth rates, and mortality rates and also as species migrations. Over a number of sessions, the International Panel on Climate Change (IPCC) has discussed how forests may be impacted and also how forests and forest management practices may be used to mitigate the impacts of changes in climate, particularly to possibly reduce the rate of change. The second edition of this volume, which forms part of Springer's book series *Managing Forest Ecosystems*, presents an update on state-of-the-art research results, visions, and theories, as well as specific methods for sustainable forest management under 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. Case studies from a wide geographic range are presented on the impacts of climate changes on forest environments and economic activities and, also, possible mechanisms for ameliorating climate changes through forest management activities. As in the first edition, the volume is subdivided into five sections.

The first section 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 section titled "Overview of Climate Change and Forest Reponses" provides 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 section 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 ecophysiological models, and recent advancements in techniques for assessing and monitoring carbon stocks. In the fourth section, several approaches for economic analyses of different management scenarios are presented, including optimizing carbon sequestration in coppice rotations, estimating carbon in forests and wood products, and examining climatic impacts on forest economies, including changes in harvest cycles and uses of wood. Finally, a range of case studies on climate change impacts and mitigation activities in different ecosystems across Africa, Asia, Europe, and America is presented in the fifth section. The case studies include both natural and planted forests in temperate and tropical biomes.

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Palencia, Spain — Felipe Bravo
Vienna, Austria — Robert Jandl
Vancouver, BC, Canada — Valerie LeMay

Contents

Part I
Introduction

Chapter 1
Introduction

Felipe Bravo, Robert Jandl, Valerie LeMay, and Klaus von Gadow

1.1 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 as well as by natural disturbances including volcanic eruptions.Of the atmospheric gases, 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 ecosystem type. Increased accumulations of atmospheric gases, particularly carbon dioxide, have resulted in positive radiative forcing (i.e., more incoming solar radiation than outgoing radiation) termed the "greenhouse effect" (IPCC 2007, 2014; Norby et al. 2007; Raupach et al. 2007). The greenhouse effect plays a key role in modifying the regulation of the Earth's temperature. In 2014, the Intergovernmental Panel on Climate Change (IPCC) stated that the temperature at the Earth's surface has been successively warmer for

F. Bravo (✉)
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain
e-mail: fbravo@pvs.uva.es

R. Jandl
Austrian Research Centre for Forests (BFW), A-1131 Vienna, Austria

V. LeMay
Forest Resources Management Department, University of British Columbia, 2045-2424 Main Mall, Vancouver, BC V6T 1Z4, Canada

K. von Gadow
Georg-August-Universität, Göttingen, Germany

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the last three decades relative to all prior decades since 1850. Further, the 1983–2012 period was likely the warmest 30-year period of the last 1400 years in the Northern Hemisphere (IPCC 2014). Several climate records were broken in 2015, for example, the May global land and ocean temperature was the highest in a 136-year period (NOAA 2015). According to the World Meteorological Organization,[1] the northern hemisphere suffered a heatwave in 2015, due to a hot air migration from south to north, which impacted human health and ecosystems. Since human activities alter atmospheric gases concentrations, 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. Improved climate change models that project temperature changes and other regional-scale climatic changes, including changes in wind patterns, precipitation, and some aspects of extreme weather events, now provide best estimates and associated uncertainties for projected temperature and other climatic changes under varying human-caused emission scenarios (Jylhä 2007; IPCC 2007, 2014).

Forests play a significant role in the Earth's climate system by contributing to the cycle of atmospheric gases. Changes in the extent and characteristics of forests can greatly alter these gases. For example, Stern (2006) stated that 18 % of total annual greenhouse gas emissions are caused by deforestation. In particular, forests have a large impact on atmospheric carbon dioxide, since trees store carbon throughout their lives and then release it after death through decomposition. As a result, assessing forest carbon budgets under current and alternative forest management scenarios has received much attention in recent years (Apps and Price 1996; IPCC 2000, 2001, 2007, 2014). As well as impacts on carbon and other atmospheric gases, forests affect water balance and soil stability, and biodiversity. In areas currently subject to flood events, changes in climate are expected to alter the frequency and magnitude of these events. Trees can reduce floods through water uptake; tree root systems can prevent soil losses and can reduce instability of steep slopes under increased precipitation. As well as affecting plant species, climate changes will impact animal species. Trees can help ameliorate these impacts by moderating local climates.

Conversely, forests are affected by climatic conditions that may either promote or reduce regeneration, growth, and mortality rates (Fujimori 2001; Peñuelas et al. 2004). Climate change impacts on a forest ecosystem will vary depending limiting factors. Where low temperature is the most limiting factor, as in parts of the temperate forest biome, increasing temperatures, jointly with resource supply (Nitrogen, CO_2) and longer growing seasons, increased tree growth (Pretzsch et al. 2013, 2014). These increases were greater for more fertile sites. However, forests under low water and high temperature stress will have decreased growth rates under hotter, dry climates (Bogino and Bravo 2008) may be more susceptible to attack by pathogens and insects as found by Prieto-Recio et al. (2015) for *Pinus pinaster* stands in Central Spain. In some cases (e.g., young *Pinus pinaster* stands), the impacts may be multiplicative rather than additive (Bravo-Oviedo et al. 2008).

[1] https://www.wmo.int/media/content/widespread-heatwave-affects-europe-wildfires-north-america Accessed september 2015.

Further, fire incidence may increase, particularly in areas where annual precipitation is expected to further decrease, such as the Mediterranean forests. However, these general trends will continue to be locally modulated by altitude and other biophysical features.

1.2 Global Policy Initiatives and Forest Management Activities

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 were explicitly included in 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. Further, the United Nations Framework Convention on Climate Change (UNFCCC) proposed a mechanism to reduce deforestation and forest degradation (REDD and REDD+ (http://unfccc.int/land_use_and_climate_change/redd/items/5607.php accessed August, 2015) in the context of climate change mitigation.Currently, more than 25 countries in Latin America, Africa, and Southeast Asia are developing REDD initiatives on forest monitoring and management with the objective of maintaining or improving the forested area extents. Included in the REDD programs are alternative silvicultural practices which may have significant impacts on the global carbon budget (Mund and Schulze 2006).

Maintaining or extending the area of forested lands can be achieved locally through afforestation, alterations in harvest rates (e.g., partial harvest, reduced extents of harvest areas), reductions in regeneration delays post-harvest, and conservation of existing forest lands. For example, a number of countries have promoted tree planting on abandoned farm land and in urban areas. This increased forest land area (and trees outside forests) will increase the rate of carbon sequestered from the atmosphere for a given period. However, there is uncertainty as to when a forest area may become a net carbon emitter, since this depends upon the growth rates of live trees and the decomposition rates of dead trees (Kurz and Apps 1999; Gracia et al. 2001; Reichstein et al. 2002). For example, productive forests can temporarily turn into carbon emitters when climatic conditions constrain their growth (Ciais et al. 2005). Increasing forest lands must be coupled with other management activities.

Tree species selection can increase or maintain tree growth rates even under less favorable growth conditions. Carbon storage varies with the species, and age composition of the stand, as well as with tree health (e.g., Bogino et al. 2006; Bravo et al. 2008). For example, Bogino et al. (2006) compared single-species stands of either Scots pine (*Pinus sylvestris*) or Pyrenean oak (*Quercus pyrenaica*) to stands with a mixture of the two species in central Spain. They found that carbon stocks were greater in single-species Scots pine stands than in Pyrenean oak stands and mixed stands were between these two levels. Similarly, Bravo et al. (2008) found that carbon stocks in Scots pine stands were generally higher than in comparable maritime pine (*Pinus pinaster*) stands. These carbon stocks differences between species are likely due to differences in growth rates, tree forms, and mortality rates.

As well as species selection, genetic selection and assisted migration can mitigate climate change impacts. Using stocks that are more resilient to droughts, disease and insects can further reduce carbon emitted as mortality rates decline. In the absence of forest management, species changes would occur in response to climatic changes through natural selection of better-suited genotypes. At the same time, trees and other plants will migrate to more suitable climates through seed dispersal and vegetative reproduction on edges of current spatial distributions. As a result, diversity in genotypes would be expected to increase the probability 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 changes (Aitken and Whitlock 2013). Using more resilient genotypes can speed up the necessary adaptation to the expected changes.

Larger natural disturbance impacts such as fire, wind or flood damage that can cause great losses in forest area locally and globally (Herold et al. 2011). These disturbances may be counteracted directly by fighting the damage agent, but also indirectly via hazard reductions through management. For example, fires may be quickly extinguished before they impact larger areas. Alternatively, the probability, intensity, and extent of fires can be reduced through thinning that reduces fuel loads. Stand structures can also be altered to provide stability under higher winds. Promoting and maintaining trees near streams and on sloped ground can increase water uptake and stabilize soils, thereby reducing flood and landslide probabilities.

Overall, climate change rates can be reduced and impacts of climate changes on environments can be mitigated through a variety of forest management activities. Specific cases where these strategies have proved beneficial include the Model Forests Network initiated in Canada (Besseau et al. 2002) and extended to installations in a number of countries in Latin America and Europe. However, the efficacy of these activities in terms of atmospheric carbon reduction and mitigation of negative effects of climate changes on forests requires monitoring. Further, alternative management strategies should be formally evaluated. Efforts by the World Bank to promote forest conservation (e.g., the Forest Carbon Partnership Facility (http://www.forestcarbonpartnership.org/) and the BioCarbon Fund (http://www.biocarbonfund-isfl.org/) are undermined by its support for the neoliberal paradigm of deregulation, privatization, and structural adjustment in indebted countries. As an alternative approach, Humphreys (2006) stated that market-based initiatives, such as certification by the Forest Stewardship Council or the carbon-trade initiatives may complement the public sector. However, he rejected complete reliance on privatization and deplored the poor performance of United Nation Forest Forum (UNFF) member governments on reporting, implementation, and failure to provide leadership and direction to other forest-related institutions. In spite of these concerns, global policy initiatives and local forest management activities can result in climate mitigations. Further, monitoring and adaptive management can improve these interventions over time.

1.3 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 if, as anticipated, these events become more frequent. 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 that impact on the 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 has been used in many jurisdictions to reduce monitoring costs. In particular, Landsat imagery is being used since these have become freely available (NASA, http://landsat.gsfc.nasa.gov/ accessed August, 2015). For more detailed information on forests, other types of remotely-sensed imagery are being proposed and tested. Of these developments, Light Detection and Ranging Laser (LiDAR) (also called Airborne Laser Scanning (ALS)) shows great promise for providing detailed aboveground information. Unmanned Aerial Vehicles (UAVs) or drones are also being tested as a means of local monitoring (e.g., Tang and Shao 2015). However, for species composition, biomass above and below ground, and other information needed, remotely-sensed data must be coupled with expensive ground sampling in complex models (Weiskittel et al. 2015).

Even under the new systems of monitoring that can provide needed information under low costs, the level of uncertainty regarding future forests is very high under climate changes anticipated. In view of the difficulty in predicting the direction and rate of changes at all levels, but particularly 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, that includes observation, analysis, planned intervention, monitoring and reflection, may be too slow to adapt in response to climate changes, however. For this reason, new paradigms of managing forest ecosystems together with improved monitoring and modelling tools are needed to provide accurate forecasting and systematic evaluation of different management options based on current and past information about the forest resource.

For this purpose, one such paradigm, which is known as the Multiple Path Concept (MPC; Gadow et al. 2007) could provide a suitable basis for designing forested landscapes. The mathematical concepts have been implemented in some management models in North America (Clutter et al. 1983; Hoganson and Rose 1984; Bettinger et al. 1997) and Northern Europe (Lappi 1992; Pukkala and Kangas 1993; Eid and Hobbelstad 2000; Öhman 2002) and have been explored for specific conditions in Central Europe (Chen and Gadow 2002), Columbia (Schwichtenberg and Sánchez Orois 2003) and Russia (Gurjanov and Gadow 2005). MPC assumes that a forested landscape is an aggregation of spatially-defined land parcels (i.e., stands) of varying size and shape that may be homogeneous or heterogeneous entities characterized by a specific tree population with a given set of current attributes and site conditions. For each stand, a variety of treatment schedules or "management paths" may be potentially suitable. Each stand-path is characterized by a specific succession of management activities, unexpected hazards and growth, and has value in terms of the ecosystem services that it provides. Thus, designing a forested landscape involves a search for a combination of management paths which provides a desirable mix of ecosystem services to the landowner. Since forests sequester carbon and carbon stocks can be an additional service, the calculation of a carbon balance for alternative management paths would be included. Specifically, the carbon balance of a particular stand-path would calculated from the increased biomass due to growth relative to losses due to harvests or other tree removals (e.g., thinnings) and mortality.Included in this calculation would be decomposition rates of dead trees as well as harvest residues (i.e., roots, cutting residues) and timber products (e.g., lumber, plywood, firewood). The carbon balance of a forest landscape would then be the sum of the stand-path carbon balances.

In the past, forest-level analyses of carbon balance have been fairly rare, with most analysis at the stand level only (Bravo et al. 2008, 2010). However, models at the regional and country-wide scale have been developed to simulate carbon balances under different management scenarios as per the MPC approach. For example, the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) simulates carbon dynamics under different management pathways at the stand or forest landscape levels (https://www.nrcan.gc.ca/forests/climate-change/carbon-accounting/13107 Accessed August 2015). Similar models have been developed or are under development in other countries and regions.

The increased rate of climate changes has introduced a larger uncertainty in managing forests. More frequent monitoring using newer technologies can provide the information needed to implement adaptive management that will be needed to address climate-related changes both for small land areas (i.e., single stands) and for larger forested areas. This can both increase carbon stocks and mitigate climate impacts. Although recent research has contributed to improvements in both monitoring and management strategies, there are notable knowledge gaps.

1.4 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.

Research on forest growth and alternative management regimes has a long tradition with field experiments being established as far back as the nineteenth century. Some of these experiments have been remeasured for over a century, providing valuable information on long-term developments (Innes 2005; Pretzsch 2009) including the impact of climate changed on stand dynamics (Pretzsch et al. 2014). Both designed experiments and observational data have been used (Kuehl 1994; Gadow and Kleinn 2005). In particular, observational studies that compare growth and other changes in stands and forests under a wide range of conditions (termed quasi-experiments by Cook and Campbell 1979) have been often used. Although the attribution of cause and effect can be difficult outside of designed experiments, data from observational studies are used to develop models of process and Shipley (2000) argued that it is possible to determine cause and correlation. Further, these field installations can be analyzed under new paradigms such as future changes in climate, given past responses to climate changes. Overall, both experimental and observation data are fundamental for assessing and building models under current and alternative management alternatives.

Research into species and genetic selection under future climates will provide the information needed to at least partially ameliorate impacts of climate changes on tree regeneration success, growth, and mortality rates. This is still relatively new research with a great deal of remaining uncertainty. As well, further improvements on resilience to adverse conditions as well as on pathogens and insects is needed, along with more research on the impacts of stand management activities on the probabilities of fire, windthrow, drought, and other natural disturbances. The confluence of forest management and monitoring is critical to any mitigation of climates, and recent research on the use of remote sensing methods that reduce monitoring costs and speed up information acquisition is encouraging.

Although knowledge gaps remain, scientific evidence is available to inform management opportunities to both reduce rates of climate changes and to mitigate impacts.

1.5 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 are addressed in the four chapters following this introduction.

Part II 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.

Part III 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.

Part IV 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 Africa, Europe, Asia and the Americas is presented in Part V. The case studies include forest plantations as well as tropical and Mediterranean forests. The contributions are truly international, including authors from Africa, Asia, Europe, North and South America.

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Part II
Overview of Climate Change and Forest Reponses

Chapter 2
A Mechanistic View of the Capacity of Forests to Cope with Climate Change

Fernando Valladares

2.1 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 and 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 and BassiriRad 2003; Petit and 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 and 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 and 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 and Peñuelas 2006). These studies show the negative genetic impact of forest fragmentation, demonstrating that trees are not at reduced risk from environmental change (Fig. 2.1). This rather unexpected sensitivity of trees to forest

F. Valladares (✉)
MNCN-CSIC, Serrano 115, E-28006 Madrid, Spain
e-mail: valladares@ccma.csic.es

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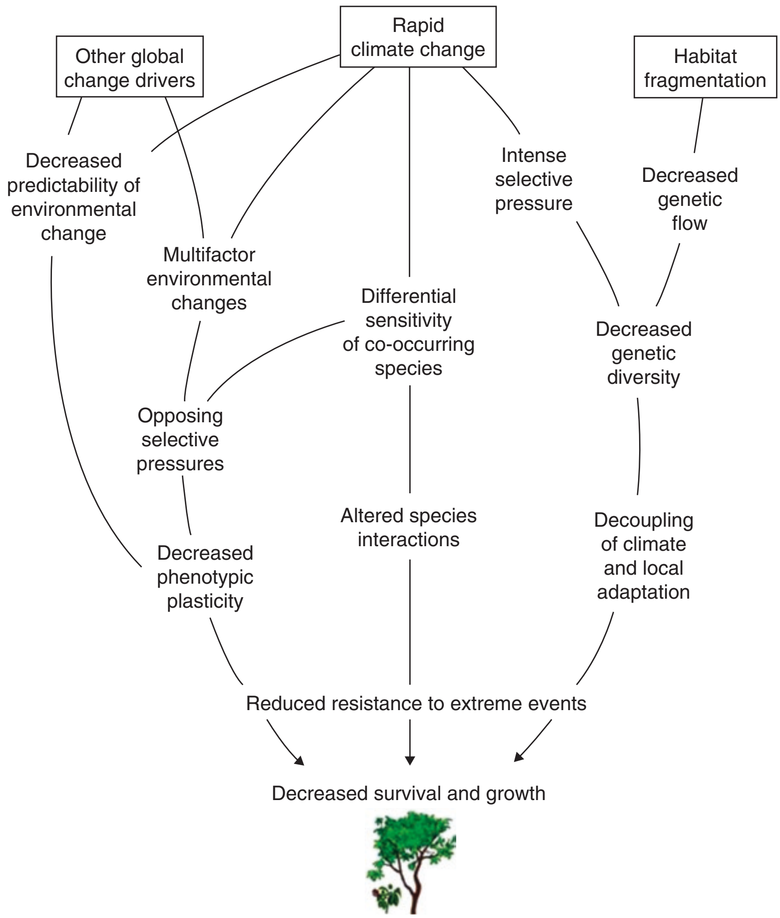

Fig. 2.1 Interactive effects of global change drivers on plant survival and growth. Rapid climate change is at the centre of a whole suite of environmental changes that is imposing complex and opposing selective pressures to forest species; this in turn limits the extent and ecological benefits of phenotypic 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)

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 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 and 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 and 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.

2.2 The Complexity of Climate Change and of Its Effects

2.2.1 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 and 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. 2009; 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 and 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. 2.2). 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).

2.2.2 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 and 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

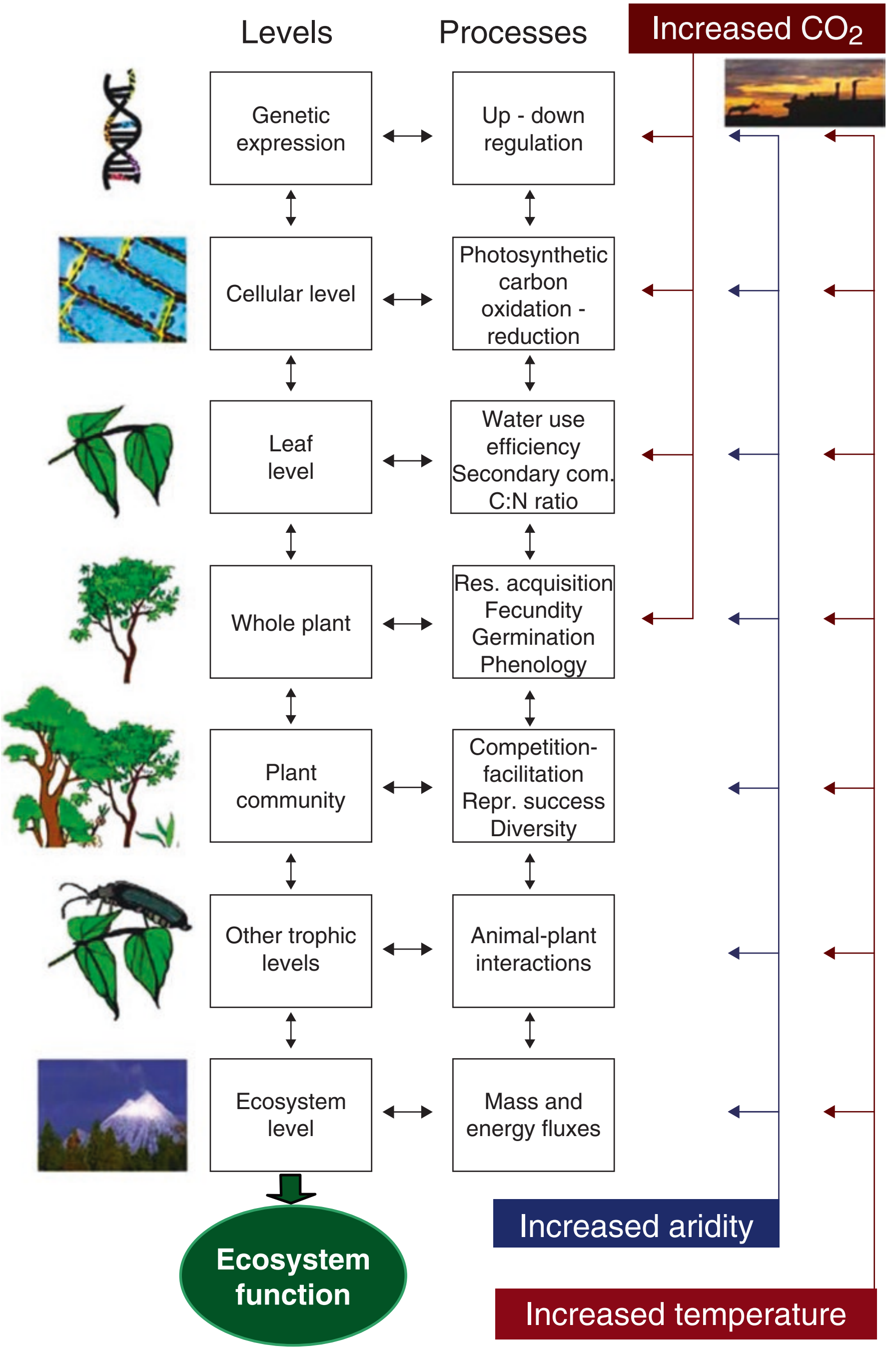

Fig. 2.2 Impacts of rising atmospheric CO_2 concentrations and the associated increases in temperature and aridity for different organizational levels from the molecule to the ecosystem. Main processes being affected at each level are indicated. Inspired on ideas by Ziska and Bunce (2006) on plant responses to rising CO_2

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 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. 2009), 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).

2.2.3 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 and 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 and Peñuelas 2003; Peñuelas and 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. 2008).

2.2.4 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 and 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.

2.3 Climate Change and Tree Physiology

2.3.1 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. 2.2, Gedney et al. 2006; Fig. 2.2, 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 and 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 and Tjoelker 2003; Zaragoza-Castells et al. 2007b). But respiration is not as well understood as photosynthesis (Cannell and Thornley 2000; Grace and 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 and 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).

2.3.2 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).

2.3.3 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 and 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.

2.3.4 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 and 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. 2.3), without overlooking biotic stresses (e.g. those induced by competing neighbours, herbivores, pathogens etc.; Figs. 2.3 and 2.4). 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

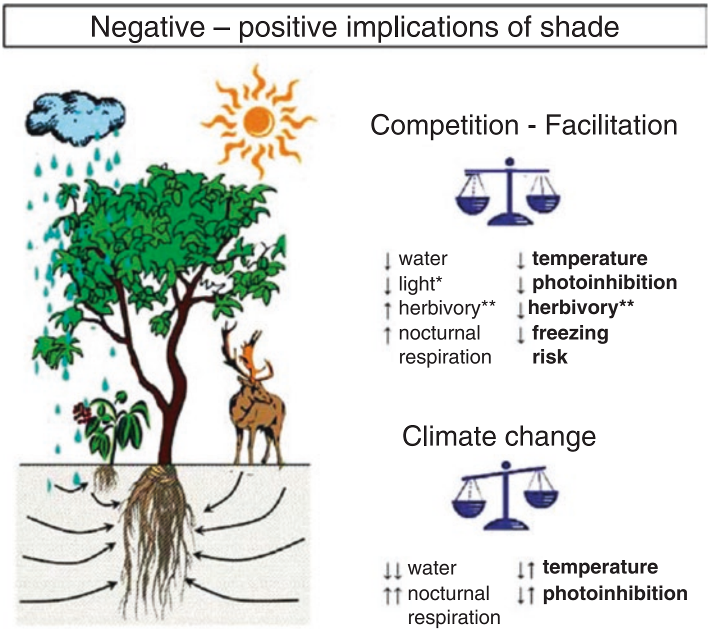

Fig. 2.3 Many forest plants get established in the shade due to either higher density of propagules or better environmental conditions than in open microsites. However, not all effects of established plants (i.e. those casting the shade) are beneficial for understory plants. There is a balance of negative 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

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), 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

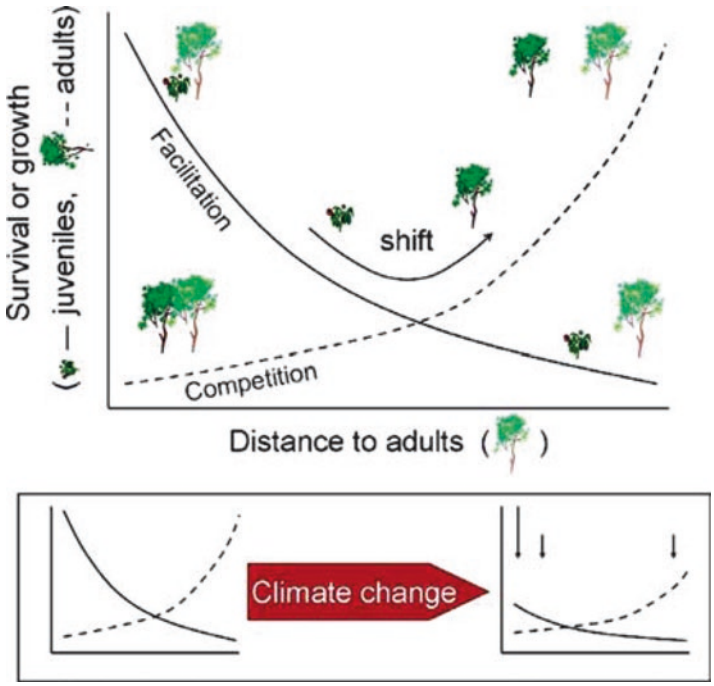

Fig. 2.4 Performance of juvenile plants (expressed by any surrogate of fitness like survival or growth) versus distance to adult, established plants in the field. The initial facilitation of juveniles by adult plants shifts into competition at a certain ontogenetic stage (indicated by the arrow in the 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)

on the ecological and physiological features of each species occurring in the understory (Fig. 2.3). 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.

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. 2.3). Shade is in itself a stress for most plants (Valladares and 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. 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. 2.3). 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 and 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 and Valladares 2006).

2.4 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 and 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. 2007).

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., 2007). 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 and 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. 2.1) (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 and 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 and 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.

2.5 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 and 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 and 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 and Kadmon 2000; Valladares and Pearcy 2002) and there is no consensus on whether the shade cast by a potential nurse is always beneficial (Fig. 2.3, 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. 2.4, 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. 2.4). 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 and 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. 2.1).

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.

2.6 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 and Gutiérrez 2004; Kullman 2002; Lloyd and Fastie 2003; Peñuelas and 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 and 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 and Petit 2005).

2.7 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 and 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. 2008). 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. 2.3), which may significantly change dynamics and long term stability of present day Mediterranean forests.

2.8 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 and 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.

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Chapter 3
Greenhouse Gas Emissions from Temperate European Mountain Forests

Robert Jandl, Mirco Rodeghiero, Andreas Schindlbacher, and Frank Hagedorn

Abstract Forests are covering a substantial part of European mountains. The elongation of the growing season due to climate change and warmer summers are increasing the rate of soil respiration. However, the effect is at some sites partially compensated by droughts. Temperate mountain forests ecosystems are mostly releasing carbon dioxide whereas nitrogen oxides are of lesser importance. Societal changes are also affecting the greenhouse gas emissions from forests. The structural change in agriculture causes an increase of the forest area. The change in land use from grassland to forest, leads to the formation of an organic litter layer on the soil surface but to a reduction of the carbon input by decaying roots to the mineral layers. Moreover, the effect of the increased carbon sequestration in the biomass of more productive forests is partially offset by losses of carbon dioxide from the soil. Climate change also calls for an adaptation of the strategy of forest management. The economical feasibility of timber production in remote high-elevation forests will not be significantly increased. Nevertheless, even marginally productive mountain forests need to be managed in order to ensure the provision of ecosystem services such as protection against natural hazards. Mountain forests with a stable stand structure can better cope with disturbances such as storms and biotic pressures, thereby reducing the risk of carbon losses to the atmosphere.

R. Jandl (✉) • A. Schindlbacher
Austrian Research Centre for Forests (BFW), A-1131 Vienna, Austria
e-mail: robert.jandl@bfw.gv.at

M. Rodeghiero
Department of Sustainable Agro-Ecosystems and Bioresources, Research and Innovation Centre, Fondazione Edmund Mach, 38010 San Michele all'Adige, Italy

F. Hagedorn
Swiss Federal Institute for Forest, Snow and Landscape Research, 8903 Birmensdorf, Switzerland

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3.1 Introduction

In many countries around the globe, mountain regions are often economically disadvantaged. However, increases in tourism along with strong political incentives such as the European Rural Development regulation have resulted in Central Europe mountain regions that are economically relatively strong. This development towards greater prosperity is a recent phenomenon. In very recent history, mountain land areas were used to the maximal carrying capacity and even slight inter-annual variations in crop productivity jeopardized the livelihood of mountain communities (Jandl et al. 2012). As well as providing tourism and agricultural economic benefits to local communities, mountain forests fulfill many ecosystem services including drinking water, protection against natural hazards, biodiversity, timber, greenhouse gases (GHGs) sequestration, and scenic beauty.

More than one third of Europe is located in mountain areas, defined by combination of characteristics in particular elevation, slope steepness, and climate. In Central Europe, about 40 % of the mountain area is covered by forests (Price 2010). The main other land types in mountain regions are alpine pasture, natural grassland, and cropland arranged in a characteristic mosaic. These mountain pastures are in decline in a large part due to encroachment by forests, thereby leading to a loss of landscape diversity with a high conservation value (Russ 2011). This relatively recent trend in conversion from pasture to forests represents a significant change. For centuries, timber extraction was the dominant aim of forest management. Timber products from mountain forests served local markets. At the same time, patches of land were cleared and maintained as pastures to seasonally expand the agricultural land from valleys into higher elevations. The historical economical forestry context balancing timber production, infrastructure protection, and settlements provisions has changed because of the steadily growing demand for land by tourism. As a result, the historical forest management complex is losing relevance as a part of the primary sector of economy. The local economy no longer depends on locally produced timber. The high costs of timber harvest and silvicultural interventions in mountain regions further marginalize timber production. The traditional focus on timber production is no longer economically viable and intermittent forms of timber production, that is brief episodes of active timber management activities along with long intervals without interventions, are a realistic scenario (Broggi 2002). Nevertheless, many ecosystem services provided by mountains, such as protection against natural hazards, are largely linked to forests (Price 2010).

Climatic change redefines environmental conditions and mountain ecosystems are listed among the most vulnerable types of ecosystems (IPCC 2013). Evidence for climate change effects on mountain plants has been documented for a long time (e.g., Grabherr and Pauli 1994; Dullinger et al. 2004; Hagedorn et al. 2014). Because of long distances to significant water bodies, warming in mountain areas is expected to be above the global average (Rebetez and Reinhard 2007; Pepin et al. 2015). Summer droughts are likely in some mountain regions and may reduce the productivity of forests (Hanewinkel et al. 2012; Lévesque et al. 2014). Scenarios about future storm frequencies are highly uncertain but could impose a considerable threat on forests (Matulla et al. 2008; Usbeck et al. 2010). There is high agreement that the pressure from pests and pathogens on forests will increase (Wermelinger 2004; Seidl et al. 2009; Cerbu et al. 2013; Jandl et al. 2013), thereby increasing the need for more frequent monitoring and management activities in forests. Climate change will also affect the tourism sector; for example, there are calls for an economic re-evaluation of currently successful skiing resorts.

In this context, we evaluate the effects of climatic change on GHG emissions from mountain forests. Estimates for global or even regional GHG fluxes are highly uncertain and fundamental data are still missing. For instance, the estimated annual global rate of C respiration from soils has been updated within 10 years from 75 to 50 Petagrams (Schlesinger and Andrews 2000; Field et al. 2004). Here, we will focus on the GHG fluxes and their controls in mountain forest ecosystems.

Mountain forest ecosystems store enormous amount of C and N in 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; Büntgen et al. 2014) as a result of several factors such as increased nitrogen deposition, a longer growing season, CO_2 fertilization, and adapted forms of forest management. In addition, the current changes in land-use dynamics are drastic in Mid-European mountain forests. For instance, in Switzerland, the forested area increased by 1 % per year at altitudes above 1400 m as compared to 0.2 % at lower altitudes (Brändli 2010). Consequently, both growth and area of mountain forests are currently increasing thereby altering the ecosystem GHG budget.

3.2 Carbon and Nitrogen Dynamics of Mountain Forests

Mountain forest soils have higher stone contents and are shallower than soils at lower elevations. On siliceous bedrock, the dominant soil formations are Cambisols and Spodosols in different stages of development, while on calcareous limestone, rendzic Leptosols are commonly found (IUSS; FAO soil classification 2014). For Austria and for many other mountain forests, high elevation soils contain similar amounts or even more C and N than low elevation soils (Fig. 3.1). Most high elevation soils are, irrespective of the bedrock material, poor in clay content. Soil organic matter plays therefore a crucial role for the sorption and storage of nutrients. Any climate-driven decline in soil organic matter will have consequences for site productivity. Commonly, the aboveground forest biomass declines with altitude, as trees are becoming smaller and the canopy cover less dense. At the country scale, the proportion of forest land is smallest at low elevation and increases up to timberline (1500–2200 m in Austria) (Fig. 3.1, rightmost graph).

Climate warming speeds up the biogeochemical cycling of C and N in forest ecosystems. In particular, the response of organic matter decomposition to changing climate is a matter of concern as increased decomposition rates would mean higher CO_2 release from the forest soil to the atmosphere (Fig. 3.2). Experimental evidence for the future development of the CO_2 balance of mountain forests is provided via a network of expensive soil warming studies. This information is important for verifying the

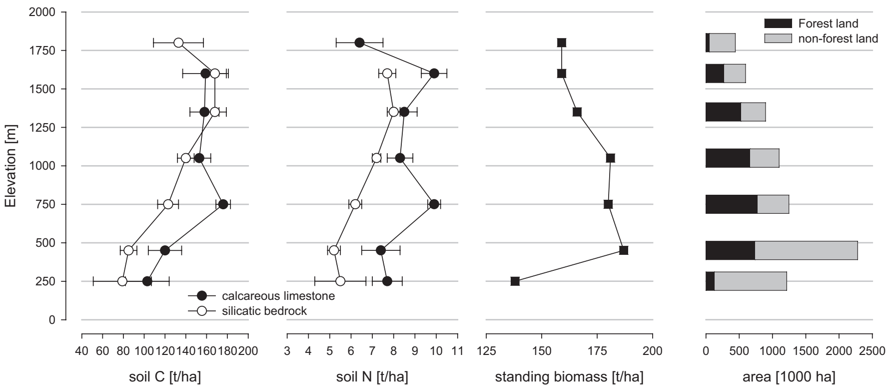

Fig. 3.1 Soil carbon, soil N, standing biomass (stems, branches, needles of conifers), and forest/non-forest areas versus elevation for Austria (Data sources: Soil data: Austrian Forest Soil Observation System, Biomass and land use: Austrian National Forest Inventory)

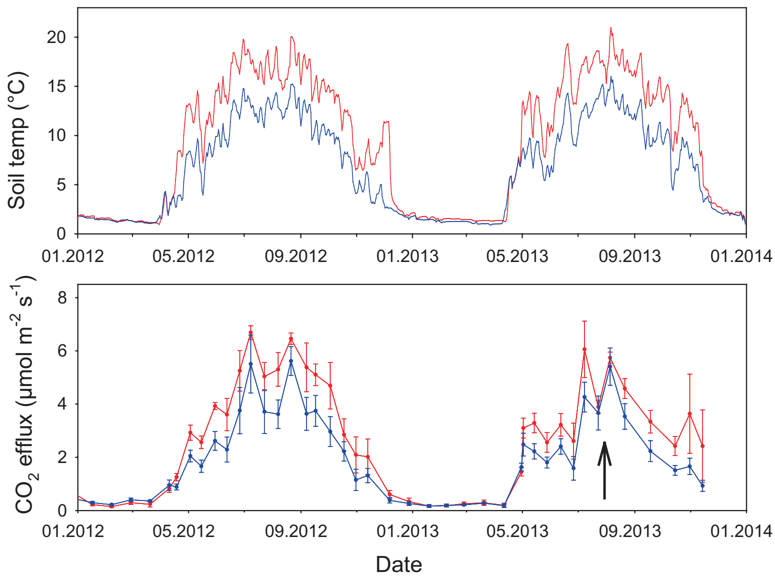

Fig. 3.2 Soil temperature and CO_2 efflux (*red* = warmed, *blue* = control) during 2012 and 2013 in the soil warming experiment at Achenkirch/Austria. The soil was warmed during the snow-free season by 4 °C. The arrow indicates a period of drought conditions during which the warming effect on the soil CO_2 efflux diminished

complex interactions between manifold processes in the C cycle. 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 sizes of the carbon pools of plants and soil, soil water regimes, and the vegetation cover (Shaver et al. 2000). Temperature is important but may or may not be the single most important factor for the turnover of soil C, since substrate quality may be equally relevant in the turnover of soil C (Davidson and Janssens 2006). The overarching question is if the soil CO_2 efflux will consistently increase under warmer conditions, or if a new equilibrium in the C pool will be established at some point. In other words, the question is how much of the large soil C pool is available for decomposition by soil microbes and how much is protected against decomposition. Soil warming experiment in a low elevation forest in USA has shown a pronounced decline in the warming response with time (Melillo et al. 2002). Similar observations have been made in a soil warming experiment in a boreal forest in Sweden (Strömgren and Linder 2002). The declining response of soil CO_2 efflux to warming was explained by a shortening of labile, easily decomposable C over time (Kirschbaum 2004) and microbial adaptations to warmer conditions (Bradford et al. 2008). Recent studies in the Alps, however, indicated that mountainous soil may have a more pronounced response to soil warming than anticipated. In a long-term soil warming experiment in the northern limestone Alps, soil CO_2 release immediately increased and continued at elevated rates during 9 years of intensive (4 °C) warming (Schindlbacher et al. 2008; Schindlbacher et al. 2012). Similarly, a soil warming at the alpine treeline resulted in sustained increases in soil CO_2 effluxes (Streit et al. 2014). The likely reason for the strong response of mountain forest soils to warming are the high amounts of C stored in the topsoil which appear to be relatively readily available to decomposing microorganism (Hagedorn et al. 2010). Therefore, old but labile C currently stored in the soil could be released to the atmosphere (Streit et al. 2014).

Aside from warming, changing precipitation patterns can affect the soil CO_2 efflux. More frequent occurrence of dry summers likely dampens the warming effects on soil CO_2 efflux (e.g. Fig. 3.2) or may even offset warming effects (Schindlbacher et al. 2012; Hagedorn and Joos 2014). In mountainous regions, a substantial amount of precipitation falls as snow which usually forms a consistent snow layer throughout most of the cold season. This has important implications on soil microclimate as the snow layer isolates the soil from low air temperatures. Under an insulating snow cover, soil microbes are still active and high concentrations in CO_2 can build up in the snow.

Soil CO_2 efflux during winter accounts for approximately 10 % of the annual soil CO_2 production at mid-elevation temperate mountain forests (Schindlbacher et al. 2014), whereas it can make up to 30 % of the annual efflux at cold, high elevation sites (Liptzin et al. 2009) with corresponding long-lasting snow cover. These respiratory losses can diminish or even offset the C sequestration during the growing season. 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) in high elevation forests. 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 (Schadt et al. 2003; Monson et al. 2006). When these microorganisms run out of fresh substrate, they can even mineralize old C, as shown by ^{14}C data (Monson et al. 2006). CO_2 production under snow is primarily driven by soil temperature and substrate availability (Brooks et al. 2004; Monson et al. 2006; Schindlbacher et al. 2014). Therefore, the onset of snow cover as well as its properties throughout the cold season is important. Under warmer conditions, the snow cover develops later and melts earlier in a year. The effect bears the surprise that, somewhat counter-intuitively, soils can become colder. Especially at high elevation sites, soils could be unprotected by an insulating snow cover for a shorter time and potentially cool down during winter (Groffman et al. 2001). At lower elevations sites, however, decreasing snow cover generally leads to warmer soil (Kreyling and Henry 2011) and thereby potentially increases the current cold season soil CO_2 efflux. A further important role of snow cover is its function as water reservoir for the following spring and growing season plant growth and carbon sequestration potential (Monson et al. 2002).

3.3 Nitrogen Oxides

Forest ecosystems across Europe receive a wide range of nitrogen inputs from the atmosphere as wet and dry depositions, with rates going from less than 1 (northern Norway and Finland) to 60 (in the Netherlands and Czech Republic) kg N ha^{-1} y^{-1} (MacDonald et al. 2002). Nitrogen-limited ecosystems such as many mountain forests usually have a tight N-cycle; in fact, they can carry the legacy of centuries of nutrient exploitation and are critically deprived in N (Führer 2000). Consequently, N is withheld tightly and limits biomass production. However, with the ongoing high N deposition, the demand of ecosystems may be at least temporarily exceeded and in case of nitrogen saturation (i.e. N losses approximate or exceed the N inputs; Agren and Bosatta 1988), N can leave the system via the aquatic or the gaseous phase (Huber et al. 2004, Perakis and Hedin 2002, Vitousek et al. 1997).

Nitrous oxide (N_2O) is one of the main greenhouse gasses having a global warming potential of about 300 times that of carbon dioxide (CO_2; Myhre et al. 2013). Nitric oxide (NO) is indirectly involved in global warming contributing to the production of radiatively active tropospheric ozone and the formation of acid rain (Williams at el., 1992). Almost 70 % of the total amount of N_2O emitted from the biosphere originates from soils (Mosier et al. 1998) due to nitrification and denitrification processes. In particular, production of NO mainly occurs via aerobic nitrification whereas N_2O is mainly released during aerobic denitrification (Davidson et al. 2000). A minor contribution is attributable to biomass burning, industrial processes and cattle breeding (IPCC 2013). While the formation of N_2O is rather difficult in 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. Moreover, N_2O emissions are acknowledged to increase at higher water contents through greater loss from denitrification, whereas the maximum NO emissions are assumed to occur at low to medium soil water contents (Brumme and Borken 2009).

Increased N availability is recognized as being responsible for high N_2O and NO emissions from European forest soils (Skiba et al. 1999; Kim et al. 2012). Zechmeister-Boltenstern et al. (2002) observed that within nitrogen enriched forests, nitrate leaching and N_2O emissions may be linked during the plant growing season. In fact, they recorded the highest N_2O emission rates (>150 mg N m^{-2} h^{-1}) when also nitrate leaching was evident in a lowland beech forest in Austria. On the other hand, Eickenscheidt and Brumme (2012) found that the long-term reduction of N deposition to pre-industrial levels turned a lowland Norway spruce (*Picea abies* L.) forest soil from a net source of nitrogen oxides to a net sink, due to a decline in net nitrification rates. Mountain ranges tend to capture precipitation, especially when they form a prominent barrier in the landscape towards the prevailing wind direction. Therefore, 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.

The formation of N_2O in soils has long been seen as a phenomenon of waterlogged soils (Bowden 1986). However, it is now evident that dry temperate forest soils also are typically sources of atmospheric N_2O (Kesik et al. 2005). In particular, nitrogen oxides emissions from N enriched lowland forests have been reported (Butterbach-Bahl and Gundersen 2011). Mountain forest soils can release relatively large quantities of nitrogen oxides as well (Härtel et al. 2002, Pilegaard 2001). 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^{-1} y^{-1} under ambient N deposition and 2.9 kg N ha^{-1} y^{-1} under experimentally increased N inputs of 30 kg N ha^{-1} y^{-1} (Mohn et al. 2000; Kitzler et al. 2006). More recently Krause et al. (2013) investigated a Norway spruce forest at 1200 m a.s.l. in central Switzerland (Alptal). Soil N_2O fluxes ranged from −95 (i.e. N_2O sink) to 235 μmol m^{-2} h^{-1}, denoting a high temporal variability, and these fluxes were positively correlated with soil temperature. These figures appear to be low compared to the annual releases of carbon dioxide of 1.5–4 t CO_2-C ha^{-1}, indicating that soil CO_2 fluxes dominate the soil-atmosphere GHG balance whereas N_2O play only a marginal role. However, emissions could increase under warmer conditions that are predicted for European summers due to climate change (Krause et al. 2013) and, given that N deposition will remain high, these changes in emissions have become an issue of environmental concern (Schindlbacher and Zechmeister-Boltenstern 2004).

3.4 Land Use Change

At the global level, human activities have directly affected most of the ice-free lands; less than 30 % of the land surface is thought to be pristine (Luyssaert et al. 2014). According to the Intergovernmental Panel on Climate Change (IPCC 2006), a land use change (LUC) is the change from one land-use category (e.g. forest land, grassland, cropland) to another. LUC can strongly affect the balance between inputs and outputs of soil carbon, thereby potentially also influencing the amount of carbon stored both in vegetation and in soil. As a consequence, LUC can play a major role in global carbon budgets (Foley et al. 2005). As a general tendency, afforestation of croplands and conversions of croplands to grasslands favor carbon accumulation, whereas the opposite land-use changes deplete soil organic carbon stocks (Vesterdal et al. 2011).

The dominant LUC and main driver for the establishment of new forest areas in mountain regions of the European Alps is the human abandonment of grasslands followed by natural colonization by tree species (Zimmermann et al. 2010b). This natural process must be considered separately from the so-called "human-induced afforestation" in which soil is more subject to carbon losses due to site preparation and tree planting activities (Poeplau et al. 2011). The reasons for land abandonment are primarily socio-economical where agricultural specialization/intensification occurs in the more fertile areas and less productive and less accessible sites are not utilized for agriculture (MacDonald et al. 2000; Tasser and Tappeiner 2002; Bolliger et al. 2008; Tappeiner et al. 2008). The grassland colonization by forests is usually thought to increase the carbon stocks of both the soil and the vegetation, therefore representing a potential opportunity for carbon sequestration contributing to climate change mitigation (Schimel et al. 2001). However, while an increase in aboveground biomass following forest establishment on grasslands is expected, changes in carbon stored in soil are more difficult to predict (Jackson et al. 2002). A number of factors can affect the direction and magnitude of soil organic carbon changes including: past use, time since abandonment, site history, climate, management, and soil type (Meyer et al. 2012).

The colonization of grasslands by forest trees also results in a build-up of organic materials from litterfall, leading to an increase of carbon stocks in organic layers (Poeplau et al. 2011; Hiltbrunner et al. 2013). Contrasting trends in soil carbon storage have been reported for the mineral layers: a decrease in mineral carbon stocks after forest expansion was noted by Thuille et al. (2000), Thuille and Schulze (2006) and Alberti et al. (2008), whereas an increase or no change were observed in other studies (Montane et al. 2007; Risch et al. 2008). Recently, Guidi et al. (2014) performed a detailed study over a land use and management gradient in a subalpine mountainous area of Trentino (northern Italy). Four successional stages starting from a managed grassland, going through an abandoned grassland towards an early stage forest, and leading to an old mixed Norway spruce and beech (*Fagus sylvatica* L,) forest, were compared. A decrease in soil organic carbon stock (to a depth of 30 cm) was evident moving from the grassland to the old forest (Fig. 3.3). In this case, the forest expansion caused a 50 % reduction in soil organic carbon stocks even after accounting for the organic layers in the stock computation. Similar results were pointed out by Hiltbrunner et al. (2013) who found lower organic carbon stocks under a 45-year old Norway spruce stand versus pasture land. This pattern can be explained by the higher inputs of root-derived carbon in the managed grassland than in the forests (Hiltbrunner et al. 2013; Guidi et al. 2014; Solly et al. 2014).

3.5 Forest Management at the Timberline

Forests in mountain regions may play a role in C sequestration. Using growth conditions for a high-elevation stone pine (*Pinus cembra* L.) forest in Obergurgl, Tyrol, we simulated an increase in the forest productivity by more than 15 %, using the

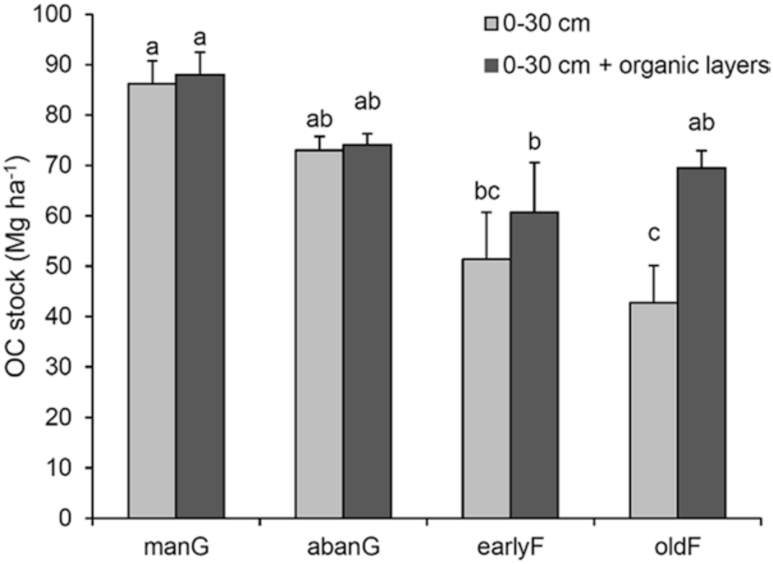

Fig. 3.3 Total soil organic carbon stocks (to a depth of 30 cm) for four land successional stages (*manG* managed grassland, *abanG* abandoned grassland, *earlyF* early-stage forest, *oldF* old forest). *Error bars* are the standard error of the mean (n = 3). Within each soil layer, different letters indicate significantly different means based on Tukey test, with $P < 0.05$. If no letters are present, means are not statistically different (from Guidi et al. 2014)

simulation program Caldis (Kindermann 2010). Forests at elevations near timberline are presently beyond the reach of bark beetles and many other pests and pathogens, and even under warmer conditions, the pressure from biotic stressors is not expected to increase substantially. At our experimental site, the risk of storm damages is very low due to the position of the forest stand. However, the expected growth enhancement under warmer conditions will not be sufficient for a commercially-viable timber production. Even under the present high prices for stone-pine timber, the yield cannot compensate for harvesting and management costs. Further, to maintain the protective function of the stand, about 30 % of the volume needs to be conserved. For the investigated region, government subsidies are available to compensate for incurred economic losses under timber management. We evaluated three forest management regimes: (i) zero intervention (i.e., unmanaged forest); (ii) extensive silviculture with harvesting at 50-year intervals; and (iii) intensive silviculture with harvests every ~30 years. Our results showed that even modest harvests in 50-years intervals reduce the standing stock in the long run. The forest management strategies that included forest harvest are not sustainable based on our simulations for this forest, especially since the overarching function of acting as a protection forest is not ensured (Fig. 3.4). Another relevant ecosystem service is carbon sequestration in the soil. Our modeling exercise showed that the unmanaged forest is a strong and persistent carbon sink. Extensive silviculture reduces the carbon stock in the ecosystem, but has the benefit of supplying

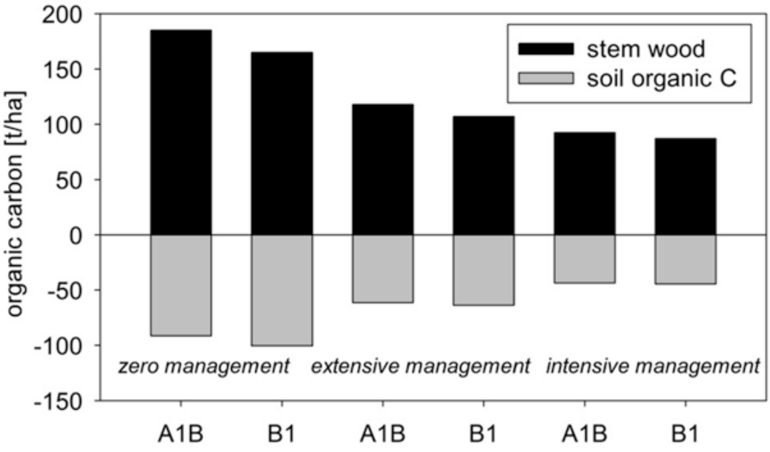

Fig. 3.4 Carbon storage in the stemwood and the soil in a high-elevation stone-pine forest in Obergurgl, Tyrol under three management regimes and two climate scenarios (IPCC climate scenarios A1B, B1)

some timber. Intensive silviculture, however, would strongly reduce the carbon stocks in the ecosystem. Further, the growth simulation runs did not show evidence for increased disturbances by bark beetle. Overall, we concluded that maintaining the forests without active management is good practice.

3.6 Indirect Climate Change Effects via Vegetation Changes

Climatic change may not only directly affect the greenhouse gases balance, but may also indirectly affect this balance via changes in vegetation. For instance, in North America, Scandinavia and Siberia, timberlines are shifting to higher latitudes indicating that the warming during the last century has induced a large-scale change in vegetation (Kullmann 2002; Harsch et al. 2009; Hagedorn et al. 2014). In the Alps, declining use of alpine meadows has led to reforestation and hence, forest areas are currently expanding (Gehrig-Fasel et al. 2007). Forest expansion will clearly increase biomass C stocks, but, as noted earlier, the effects on soil C are less predictable. While this shift in vegetation will increase C inputs into soils, at the same time increases in soil temperatures with increase respiratory C losses from soils (Hagedorn et al. 2010). Sjögersten and Wookey (2002) found that tundra soils above the treeline contain more labile C than forest soils, suggesting that a warming and a shift in treeline to more northern latitudes would induce CO_2 losses from soils. On gentle slopes in remote areas of the Ural mountains, Kammer et al. (2009) assessed the effects of timberline advances on soil organic matter cycling using a space for time approach by sampling soils along altitudinal gradients which experienced well-documented treeline advances. Their main finding was that the upward

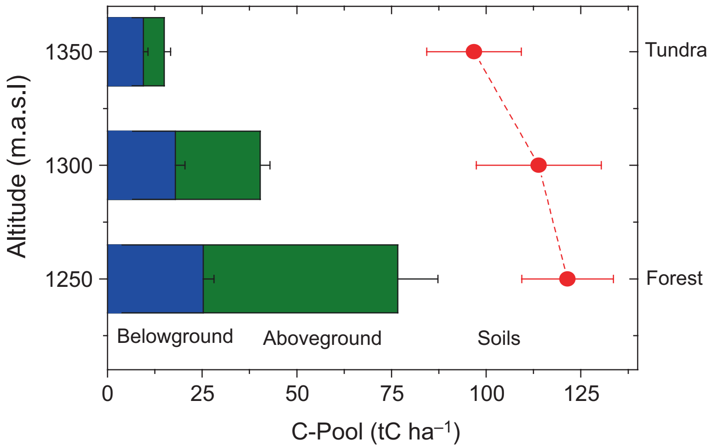

Fig. 3.5 Below- and above-ground C stocks along altitudinal gradients of the gentle slopes of the Southern Ural (*left*) with associated means and standard errors of five plots (*right*)

expansion of forest had small net effects on C storage in soils (< 30 g C m^{-2} y^{-1}) despite the strong increase in biomass (Fig. 3.5). However, there was a strong change in soil organic matter (SOM) quality, with a declining thickness of the organic layer and a litter layer with higher quality (e.g., a lower C/N ratio) from the tundra to the forest. As a consequence, soils within the forest had higher rates of C cycling and an increased net N mineralization than tundra soils, which in turn might stimulate plant growth and thus C sequestration in tree biomass (Kammer et al. 2009). Similarly, in the Peruvian Andes, Zimmermann et al. (2010b) found no changes in soil carbon pools across treelines, but the distribution changed with significantly smaller C contents in topsoils and higher C stocks in the deeper soils in the montane forest versus the 'Puna' above treeline. This is also indicative for a faster C turnover and a greater incorporation of soil C in the mineral soil at a lower elevation with a more favourable microclimate. In summary, these findings along natural gradients suggest surprisingly small effects on total soil carbon storage and hence on C sequestration, but large effects on SOM cycling rates. The effects on other greenhouse gases remains uncertain, but a faster N cycling might also be associated with higher N_2O emissions.

3.7 Conclusions

Climatic change will lead to an increase in biomass production. Due to the low overall productivity of many mountain forests, the rate of the incorporation of CO_2 in the biomass is slow. At the stand level, an overall increase in the C and N pools is expected. The effect is reinforced by an increase in the forest area as a result of a rising timberline and an encroachment of trees into 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 sequestration of C via elevated growth rates. However, if natural disturbance rates (storms, pest infestation, and others) also increase with climate changes, the mitigation potential of forests decreases.

While the productivity of mountain forests benefits from climatic change, the soils behave differently. Mountain forest soil store large amounts of C and N, particularly in labile forms. 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 results in expansion of productive forests, it is expected that mountain forests continue to be sinks of GHGs. Even modest increments in the soil temperature can increase the CO_2 emission by 20–30 %. It remains to be shown in long-term ecological research programs how long this ecosystem response prevails and whether a new equilibrium state of soil C and N pools will be reached in a warmer future.

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Part III
Monitoring and Modeling

Chapter 4
Estimating Carbon Stocks and Stock Changes in Forests: Linking Models and Data Across Scales

V. LeMay and W.A. Kurz

Abstract The increasing amount of atmospheric carbon has been linked with changes in climate, prompting efforts to reduce the amount of carbon emissions. Estimates of forest carbon stocks and stock changes are needed, along with how these change over time, and how sequestration might be increased through forest management activities such as afforestation, reforestation, stand management, and forest protection. 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. For large land areas, estimating the amount of carbon sequestration into and out of a forest system involves integrating a number of data sources and models at a variety of spatial and temporal scales. The methods used to integrate data and models across time and spatial scales vary. In this paper, we present a discussion of methods used to obtain information on carbon stocks for very large land areas, using reported analyses as examples.

4.1 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 1774 ppb[1]) (IPCC

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

V. LeMay (✉)
Forest Resources Management Department, University of British Columbia,
2045-2424 Main Mall, Vancouver, BC V6T 1Z4, Canada
e-mail: Valerie.LeMay@ubc.ca

W.A. Kurz
Natural Resources Canada, Canadian Forest Service,
506 West Burnside Road, Victoria, BC V8Z 1M5, Canada
e-mail: wkurz@nrcan.gc.ca

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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 twenty-first 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 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.

4.2 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. 4.1).

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

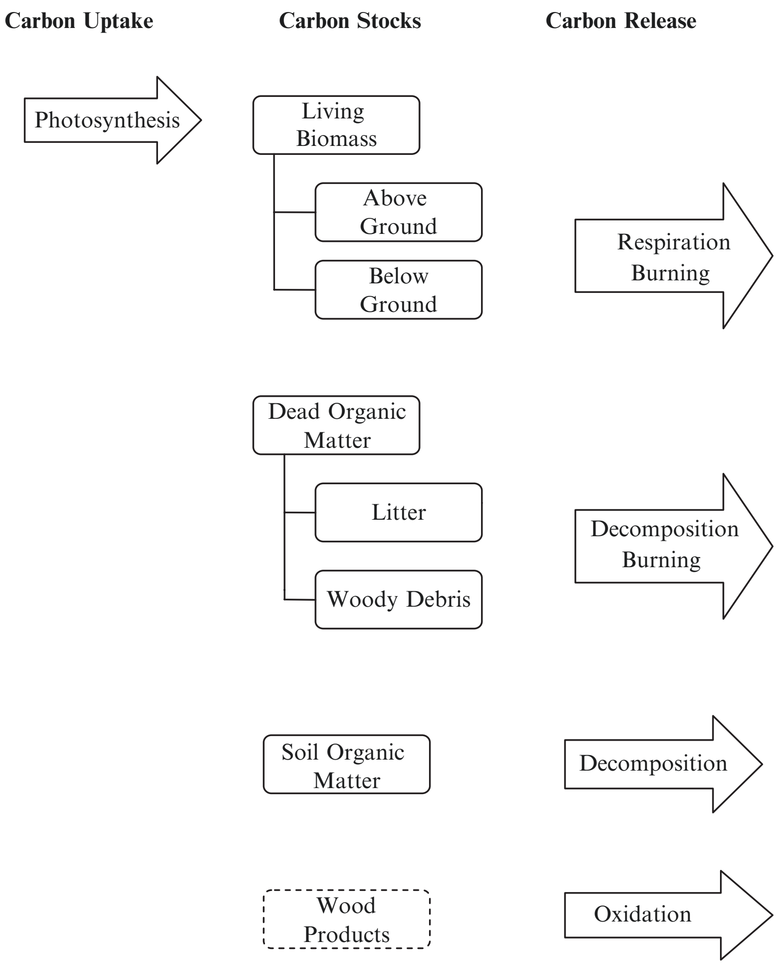

Fig. 4.1 Breakdown of carbon stocks and processes that result in changes to carbon stocks

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.

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.

4.2.1 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.

4.2.2 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.

4.2.2.1 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 and Field 2005; White and 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.

4.2.2.2 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.

4.2.2.3 Natural Disturbances

In many forests of the world, natural disturbances play a dominant role in forest carbon dynamics (e.g., Kurz and Apps 1999; Li et al. 2003b; Mouillet and 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.

4.2.2.4 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.

4.2.3 Estimates of Carbon Stocks and Stock Changes

Estimates of carbon stock and stock changes at the national, regional or landscapes-cale 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 and 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 and 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 and 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 and Apps 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 and Apps 1999) are being developed and refined.

4.2.4 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 and 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 and Woodward 1998). Forecasting requires models that can reflect the stand (or strata) changes, and methods to scale these up to the larger land unit.

4.3 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 aggre-

gated 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".

4.3.1 "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. 4.2). 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

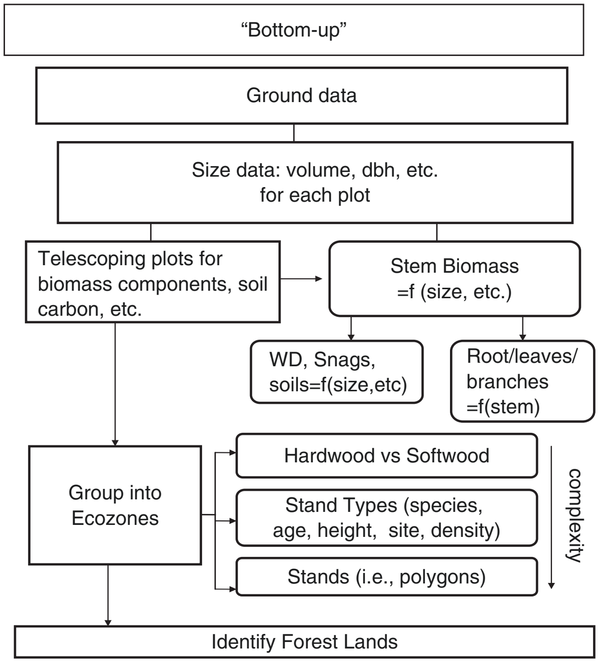

Fig. 4.2 "Bottom-up" approach beginning with an extensive network of telescoping plots, aggregated into strata, and used to delineate forest land where WD is woody debris

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 and 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 and 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 and Smith 2006).

4.3.2 "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 and 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. 4.3), whereas for other approaches, the smallest spatial scale is the pixel (Fig. 4.4).

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 (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., 1:2000) 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

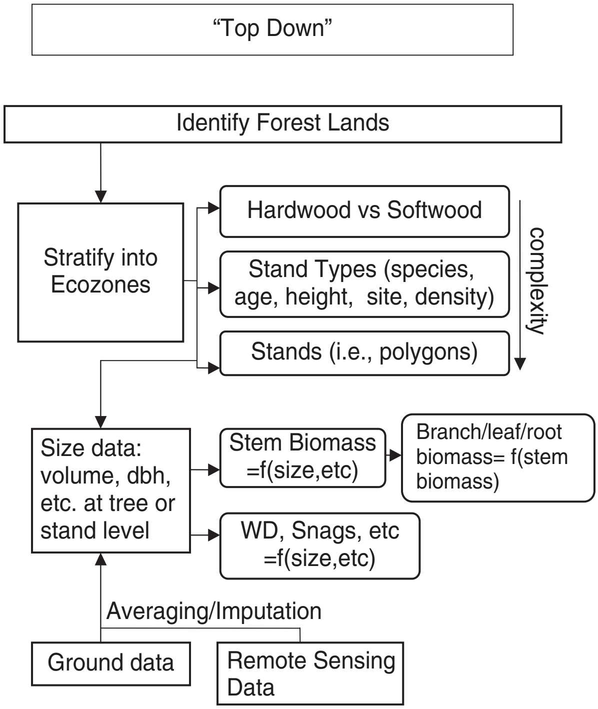

Fig. 4.3 "Top-down" approach using remotely sensed data to identify forest lands, and to stratify the forest land, coupled with ground and other remote sensing detail to obtain information for each stratum, down to the stand level of complexity

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. 4.3. 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 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

[2] Thematic mapper.

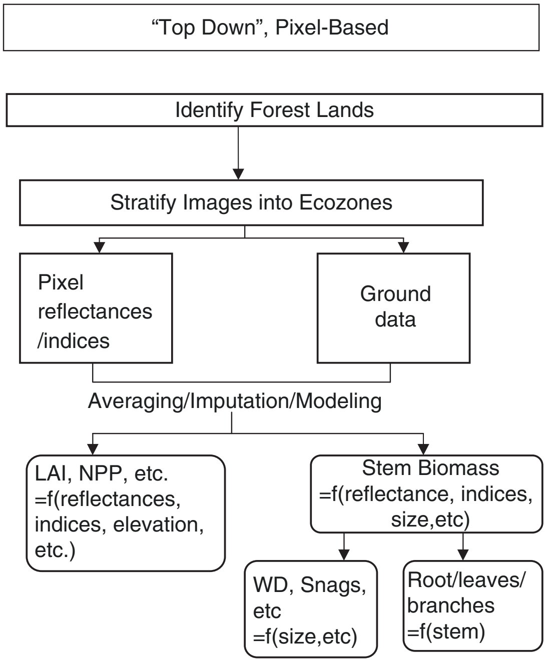

Fig. 4.4 Variation on the "Top-down" approach with a greater emphasis on remotely sensed data to obtain information down to the pixel level

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 and 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 and 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 and Temesgen 2005a). Moreover, matching ground data to strata (stands or pixels) is often difficult (Halme and Tomppo 2001; LeMay and Temesgen 2005b).

Using the general approach outlined in Fig. 4.4, 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 and Waring 2001).

[3] Light detection and ranging.

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

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 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.

4.3.3 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 and 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).

4.4 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 and Woodward 1998; Beets et al. 1999; Kurz and 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.

4.5 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.

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Chapter 5
Forest Eco-Physiological Models: Water Use and Carbon Sequestration

D. Nadal-Sala, T.F. Keenan, S. Sabaté, and C. Gracia

5.1 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, as well as the feedback and trade-off mechanisms between climate change and ecosystem carbon balances.

The study of the effects of climate change on terrestrial ecosystems is one field of interest that 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 wellbeing (such as water supply, soil fertility, bioeconomy), the effect of management on forest productivity, or in assessing the suitability of a certain site for plantation.

D. Nadal-Sala (✉)
Ecology Department, University of Barcelona, Barcelona, Spain
e-mail: D_Nadal@Ub.Edu

T.F. Keenan
Lawrence Berkeley National Lab, Berkeley, CA, USA

S. Sabaté • C. Gracia
Ecology Department, University of Barcelona and CREAF researcher, Barcelona, Spain

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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. The only model that matches exactly reality is reality itself. All models are, by definition, simplifications of reality. However, the qualitative descriptions of ecosystem responses provided by models give a very valuable insight into the functioning and potential response of the ecosystem as a whole. “All models are wrong but some are useful”, as said by Box (1979).

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 different Scots pine (*Pinus sylvestris* L.) plots in Spain, as an example of a country-wide model application, and a further analysis of the suitability of the current forest management procedures under this changing conditions. Briefly, this exercise consists in the simulation of several national inventories’ based plots to a 2100 horizon, a calibration of the GOTILWA+ output, and a simulation of the performance of Scots pines forests across Spain during twenty-first century, besides an insight of how management can change carbon balances in Scots pine forests.

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.

5.1.1 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 and non-woody products production, fire hazard reduction, etc.) that are vital for human wellbeing, as well as a net source of economic activity. 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 assess 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. Currently, forest ecosystems act globally as carbon sinks sequestrating from the atmosphere 2.4±0.4 PgC·year^{-1} and European forests contribute with a net atmospheric carbon uptake of 0.44±0.1 PgC·year^{-1}(Pan et al. 2011). However, given the rising temperatures and the increasing of water stress projected by climate change (IPCC 2013), many EU forest ecosystems may shift their behaviour from net sinks to net sources of C. Differences in projections of the effects of climate change within the EU regions are acute, with a southern Mediterranean forest ecosystems under a severe threat by the worsening its growing conditions due to the aridification of their environment, contrasting with the increase in tree growth projected in the North-European temperate forests due both a slight increase in temperature and the fertilizing effect of an increasing atmospheric CO_2 concentration.

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 forests and ecosystem service supply. Here, GOTILWA+ projections are 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.

5.1.2 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 dependent onsite 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, in abrupt and generalized mortality events under severe climatic disturbances, etc.). Models allow for the evaluation of many alternative strategies, and the effectiveness of each can thus be tested based on the requirements of the manager. This is relevant in maximising the potential for the forest to sequester carbon, in modifying forest green water to blue water balances, 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.

5.1.3 Process Based Models vs. Empirical Models

There are two main approaches available to modellers (Fontes et al. 2010): 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 both.

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 such as climate change.

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 fieldwork or lab experiments. This allows an accurate description of all factors affecting a process. However, 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: we are doomed to never succeed in designing a herring. 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.

5.2 Gotilwa+: A Process-Based Model

5.2.1 The Model

GOTILWA+ (Growth Of Trees Is Limited by WAter, http://www.creaf.uab.cat/gotilwa/) is a process-based forest growth simulation model (Gracia et al. 1999; Keenan et al. 2009a, b; Fontes et al. 2010; Nadal-Sala et al. 2014). GOTILWA+ model has been tested using data from Forest Inventories (e.g. National Forest Inventories), Eddy Flux towers outputs, as well as compared to other process-based models (see Kramer et al. 2002, Morales et al. 2005). GOTILWA+ has been successfully applied Europe-wide (see Schröter et al. 2005; Keenan et al. 2009a, b, 2010; Keenan et al. 2011; Serra-Diaz et al. 2013).

GOTILWA+ performs forest growth under different climates, stand structures, management options and soil traits. GOTILWA+ describes carbon and water fluxes through forests and has been applied on a wide range of environmental conditions, from boreal northern Europe to Mediterranean basin, and also in the Ecuadorian Andes in *Polylepis reticulata* tree species. Its programme code is built using Microsoft Visual Basic (6.0) platform.

The GOTILWA+ time step resolution is hourly, and calculations are integrated into daily, monthly and yearly values. Leaf area vertical distribution distinguishes two canopy layers (under sunny and shaded conditions) but there is no explicit description of the leaf area horizontal distribution.

Trees are grouped by DBH size classes, individuals within a DBH class are treated mostly as identical. Light extinction coefficient is estimated using Campbell's equation (Campbell 1986). Photosynthesis is calculated using Farquhar's equations (Farquhar and Von Caemmerer 1982). Stomatal conductance calculation uses the Leuning, Ball and Berry approach (Leuning 1995). Leaf temperature is determined by the leaf energy balance equations described by Gates (1962, 1980). Potential evapotranspiration is estimated by the Penman–Monteith equation (Monteith 1965; Jarvis and McNaughton 1986).

Specific tree species parameters related to photosynthetic capacity, leaf morphology and leaf hydraulic conductivity are used (taken from measurements or literature) and environmental input variables are incident radiation, wind speed, atmospheric water vapour pressure, temperature max and min, precipitation and atmospheric CO_2 concentration.

5.2.1.1 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, Interception, 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.

5.2.1.2 How Gotilwa+ Copes with Processes

Process based models start at the very basic physiological leaf level, combining and describing the different processes involved (Fig. 5.1). Figure 5.2 shows a schematic of the most fundamental compartment, the leaf. Here, photosynthesis is hourly calculated dynamically, based on internal and external conditions.

The key environmental forcing factors taken into account are precipitation, air temperature, vapour pressure, global radiation, wind speed, and atmospheric carbon

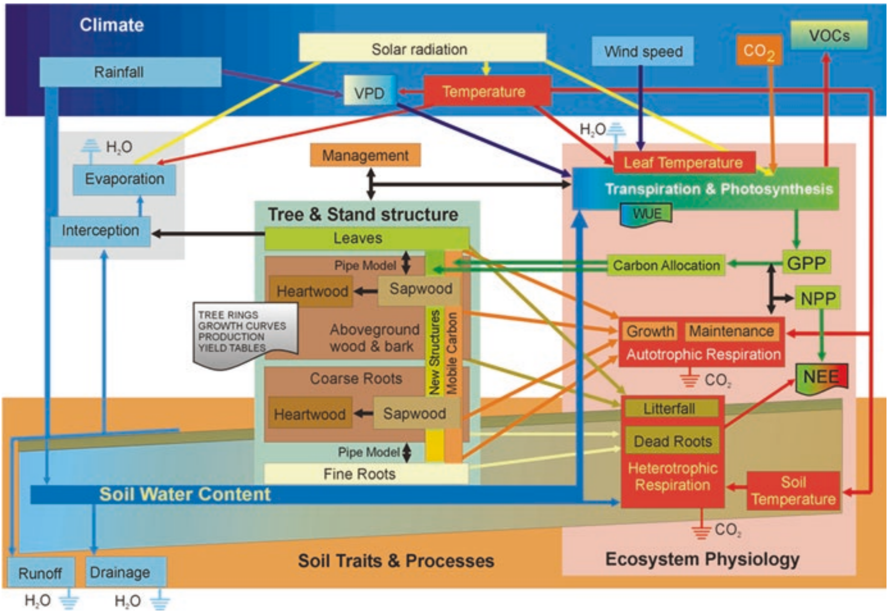

Fig. 5.1 A schematic graph of processes and interactions accounted for in the GOTILWA+ model

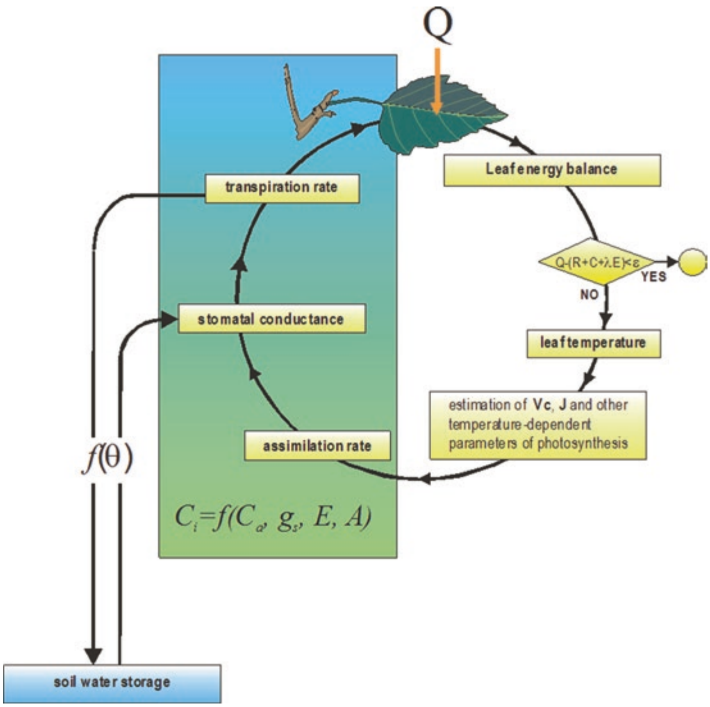

Fig. 5.2 A schematic diagram of the representation in GOTILWA+ of the photosynthetic assimilation rate

dioxide concentration. Using this data, the response of ecosystem processes is calculated to estimate the carbon and water fluxes in a forest ecosystem.

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

Net Primary Production (NPP) is obtained from Gross Primary Production (GPP) minus maintenance respiration (MR) following Eq. 5.1:

$$NPP = GPP - \left(\frac{\mathrm{M_R}}{\mathrm{E_E}}\right) \tag{5.1}$$

Where NPP and GPP are expressed in kg·hour^{-1}·ha^{-1}, MR is maintenance respiration, expressed in kcal·hour^{-1}·ha^{-1} and EE is the energetic equivalence of organic matter, assumed as a constant value of 9.4·10^3 kcal·kg^{-1}.

M_R is determined by the sum of the respiration of leaf biomass, fine root biomass and living wood biomass. Living wood biomass is a species-specific percentage of wood biomass. M_R rates depend on temperature according to a Q10 approach.

The respiration rate is 33.3 kcal·kg^{-1}·day^{-1} at 25 °C for structural carbohydrates and 55.5 kcal·kg^{-1}·day^{-1} at 25 °C for mobile carbohydrates, following Ovington (1961). Thus, M_R for a given tissue follows the Eq. 5.2:

$$M_R = \mathrm{B} \cdot Q10_t \cdot RR_c \tag{5.2}$$

M_R is the tissue maintenance respiration for a given tissue, in kg·ha^{-1}·hour^{-1}, B is the respiring biomass of a given tissue, in kg·ha^{-1}, Q10t is the value of Q10 at a given tissue temperature, RR_C is the respiration rate for a given carbon fraction – i.e. structural or mobile carbon fraction – in kcal·kg^{-1}·hour^{-1}.

NPP is then allocated through the tree compartments following a set of hierarchical decision criteria (Fig. 5.3). First, NPP refills tree mobile carbohydrates (MCH) reserves up to the maximum replenishment values. Then NPP is used to equilibrate according to the pipe model leaf area, fine root biomass and sapwood area (Shinozaki et al. 1964). When new tissues are produced, carbohydrates are also spent on growth respiration (G_R). G_R is set as 32 % of the invested carbohydrates for growth – i.e. a constant efficiency of 0.68 g of new tissue per g of carbohydrate (Ovington 1961). Finally, if there is still NPP available , trees generate new sapwood area, new leave area and new fine root biomass according to the pipe model and accounting for GR costs as above.

When there is no photosynthetic activity or assimilated carbon is not sufficient to compensate respiration rates, NPP values turn to negative. If so, the lack of photosynthesis is offset by MCH reserves. When mobile carbon pool is fully available, it can be depleted without consequences for tree population. When MCH reserves fall close to the mortality threshold, trees lose specific respiring tissues such as leaf and fine roots biomass. If carbon starvation continues and MCH falls below a certain species-specific threshold, a tree mortality event occurs. Mortality also occurs if the

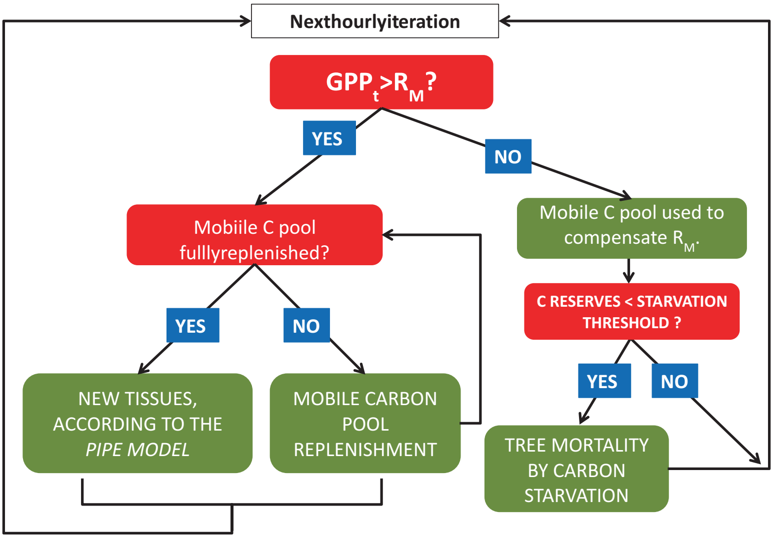

Fig. 5.3 Flow diagram of carbon allocation hierarchical criteria followed by GOTILWA+. *Red boxes* represents hierarchical questions. *Green boxes* represent main processes

diametric class is completely defoliated, and MCH is not available anymore, during the vegetative period in the case of deciduous tree species or at each moment of the year for the evergreen ones.

GOTILWA+ does not consider a homogeneous distribution of mobile carbon reserves within a DBH class. Concerning mortality, GOTILWA+, instead of working with the DBH class average tree, assumes differences within trees in the same DBH class. As the pool of MCH is not homogeneously distributed, there are trees in a better condition than others. The number of dead trees is established as follows: The number of trees that can be sustained by the current MCH value is calculated. Then, the difference between the current number of trees and the number of trees that can be sustained is the mortality within the DHB class. In addition to the MCH, the rest of the tree structure compartments are restructured accordingly.

Fine litter fall (e.g. leaves), gross litter fall (e.g. bark, branches, dead stems) 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 fixed rate of transfer between them. Soil organic carbon is decomposed depending on to which layer it belongs, with both decomposition rates depending on a Q10 approach taking into account soil water content and soil temperature. Soil temperature is calculated from air temperature using a moving average of 30 days. The amount of soil water available for organic layers is calculated taking into account the cumulated rainfall of the previous 30 days. Soil water availability for mineral layers depends on the soils water filled porosity that in turn is a function of the organic matter present in soil.

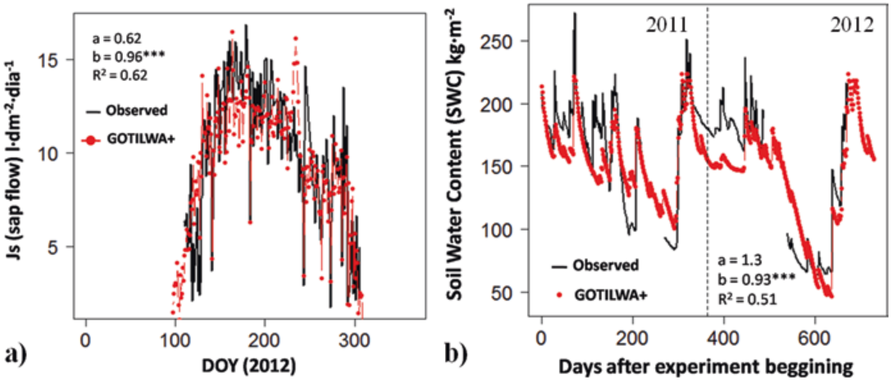

Fig. 5.4 Modelled gotilwa+ versus measured "in situ" values for (**a**) daily sap flow density (J_s, in $l \cdot dm^{-2} \cdot day^{-1}$) in the el regàs *Fraxinus excelsior* spanish forest study plot, and (**b**) daily soil water content (SWC, in $kg \cdot m^{-2}$) for the same plot

Soil water content is described as one layer, taking inputs though precipitation less leaf interception, which is evaporated, (stem interception, or stem flow, is not evaporated, but directed to the soil), and outputs though drainage, runoff, and transpiration.

5.2.1.3 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, Schröter et al. 2005; Keenan et al. 2009a, b, 2010, 2011; Serra-Diaz et al. 2013), using canopy level measurements gathered by the FLUXNET network. Figure 5.4a shows the results at one site, a *Fraxinus excelsior* riparian forest in the northern of Spain, providing simulated GOTILWA+ daily sap flow density values against 1 year of field data (2012) measured values. Besides, comparison between modelled and measured soil water content for the same plot during 2 years (2011, 2012), is noted in Fig. 5.4b.

The model successfully matches the sap flow density, with a R^2 of 0.62 and a slope of 0.96. It also captures both the trend of high sap flow values observed in spring and the decrease in sap flow density during summer drought. Besides, the soil water content (SWC) pattern follows the same trends in both 2011 and 2012, with a summer depletion of SWC, followed with an autumn refilling matching the end of the growing season and the beginning of the autumn rains.

5.2.1.4 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, 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, cambium cell division) interact with slow processes (tree ring formation, sapwood to heartwood changes, tree mortality, cavitation, wood production, management, soil organic matter 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.

5.3 Model Applications

5.3.1 Carbon Balances and Water Cycles in Spanish Forests

5.3.1.1 GOTILWA+ Calibration Against National Forest Inventories Data

As an example model application, we assess results from model simulations at 64 Spanish sites. Those sites have been randomly sampled from the Spanish National Inventory Data (IFN). The sampling criteria used was: (a) The exercise is focused in Scots pine *Pinus sylvestris* pure stands, (b) we consider pure stands those in which 80 % of total basal area is sustained by Scots pine, as well as the number of trees in the plot was greater than 200 trees·ha^{-1} during the IFN2 (year 1995), (c) initial sample was formed by 100 random plots. From these plots, managed sites or that experienced forest fires during the IFN2 (year 1995) – IFN3 (year 2005) interval have been removed.

Due to lack of information about soil conditions, three simulations under three different soil characteristics (soil depth of 0.25, 0.5, and 1 m) were run under observed (Ninyerola et al. 2007a, b) climate conditions in each plot. Simulated basal area

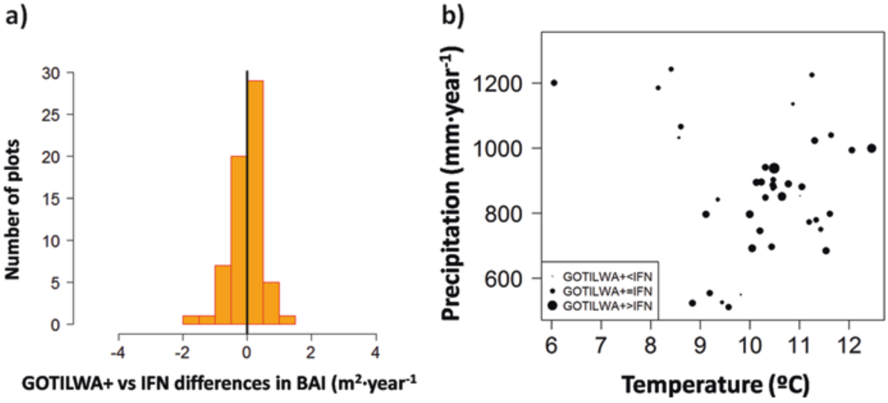

Fig. 5.5 (**a**) Differences in basal area increment bai ($m^2 \cdot year^{-1}$) for the 64 *P. sylvestris* plots between GOTILWA+ best fit simulation and ifn2-ifn3 inventories. Positive values indicates overestimation in GOTILWA+ simulations, and negative values indicates underestimation of GOTILWA+ simulations. (**b**) Plot distribution for the 1971–2010 reference period mean yearly precipitation and mean yearly temperature axis. The size of the *dot* indicates the degree of deviation from the ifn predictions. Thus, *smaller dots* represent underestimation in GOTILWA+ simulations, and *bigger plots* represent overestimation in GOTILWA+ simulations

increment for each plot was compared to the IFN2-IFN3 basal area measured increment, and the soil depth that minimizes the difference between the observed and modelled basal area increments was selected for each plot. According to the Fig. 5.5a, Root Mean Square Error (RMSE) between observed and simulated basal area increment was 0.473, with 51 % of plots within the ±0.25 $m^2 \cdot ha^{-1} \cdot year^{-1}$ range, and 77 % of the plots within the ± 0.5 $m^2 \cdot ha^{-1} \cdot year^{-1}$ range. Besides, there is no precipitation related trend in differences between observed and modelled basal area increments 5b. However, GOTILWA+ systematically slightly overestimates basal area increments in warmer plots.

5.3.1.2 Future Climate Uncertainty: Coupling Gcm's and Socio-Economic Forcing

Projecting forest growth into the future is highly dependent 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 behaviour 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 climate forcing scenarios are derived from the IPCC (2013): RCP8.5, RCP 6.0, RCP4.5, and RCP2.6, ranging from pessimistic to optimistic regarding future anthropogenic impact on the climate system. As this exercise was run before the new IPCC (2013) climate forcing scenarios, we used A2 and B2 socioeconomic forcing scenarios from IPCC 2007. They both match pretty well with RCP 6.0 and RCP 4.5 CO_2 emissions respectively.

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. 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. In this exercise, an ECHAM4 GCM was considered, obtaining two possible combinations of scenario projections: ECHAM4-B2 and ECHAM4-A2.

5.3.1.3 Stand Performance at the Selected Sites

Here, to ease the interpretation of the results, we present results from the ECHAM4 model predictions with the A2 and B2 emission scenario as our description of future climate (this gives mid-range levels of future climate change). Regarding to carbon balances, Fig. 5.6 shows the mean Gross Primary Production (GPP) and mean Net Ecosystem Exchange (NEE) predicted by the model for all 64 plots used under the two socio-economic scenarios. In both cases, an increase in GPP can be observed, resulting from higher temperatures and CO_2 fertilisation. This increase in GPP is greater in A2 scenario than in B2 scenario. However, NEE increases along the twenty-first century for both climates change scenario projections. Thus, resulting in a reduced carbon sink capacity of Scots pine forests in Spain.

With regard to forest water balances, Fig. 5.6 shows that the projected fraction of the precipitation transpired by the forest would increase during the twenty-first century and drainage and runoff would decrease downstream. This decrease of water availability would imply relevant consequences for downstream ecosystems, as well as for human societies' water resources. That would be exacerbated under the higher CO_2 emissions scenario (ECHAM4-A2) than under the lower one (ECHAM4-B2).

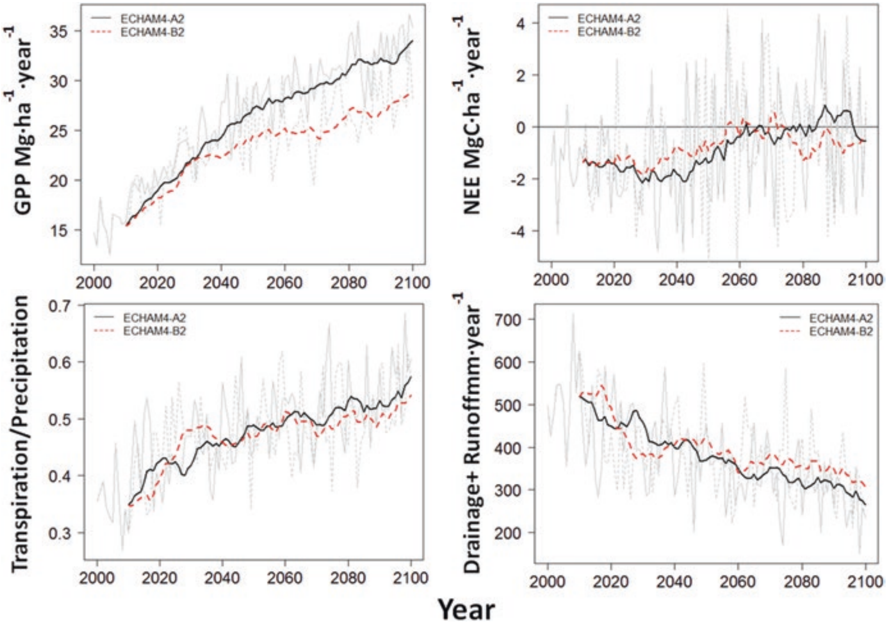

Fig. 5.6 Gotilwa+ projections for the twenty-first century forest ecosystem carbon and water balances. Mean gross primary production (gpp, Mg·ha^{-1}·year^{-1}), net ecosystem exchange (nee, mgc·ha^{-1}·year^{-1}), fraction of the precipitation transpired by tree population (unit less), and drainage + runoff exiting downstream the plot (mm·year^{-1}) for all sites are indicated between 2000 and 2100, both for the a2 and the b2 socio-economic scenarios. *Grey lines* represent the yearly mean values for the 64 plots, and *black* and *red lines* represent the 10 years average trend for the a2 and b2 socio-economic scenarios

5.3.2 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 mitigate the effects of drought on forest stands. Here, an example is given of a management strategy applied to a typical Scots pine Mediterranean plot. The management strategy in this simulation is to enter the forest when the basal area reaches 42 m^2·ha^{-1} and remove the larger trees until a basal area is reached of 38 m^2·ha^{-1}. 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 CO_2. This strategy can be contrasted against alternatives, and an optimum strategy found, depending on the prerequisites of the user.

Figure 5.7 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

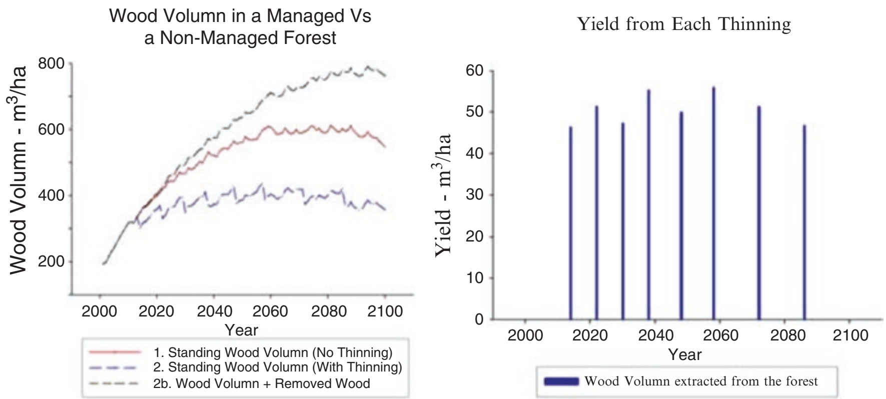

Fig. 5.7 *Left*: GOTILWA+ projection of wood volume remaining in an unmanaged forest and in the same forest with management. *Right*: The wood volume extracted from the managed forest at each intervention in the simulation

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 dependent on the product, thus any additional sink that results from the extraction of wood from the system can be presumed to be short lived.

5.3.3 The Future of European Forests – Europe Wide Simulations

GOTILWA+ has been validated extensively in Europe (Kramer et al. 2002, Morales et al. 2005, Schröter et al. 2005; Keenan et al. 2009a, b, 2010, 2011; Serra-Diaz et al. 2013). This has helped to 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, and providing that ECHAM4 GCM was not available all over the UE, we chose the HadCM3 IPCC (2007) scenario to simulate future forest stands over Europe.

Here, in Fig. 5.8, 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 Spanish plots in Fig. 5.6. 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.

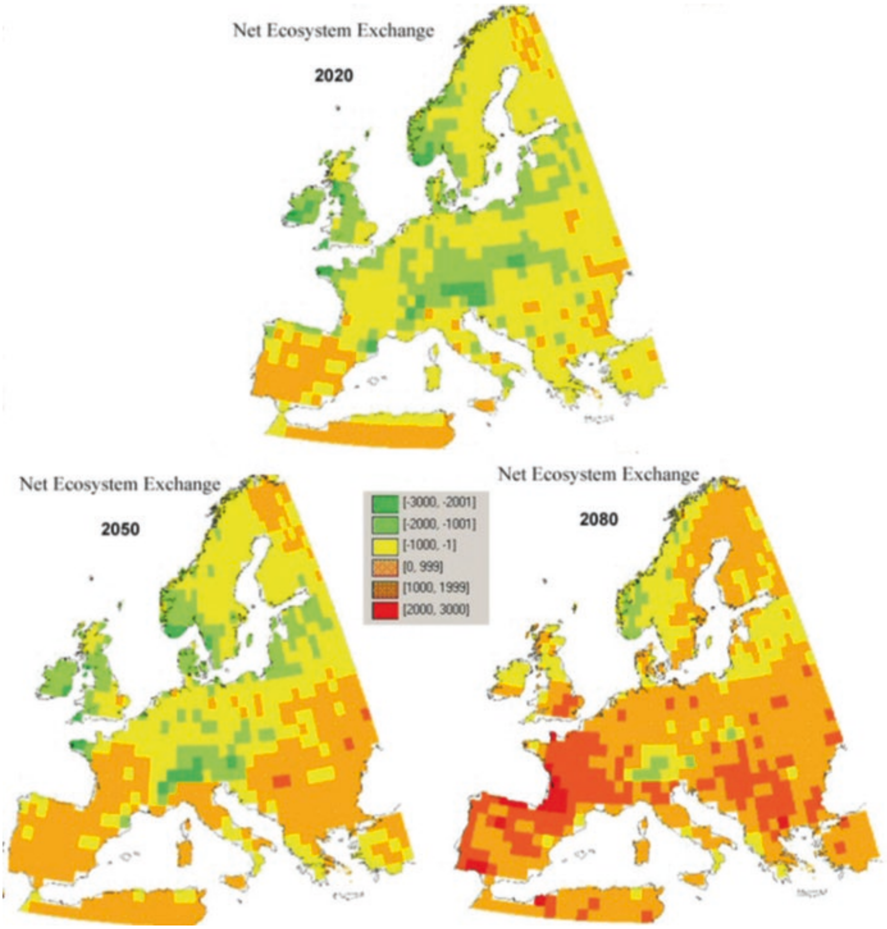

Fig. 5.8 GOTILWA+ projections of net ecosystem exchange (kg/ha/year) for *P. sylvestris* forests in europe-wide simulations (values represent the annual average for each time slice 2020: 2010 to 2030, 2050: 2040 to 2060, and 2080: 2070 to 2090)

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

Figure 5.9 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, Janssens et al. 2005), though undoubtedly further work is required to reduce our uncertainty. As exact future climate scenario is still unknown and the soil respiration processes are not fully understood, quantitative conclusion may be questionable, but the qualitative conclusion should not vary greatly with a better understanding of the processes involved.

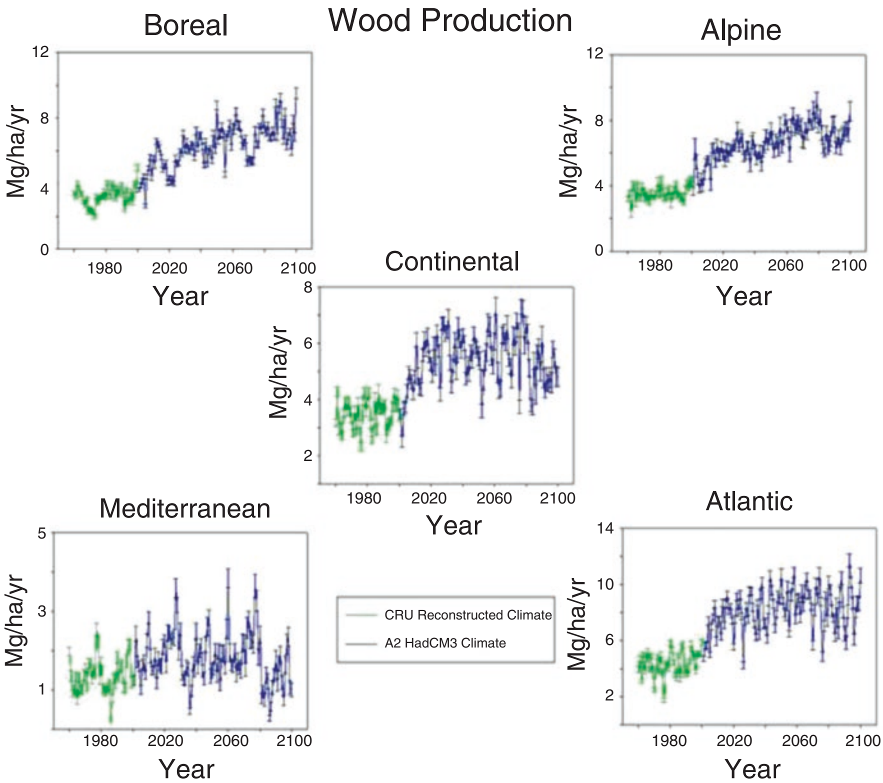

Fig. 5.9 GOTILWA+ projections of wood production (mg/ha/year) for five main european climate regions from 1960 to 2100 under had CM3 socio-economic scenario

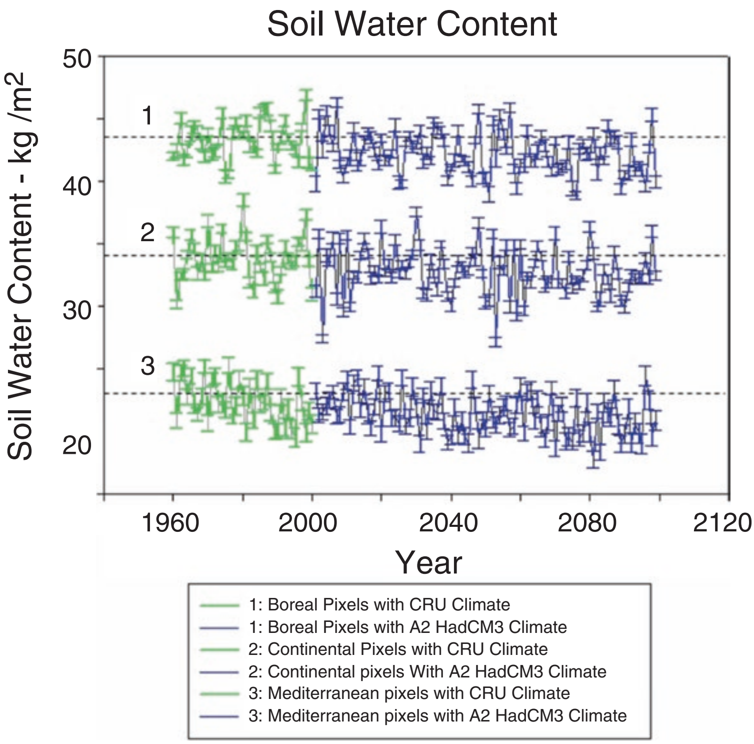

Fig. 5.10 GOTILWA+ projections of the quantity of water in the soil for three climate types, boreal, continental, and mediterranean climates, with reconstructed climate from 1960 to 2000, and the hadley center had CM3 model with scenario a2 until 2100

In Fig. 5.10, 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.

5.4 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.

5.4.1 *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 affect the distribution and functioning of vegetation, but vegetation also has an effect on climate, and the two are inextricably linked. This feedback mechanism is now recognised as being crucial to the evolution of the Earth's climate (Bonan 2002), and equally crucial in predicting the anticipated change in the earth's 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.

5.4.2 *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 acclimation; 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, very frequent in the Mediterranean environments, and often poorly integrated in forest dynamic models.

Besides, finding truthfully parameters to feed the model to run is a key issue when process-based models are run. Considering that GOTILWA+ uses more than 100 simulation parameters, there is a huge need for robust and reliable parameterization techniques. Innovative iterative parameterization techniques, such as Bayesian or neural network approaches, would help us to improve model functioning and applicability, as well as to determine accurately the strengths and weakness of each model.

5.5 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), and local intensive experimental stand measurements, such as the El Regàs MED-FORESTREAM plot) 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. Besides, new parameterization and calibration techniques, such as Bayesian parameterization, are gaining importance in modelling. Those techniques are going to help us to improve the accuracy of the model projections.

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.

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Chapter 6
Influence of Climatic Variables on Crown Condition in Pine Forests of Northern Spain

A.V. Sanz-Ros, J.A. Pajares, and J.J. Díez

Abstract 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. Greenhouse gasses emission has been modifying global climate, affecting ecosystems in very diverse ways (FAO 2012). Current mitigation policies derived from Kyoto Protocol are following two main strategies: reduction of gas emissions, and implementation of a sustainable development assuring persistence of greenhouse carbon sinks, including forests lands.

6.1 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. Greenhouse gasses emission has been modifying global climate, affecting ecosystems in very diverse ways (FAO 2012). Current mitigation policies derived from Kyoto Protocol are following two main strategies: reduction of gas emissions, and implementation of a sustainable development assuring persistence of greenhouse carbon sinks, including forests lands.

Some 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),

A.V. Sanz-Ros (✉)
iuFOR - Sustainable Forest Management Research Institute (UVA-INIA), Plant Production and Forest Resources Department, Universidad de Valladolid, Avenida de Madrid 44, 34004 Palencia, Spain

Forest Health Centre of Calabazanos (Junta de Castilla y León), Polígono Industrial de Villamuriel, S/N. 34190. Villamuriel de Cerrato, Palencia, Spain
e-mail: asanzros@gmail.com

J.A. Pajares • J.J. Díez
iuFOR - Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain

Departamento de Producción Vegetal y Recursos Forestales, ETS de Ingenierías Agrarias, Universidad de Valladolid, Palencia, Spain
e-mail: jdcasero@pvs.uva.es

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although the fifth assessment report of IPCC stated that the effects of climate change will remain for centuries, even if we stop emitting greenhouse gasses now (IPCC 2013). Sustainable forest management is an essential tool to assure the permanence of our forests and to maintain properly their ecological functioning. They can offer a permanent carbon sink by the technological option of capturing carbon from biomass conversion facilities (Kraxner et al. 2003). The role of forest as a CO_2 sink could be influenced by the occurrence of forest pests and diseases causing tree defoliation and canopy reduction. In this sense climate change effects can increase the frequency of insect outbreaks due to drought and warming (Allen et al. 2010). 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 CO_2 sequestration depends on its crown development. Crown condition can be evaluated though the estimation of defoliation. 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 and UN/ECE 2000). There have been some criticism about the use of this indicator, but the consistency of this evaluation along time has been recently demonstrated (Eickenscheidt and Wellbrock 2014).

The term defoliation is defined as the defoliative effects of biological agents, premature needle loss or reduction in the needle holding period (Ferretti 1994). This parameter does not takes into consideration factors like unusual reduction of leaf size, presence of flowers and cones, branching deformation or shoot death (Ferretti 1994). Estimation of defoliation, 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). 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 (drough) 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 climatic parameters are the most determining for each region. It is expected that variation of the climatic factors would be different among the diverse regions in future climate change scenarios, as showed by Andreu et al. 2007, so it is 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 and Rapp 1997; Piervitali et al. 1997; Buffoni et al. 1999; Brunetti et al. 2000, 2001; Sarris et al. 2007). It seems that, in Spain, annual rainfall shows a trend towards a decrease over the whole Iberian Peninsula, the greatest decreases occurring in summer, while the winters will become wetter (Karas 1997; Esteban-Parra et al. 1998; Hulme and 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 and 0.8 °C along the twentieth 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 nineteenth century, due to either anthropogenic (IPCC 2001) or to astronomic causes (Landscheidt 2000; Soon et al. 2000). Some studies pointed that the decade 1995–2006 was the warmest record ever registered (IPCC 2007), but global mean surface temperature trends are calculated using much longer periods, since it exhibits substantial decadal and interannual variability, showing an increase of 0.85 °C over the period between 1880–2012 (IPCC 2013).

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. In Europe, projections for year 2100 an increase of 0.3–4.8 °C (IPCC 2013), whereas previous models have forecasted approximately an increase of 1.5–3 °C (Kattenberg et al. 1996), or between 1 and 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 100 years, with highest increases in summer (approximately 2 °C) and the lowest in winter (Hulme and Sheard 1999; Parry 2000; Prieto et al. 2004). This change is also reflected in the behavior of the extreme values, which depends on local conditions, showing significant trends in some regions of the globe, but not in others, where no significant changes were detected (De Gaetano 1996; Heino et al. 1999; Bonsal et al. 2001).

In some respects, these climate changes are likely to act as an important driving force on natural systems (Parmesan and Yohe 2003). The increase of temperature along the next 100 years would be equivalent to a pole ward shift of the present geographic isotherms of approximately 150–155 km (Watson et al. 1997), causing changes in forest tree species distribution and limits. Risk of pests and diseases will be increased due to these limits displacement, so that many forested surface will be placed in a stressing environment. In this scenario, tree vigour of these species will decrease, leading to canopy decline manifested in symptoms as defoliation and discoloration.

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 (defoliation), and to contribute to the understanding of how these parameters will affect under future climate change scenarios.

6.2 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. 6.1), and were covered by three *Pinus* species (37 by *P. sylvestris*, 22 by *P. nigra*, and 9 by *P. pinaster*).

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 186642 ha, 60000 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 1000 m.a.s.l. (Fig. 6.1). The climate is Mediterranean with a slight Atlantic influence, with a mean temperature of 11.49 °C and an annual rainfall of 519 mm.

Sampling method involved four subplots (Fig. 6.2). One fixed subplot of 25 m radius (National Forest Inventory plot) and three subplot of 17.5 m radius, linked by two orthogonal linear transects of 50 m. In each subplot, the 20 nearest trees in a spiral pattern were evaluated.

Defoliation was estimated visually in the field according to the European Programme for the Intensive Monitoring of Forest Ecosystems, Level I (ICP forests 1992). The visual estimation of the amount of light passing through the tree live crown was evaluated in 5 % intervals and compared with a reference tree with complete foliage. Reference pictures of defoliation degrees for each species in Mediterranean areas were used in this comparison (Cadahia et al. 1991; Ferretti 1994). The assessment was done by two different evaluators at a distance equal to tree height and avoiding to face the sun.

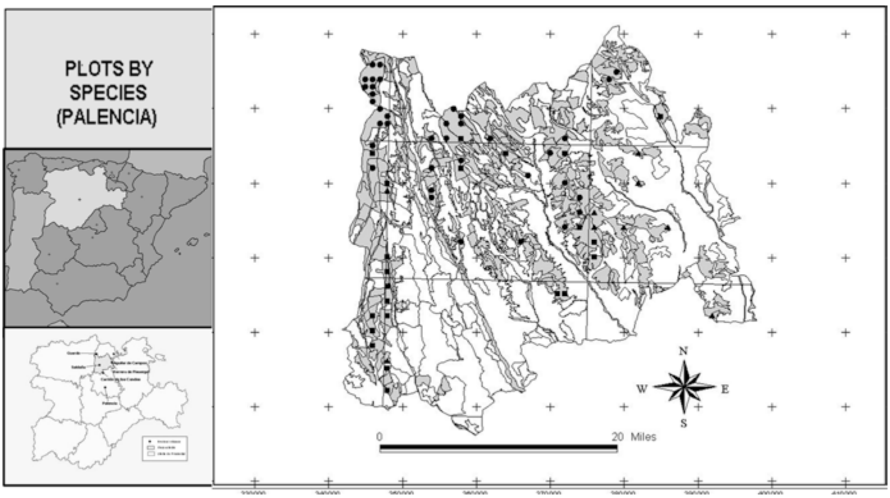

Fig. 6.1 Distribution of plots in pilot zone in Palencia Province, Castilla y León, Spain (Plots were taken from a 2 km grid on tree covered area. Pinus sylvestris (●), *Pinus pinaster.* (▲), *and P. nigra* (■).*Gray surface is forest covered area*)

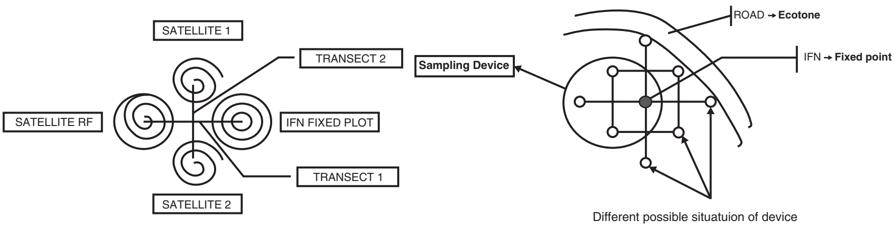

Fig. 6.2 Sampling method with four subplots and two linear transects linking them (*Left*). IFN plot is a fixed plot, original transect orientation was N-S and E-W, but it wasable to be rotated in order to avoid roads or firewalls *(Right)*

Table 6.1 Likely predictor climatic variables used to find correlations with plot defoliation

	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						

Data showed for plot defoliation 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 shown that there were no differences in defoliation between inside stand trees and edge trees (Durrant and Boswell 2002).

Climatic long-term data for each plot were obtained from the Digital Climatic Atlas of Iberian Peninsula (Ninyerola et al. 2005), a 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 means of the last twenty years, and temperatures to means the last fifteen previous years. Several climatic variables were chosen (Table 6.1) 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 6.2) with the aim of comparing plot defoliation among different levels of each climatic variable.

Plot defoliation 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 plot defoliation among different levels of rainfall, temperatures and solar radiation. Finally, the Bonferroni test was used for multiple comparisons. To study the relationship between defoliation and climatic data, simple regression 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 defoliation.

Table 6.2 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 pilot zone

	Cat	Annual	December	January	February	June	July	August
Mean temperature (°C)	1	9–9.4	2.4–2.76	1.6–1.94	2.7–3.1	14.2–14.74	20.7–21.42	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 (°C)	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 (°C)	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 radiation Kj/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	–	–	–	–	–	–

6.3 Results

The Kolmogorov-Smirnov, Shapiro-Wilks and Bartlett tests proved normality and homocedasticity of the logarithm of mean plot defoliation (Def) data. The One Way ANOVA analysis showed that there were statistically significant differences between mean Def values of plots with different levels of July rainfall, Mean annual temperature and Mean august temperature, whereas there were no significant differences in plot Def in relation with the other variables analyzed, such as solar radiation, maximum and minimum annual, summer and winter temperatures or annual and winter rainfall (Table 6.3).

In the analysis of rainfall, ANOVA showed significant differences in plot defoliation among different homogeneous levels of July rainfall (Fig. 6.3, Top). Bonferroni test showed differences between levels 1 and 5 (Table 6.4), with a difference in precipitation of 60 mm. The erratic behavior of rainfall distribution (Dore 2005) could preclude from making accurate predictions for the future.

On the other hand, there were significant differences in plot defoliation among Mean annual temperature levels (Fig. 6.3, Medium). Duncan multiple comparison test demonstrated significant differences ($p < 0.05$) between defoliation in level 1 and 5 (Table 6.5), 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 defoliation among plots with different levels of Mean August temperature also showed significant ($p < 0.05$) differences (Fig. 6.3, Bottom). Bonferroni multiple comparison test revealed differences between levels 1 and 5 (Table 6.6), with a variation in August temperature of 2.48 °C.

Table 6.3 Results of ANOVA between plot defoliation and 5 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	2.736
	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						

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

*Abbreviations: *Temp.* temperature, *Max.* Maximum, *Min.* Minimum, *Rad.* Radiation

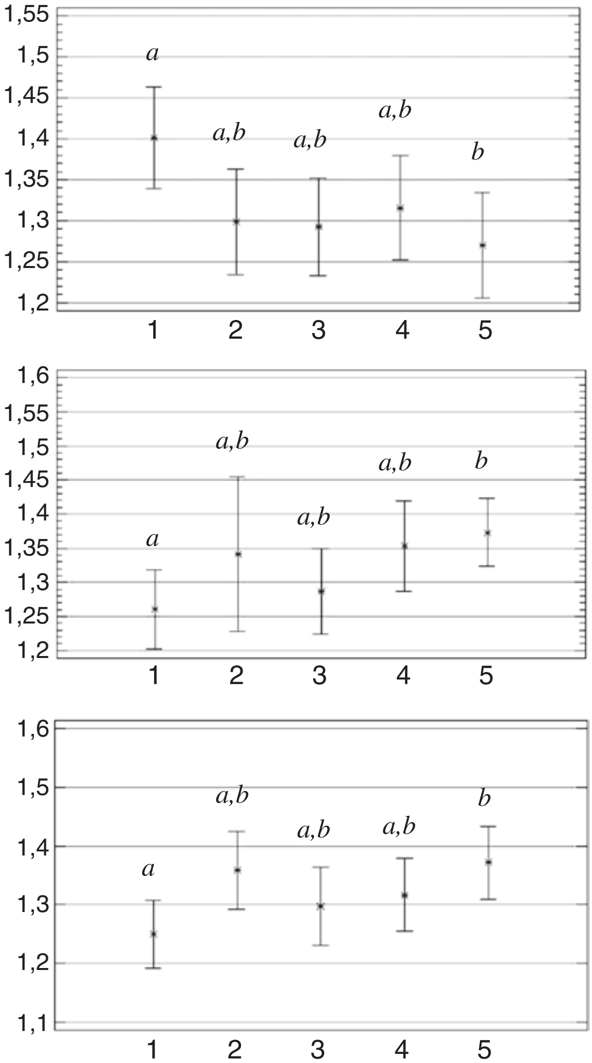

Fig. 6.3 Confidence intervals of ANOVA analysis among the logarithm plot defoliation and July rainfall (*Top*), mean annual temperature (*Medium*), and mean August temperature levels (*Bottom*). Each variable was categorized in 5 homogeneous intervals. *Letters* indicate significant differences in defoliation among levels of climatic variables

Table 6.4 ANOVA between logarithm of defoliation and 5 July rainfall levels

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**

There were significant ($p < 0.05$) differences between defoliation in level 1 and 5, as it is showed by Bonferroni multiple comparison test

Table 6.5 Bonferroni multiple comparison test results, which demonstrates significant differences in plot defoliation 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	**ab**
4	16	1.35964	**ab**
5	4	1.41152	**b**

Table 6.6 Tukey multiple comparison test, showing significant differences in plot defoliation 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 defoliation 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 defoliation, a simple regression was done for each of the significant parameters. The regression models for defoliation versus July rainfall, Mean annual and August temperature were significant ($p < 0.05$), with a negative slope for the precipitation model and positive for both 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. 6.4), suggesting that defoliation was not only explained by climatic parameters, however there was an evident influence of these parameters on the observed defoliation.

Multiple regression analysis with backward selection showed that July Rainfall was the variable with higher effect on defoliation. However, although the model is significant, R^2 was quite low, with only 10.66 %. A multiple regression model was built, but it is not improving R^2_{adj} of models with only one climatic variable.

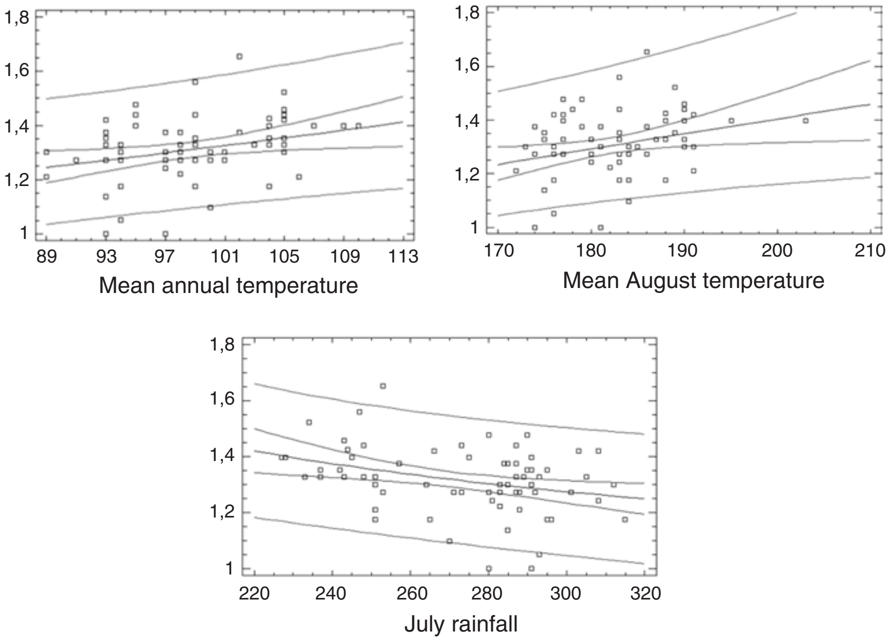

Fig. 6.4 Regression models for defoliation and climatic parameters (mean annual and mean august temperatures and July rainfall). Reciprocal-Y regression model related to Mean temperature [log Def = 1/(1.42482–0.000412949 * Mean temperature)](*Left*). Double reciprocal model related to Mean August temperature [log Def = 1/0.155,929 + 111.238/Mean August temperature] (*Right*). Reciprocal-X regression model related to July rainfall [log Def = 0.872,209 + 120.684/ July rainfall] (*Center*)

6.4 Discussion

6.4.1 Current State

According to ANOVA results, a negative deviation in July rainfall of 60 mm led to a significant increase of defoliation (about 10 %). Precipitation started to decrease in June, but temperatures remained at moderate levels. In July temperatures were higher but rainfall was much lower, thus 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 tolerance 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 temperature also influenced defoliation. As it was showed by the ANOVA analysis, an increase of 1,6 °C (between levels 1 and 5) caused an increase of 8 %. This increment of temperatures corresponds exactly with the observed increase of temperatures in the Iberian Peninsula over the last hundred years (Hulme and Sheard 1999). Plots with higher mean temperatures reached higher defoliation values, mainly when rainfall decrease was more acute. In addition, mean August temperature also was influencing canopy, as there was an increase in defoliation (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 and Gibbs 1996; Houston 1998).

A pattern in temperature and precipitation was observed in relation to plot defoliation in the pilot zone. Thus, most of plots that were placed in warmer and drier conditions showed higher defoliation values, and those that were located in colder and wetter sites showed lower defoliation values. This agrees with results of the Programme for the Intensive Monitoring of Forest Ecosystems in Europe that showed that between 30 and 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 abiotic and biotic 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, although it has been proved to be very low. Thus, more reliable methods are being developed for crown assessment. Most of those methods are based on indirect measures of light environment, and they have become more widely accepted, such as remote sensing based methods (De Santis and Chuvieco 2009; Somers et al. 2010), airborne laser methods (Solberg et al. 2006) and ground-based methods. Some ground-based methods use canopy light interception to estimate leaf area optically, e.g. by hemispherical photography (Chen et al. 1991; Kucharik et al. 1998; Valladares and Guzmán 2006; Montes et al. 2007).

Results obtained from this study suggest that defoliation 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 and past climatic conditions. Although the influence of these climatic variables on defoliation is demonstrated, other kind of variables must be included with the aim of predicting defoliation for a particular site, such as silvicultural, structural, nutritional and disturbance parameters. Even other factors, such as the presence of pest and diseases should be required.

6.4.2 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 and Sheard 1999; Parry 2000; Del Río et al. 2004). Therefore, the results of this study suggest that defoliation 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 2 °C increase in temperature values, defoliation could increase about 12 %, which may represent an important reduction of canopy. A rise of maximum temperatures could also 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).

Future projections predict that rainfall will be more erratic (Dore 2005) with a decreasing trend in Spain (Schönwiese and Rapp 1997), so it is likely that defoliation levels become higher due to this decrease. In Mediterranean ecosystems, summer is a marked dry season, and at the end of the summer there is already a notable reduction in the canopy of pine plantations (Bryant et al. 2005), so an increase of drought may produce critical effects on forest health. In addition, 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 parameters are needed for building predictive models of crown condition, which could help to understand which are the main factors involved in this process. These surveys will be useful to help forest managers to minimize canopy reduction and to ensure the permanence of our forests in a good condition, thus helping to mitigate climate change effects over the atmosphere.

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Chapter 7
Changing Trends of Biomass and Carbon Pools in Mediterranean Pine Forests

Cristina Gómez, Joanne C. White, and Michael A. Wulder

Abstract The amount of biomass in forest ecosystems is critical information for global carbon cycle modelling. Determination of forest function as a sink or source of carbon is likewise relevant for both scientific applications and policy formulation. The quantity and function of forest biomass in the global carbon cycle is dynamic and changes as a result of natural and anthropogenic processes. This dynamism necessitates monitoring capacity that enables the characterization of changes in forest biomass over time and space. By combining field inventory and remotely sensed data, it is possible to characterize the quantity of biomass for a single date, or to characterize trends in quantity and function of forest biomass through time. Field inventory data provides accurate information for calibration of spatially extensive remotely sensed data models and for model validation as well. Historical, repeat measures of the same field plots facilitate the estimation of temporal trends in biomass accrual or removal, as well as carbon pooling processes. Remotely sensed data enable the inference of trends over large areas, and historical data archives can support retrospective analyses and the establishment of a baseline for future monitoring efforts. This chapter describes some of the opportunities provided by synergies between field measures and remotely sensed data for biomass and carbon assessment over large areas, and describes a case study in the Mediterranean pines of Spain, in which biomass and carbon pooling for the period 1984 to 2009 are estimated with a time series of Landsat imagery supported with data from the Spanish National Forest Inventory.

Synergistic assessment with Landsat time series and forest inventory data in the Spanish Central Range

C. Gómez (✉)
Sustainable Forest Management Research Institute, Universidad de Valladolid & INIA, Valladolid, Spain

Department of Geography and Environment, School of Geoscience, University of Aberdeen, Aberdeen AB24 3UE, Scotland, UK
e-mail: c.gomez@abdn.ac.uk

J.C. White • M.A. Wulder
Canadian Forest Service (Pacific Forestry Centre), Natural Resources Canada, Victoria, BC, V8Z 1M5, Canada
e-mail: joanne.white@canada.ca; mike.wulder@canada.ca

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7.1 Introduction

Forests play an important role in the terrestrial carbon budget (Le Quéré et al. 2015), by means of the biomass stock and carbon fluxes involved in photosynthesis and respiration. Forests are the most important land carbon sinks (Le Quéré et al. 2009), up-taking the equivalent to ~26 % of human fuel emissions (Pan et al. 2011). Aboveground biomass (AGB), the amount of living or dead material over the ground, comprises the main carbon pool in forest ecosystems. AGB is also significant for its potential as source of timber and production of bio-energy (Smeets and Faaij 2007). Biomass amount per surface unit indicates the condition and productivity of a forest (Hall et al. 2006) and it is associated with biodiversity and other ecological benefits. Assessment of forest AGB and its spatiotemporal dynamics is important to inform sustainable forest management (Herrero and Bravo 2012), for ecological applications (Barlow and Peres 2004), direct carbon accounting (Houghton 2005), for providing information in support of carbon markets (Goetz et al. 2009), and for national and international reporting commitments (Andersson et al. 2009).

The global carbon balance is markedly altered by the extent of forests, as well as the biomass content per surface unit (Houghton 2005). The character of forests as a sink or source of carbon dioxide is determined by the ratio of respiration to net primary production (Law et al. 1999), and this relation is strongly influenced by the stand successional stage (Odum 1969) and health condition (Brown 2002). Net ecosystem carbon balances are complex and multifaceted, resulting in evaluation difficulties (Schulze et al. 2000). To reduce complexity, a simple rule for above-ground forest components is that mature stands are more stable stocks of carbon and growing stands are net sinks of carbon (Goetz et al. 2006), but the age at which a forest becomes a net carbon sink varies according to forest type, site productivity and other factors (Goward et al. 2008). However, while the ability to capture carbon can be difficult to determine, the stocking magnitude of a forest stand is known to be proportional to the biomass it stores (Masera 2003; Houghton 2007). Maps depicting the dynamics of the distribution of biomass content and successional stages of forests through time are invaluable for spatially explicit assessment of forest carbon stocks, sinks and sources (Powell et al. 2010). Together with a timeline of change events, the effectiveness of various management approaches can be evaluated (Hayes and Cohen 2007; Huang et al. 2009).

Remote sensing has become the primary data source for large area biomass estimation (Lu 2006), providing spatial detail to capture variability present on the ground (Wulder et al. 2008a), and temporal repetition to account for change (Powell et al. 2010). As summarized by Kangas and Maltamo (2006), national forest inventories (NFI) supply precise information based on plot measurements (e.g., Finland, USA), frequently supported by aerial photography or satellite data (e.g., UK, Canada) that can be scaled and extended to unmeasured areas through direct modelling with passive or active remotely sensed data (Baccini et al. 2004; Blackard et al. 2008). Estimation and monitoring of AGB with remotely sensed data sources can be fast and relatively low cost, providing information for remote and inaccessible areas (Bortolot and Wynne 2005). While error estimates are ultimately linked to the quality of the reference data (Baccini et al. 2007), improved processing algorithms and techniques for data analysis can enhance the accuracy of AGB estimates from remotely sensed data sources (Lu 2006). At high levels of biomass radiometric saturation hampers estimations from optical sensors (Gemmell 1995; Lu 2005; Turner et al. 1999) and radar instruments (Englhart et al. 2011; Mitchard et al. 2009). Airborne and spaceborne LiDAR can provide an important alternative source of forest structural information (Duncanson et al. 2010; Kwak et al. 2010; Næsset and Gobakken 2008), and combinations of data from multiple sensors provide robust options for estimation of forest biomass (Sun et al. 2011; Yu et al. 2010). Retrospective estimation of AGB to establish a historical baseline and enable change reporting is feasible with archival data, where the Landsat program provides the longest and most consistent repository of imagery, going back to 1972 (Wulder et al. 2012). Due to requirements often related to setting a historic (e.g., 1990) baseline, as well as the capture of land use and land cover dynamics at management scales over large areas, Landsat data and related archive is uniquely suited for addressing the information needs of international treaties (e.g., Kyoto Protocol) and supporting national programs. Other data sources exist to augment historic conditions (e.g., SPOT, IRS, DMC) as well as current and newly launched systems (e.g., Sentinel-2). The incorporation and integration of these datasets to generate large area, temporally relevant, data products is non-trivial and subject to ongoing research (Wulder et al. 2015).

7.2 Time Domain in Estimation of Forest Aboveground Biomass

Increasing data availability resulting from archival legacy, free and open data policies, and a growing number of orbiting Earth Observation (EO) satellites, have resulted in the widespread use of image time series for environmental applications (Kennedy et al. 2014). Novel data processing techniques and analytical methods to obtain valuable information of ecosystem status and processes of change are in continuous development, and Landsat time series are currently a field of rapid growth, in particular for forestry applications (Banskota et al. 2014).

7.2.1 Continuous Monitoring of Biomass Dynamics

A host of techniques have been applied for estimation of forest carbon and biomass with single date images, capitalizing on strong direct or indirect relationships between biomass and spectral properties. Image time series, a sequence of consecutive images captured over the same area, provide novel analytical opportunities supported by a leveraged temporal dimension. Time series facilitate the study of long-term trends in spectral response (Vogelmann et al. 2009)—and related forest attributes—while controlling for the variability associated with solar angle, atmospheric effects (Wulder et al. 2008b) and phenology (Sonnenschein et al. 2011), and provide the opportunity to determine rates of change (Gillanders et al. 2008a). Methods to model and map forest attributes like AGB at consecutive dates may apply date invariant relationships (Healey et al. 2006; Powell et al. 2010) to all data in a time series, employing static measures (i.e., spectral predictors obtained at a given time). The relationship between contemporary spectral and reference data is extended to other dates of interest, relying on a robust process of relative radiometric normalization of imagery. With more explicit use of the temporal dimension, time series (or temporal trajectories) of long enough duration can be partitioned into linear segments for interpretation of individual components (Kennedy et al. 2010), deriving descriptive measures (e.g., magnitude, duration) that are useful for estimation of biomass (Frazier et al. 2014) and biomass change (Main-Korn et al. 2013). Alternatively, temporal trajectories can be interpreted as a dynamic variable, a single predictor made from data captured at multiple consecutive dates, where distinctive patterns and trends provide useful information for estimation of biomass and change.

7.2.2 Retrospective Estimation of Forest Biomass and Carbon Stocks

Monitoring historical change over large areas is only feasible with long-term satellite data records and rigorous calibration programs. Medium spatial resolution satellite sensors such as those of the Landsat series (Multi Spectral Scanner (MSS), Thematic Mapper (TM), Enhanced Thematic Mapper Plus (ETM+), and Operational Land Imager (OLI)) are well suited to capture forest change at the stand level in support of research and reporting, relating both natural and anthropogenic drivers of change. Directly linked to field derived measurements, the information provided by Landsat spectral trajectories has proven useful to improve estimation of current biomass and other structural attributes, particularly in ecosystems with stand replacing disturbances (Pflugmacher et al. 2012). For assessment of change, a baseline reference is often required (e.g., Kyoto Protocol). Estimating historical attributes is particularly challenging, because limited reference data precludes validation and accuracy assessment with reliable standard methods (Olofsson et al. 2014). For retrospective estimation of change, evaluation of relative change is sometimes the only viable option, and determining absolute quantities remains unfeasible. It has been demonstrated that temporal trajectory patterns calibrated at more recent times can be temporally transferred for retrospective estimation of AGB. For example the identification of temporal patterns in the trajectory of vegetation indices (i.e., dynamic variables) has been found to provide useful information to model and explain historical biomass variability (Gómez et al. 2014). Extensive research is currently directed at assessing historical change in regional and global forests, with increasingly sophisticated image processing algorithms (e.g., Huang et al. 2009; Kennedy et al. 2010; Hansen et al. 2014) that take advantage of the temporal information leveraged by a dense series of calibrated images. Mediterranean environments in particular have not been intensively researched with image time series. In the remainder of this chapter, we present a case study that demonstrates the opportunities afforded by synergies between field measurements and a time series of remotely sensed data to characterize forest biomass and carbon emissions and removals in a Mediterranean forest area of Spain over the period from 1984 to 2009. This case study is supported by Landsat time series data and repeat field measures from two iterations of the Spanish National Forest Inventory.

7.3 Landsat Temporal Trajectories and NFI Data for Characterization of Change in the Area, Distribution, and Biomass and Carbon Pooling of Mediterranean Pines of Spain

7.3.1 Introduction

Forest areas in countries of the Mediterranean basin account 85 million ha (FAO 2013), and have a notable richness in species diversity (Myers et al. 2000). Pine forests in Mediterranean ecosystems generate wood and non-wood products such as resin, mushrooms, and pine nuts (Calama and Montero 2007) and serve important ecological functions including water regulation, erosion control and provision of recreational opportunities and wildlife habitat (Merlo and Croitoru 2005). Pine forests have a large carbon sink capacity that could help signatory countries of the Kyoto Protocol achieve their targets for the reduction of greenhouse gas emissions (Myneni et al. 2001).

In Spain, similar to other Mediterranean countries, a National Forest Inventory (NFI) provides periodic detailed data for the assessment of biomass and carbon pools through sampling and reporting supported by statistics (MMA 2008). The NFI's 10 year re-measurement cycle enables comparison of data over time, but similar to other sample-based NFIs, has some known limitations, including the discrete character of the sampling, which obliges extrapolation of data (Salvador and Pons 1998), and the use of different basic cartography in subsequent updates of the NFI database (Villaescusa et al. 2001). For areas undergoing rapid change that require up-to-date information on change events, a decade might be too long as a reporting interval (FAO 2010).

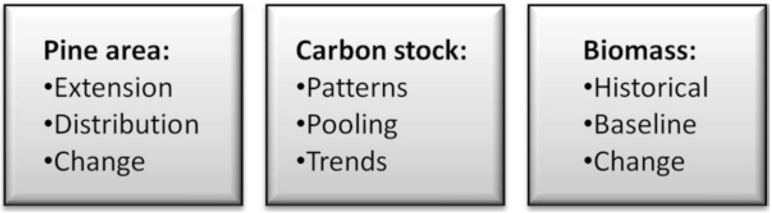

Fig. 7.1 Main objectives of the project: characterization of pine area and change, identification of trends in carbon stocks and pooling, and estimation of biomass and change

7.3.1.1 Goals and Overview

The aim of this work was to characterize the changes in area, distribution, and carbon stocking processes of pines in the Central Range of Spain during a period of 25 years (1984–2009) and to estimate and map historical AGB (at dates 1990 and 2000) in the area permanently covered with pines, as well as a decade of change in AGB (Fig. 7.1). All work was based on the analysis of a medium spatial resolution time series of images from the Landsat series of sensors (TM, ETM+), with support of repeat field measures from a network of NFI plots.

For identification and classification of pine dominated areas at key times during the 25-year period, a multilevel object-oriented methodology was applied. Object-oriented image processing methods base all analysis on spatial units made up by multiple pixels rather than focusing on individual pixels. In the Mediterranean forest under study, natural change is relatively slow and human induced change has historically been controlled (Bravo et al. 2010). A constrained multi-level (with higher levels' objects imposing boundaries on lower level objects) classification of spatial units avoids the "salt and pepper" effect typical of pixel-based classification methods, while enabling identification of small areas, otherwise blurred into averaged larger objects.

Carbon stocking areas and carbon pooling trends (sourcing or sinking) were characterized with the Tasseled Cap Angle (TCA), an index derived from the Tasseled Cap Transformation (Crist and Cicone 1984) that condenses information from the visible, near-infrared and mid-infrared, and bridges between all sensors of the Landsat series (Gómez et al. 2011).

To estimate AGB over the entire pine dominated area the relationship between live AGB derived from NFI ground plot measurements and various vegetation spectral indices derived from Landsat data was explored. Understanding these relationships, a model of past AGB was created with historical spectral data, including single-date data and temporal trajectories, providing a baseline for comparison with more recent estimations. Finally, maps of historical AGB at two dates coincident with NFI rotations (1990 and 2000) were derived, facilitating interpretation of patterns. The distribution of AGB change was evaluated in view of the uncertainties associated with the process of modelling and mapping.

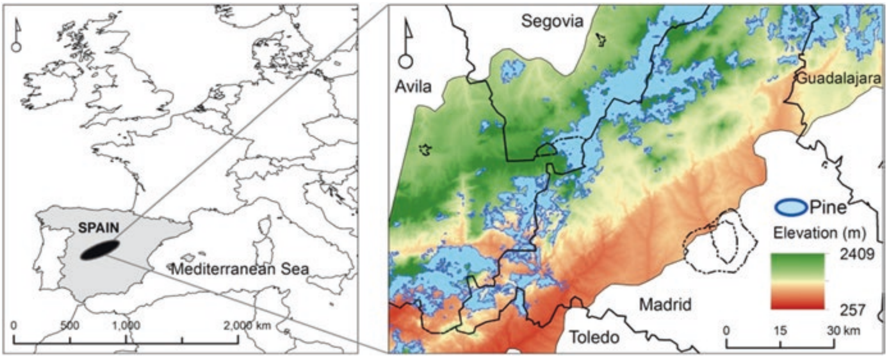

Fig. 7.2 Study area in the central range of Spain

7.3.1.2 Study Area

The area of interest covers approximately one million ha in the Central Range of Spain, occupying part of the Ávila, Segovia, Madrid, Guadalajara, and Toledo provinces (Fig. 7.2). It is centred at latitude 40° 37′ 56′′ N and longitude -4° 6′ 47′′ E. Pines (*Pinus sylvestris* L., *Pinus pinaster* Ait., *Pinus nigra* Arn.) are the dominant tree species, except in the most western area where broadleaf species (*Quercus pyrenaica* Willd.) dominate. Forests extend to elevations of 2000 m, beyond which shrubs (*Cytisus* sp., *Genista* sp., *Erica* sp., *Echinospartum* sp.) are the prevalent vegetation (Rivas-Martínez 1963). Some of these Mediterranean pines have traditionally been managed for production of resin and timber, recreation, and protection, with the last two objectives having increasing importance. A range of structural conditions (e.g., mono-specific and even aged, multi-species, multi-layer) have resulted from the various management practices that have been applied, including the non-management option (e.g., multi-aged, multi-story). Stand age classes are typically defined as 20-year intervals for the species in the entire area (Serrada 2008). Silvicultural practices include pruning and thinning, with timber extraction implemented over time by progressive cuts of low intensity.

7.3.1.3 Data

Medium spatial resolution remotely sensed data (i.e., 10–100 m pixel) is well suited for characterizing forest condition and change (Wulder et al. 2008c) and is the only feasible, cost-effective option for extensive areas (Lunetta et al. 2004). Landsat images sense information from the visible, near-infrared and mid-infrared wavelengths. Since 1972 the United States Geological Survey (USGS) has been archiving Landsat imagery captured over almost any part of the Earth. In 2008 the USGS opened the archive to unfettered public access to analysis ready imagery (Woodcock et al. 2008), removing access and cost limitations and creating myriad opportunities for characterizing both spatial and temporal landscape processes (Goodwin et al.

2008; Potapov et al. 2011). Landsat data is also available for research in the European Space Agency (ESA) repositories and in the Spanish Instituto Geográfico Nacional (IGN) as part of the Plan Nacional de Teledetección (PNT) (Villa et al. 2009). With easy access to the Landsat archives it is now possible to acquire a time series of imagery for almost any area of the Earth (Wulder et al. 2011). Still, due to long-term acquisition plans, international co-operators' capacity, and cloud regime, the acquisition of a historical time series of multiple adjacent Landsat images (relatively cloud-free) is not easy for some regions (Hansen and Loveland 2012), especially where clouds persist (Roy et al. 2010). Spain is well represented by Landsat imagery, although as Senf et al. (2015) describe, accessing imagery from the ESA archive remains with limitations and controls not in place with the USGS.

This research focused on a single Landsat scene (WRS-2 Path 201, Row 032) (Fig. 7.2), selected as it encompasses the most extensive continuous pine stands of the Spanish Central Range. Anniversary images were selected when possible (Table 7.1); in order to capture similar phenological conditions and to avoid the presence of snow in high altitudes, summer images were selected. To detect and avoid possible phenology artefacts, the spectral suitability of early summer images (years 2000, 2001, and 2005) was thoroughly checked through the processing stages.

The time series consisted of nine Landsat TM and two ETM+ (Scan Line Corrector (SLC) on) images. To ensure a more complete time series, we increased our tolerance for a small amount of cloud cover in the images, but still, a yearly time series of images was not possible to obtain and the time step is not constant; there is a gap in images in the 1990s corresponding to the private sector distribution era (Tolomeo et al. 2009). Longer intervals between images may reduce detection accuracy for subtle changes (Wilson and Sader 2002; Jin and Sader 2005).

Table 7.1 List of Landsat images used in the study

Landsat / Sensor	Source	dd/mm/yyyy	Sun elev.	Carbon pooling	AGB modelling
5 / TM	EarthExplorer	18/08/1984	52.89	X	X
5 / TM	EarthExplorer	11/08/1987	54.11	X	X
4 / TM	EarthExplorer	11/08/1990	54.38	X	X
4 / TM	EarthExplorer	14/08/1991	51.68	X	X
7 / ETM+	*EarthExplorer*	*22/08/2000*	*54.87*	*X*	*X*
7 / ETM+	EarthExplorer	06/06/2001	64.24	X	
5 / TM	EarthExplorer	17/06/2002	62.20	X	
5 / TM	EarthExplorer	07/08/2003	56.50	X	X
5 / TM	Aurensis	25/08/2004	53.15	X	X
5 / TM	Junta de Castilla y León	24/05/2005	62.80	X	
5 / TM	EarthExplorer	23/08/2009	54.48	X	X

Reference image for radiometric normalization (22/08/2000) is *highlighted*. Images used in each stage of work are indicated

Table 7.2 Statistics of the attributes related to biomass and structural complexity evaluated at NFI plots

Attribute type	Plot Attribute	Description	Mean	Std. dev.	Min.	Max.
Biomass	AGB_{1990}	Above ground biomass NFI2 (1990)	93.29	67.09	1.45	352.08
	AGB_{2000}	Above ground biomass NFI3 (2000)	109.36	68.99	0	398.90
	ΔAGB	Increment of AGB between NFI2 and NFI3	14.8	50.07	−236.86	242.38
	Rel_{1990}	Increment of AGB relative to AGB_{1990}	0.79	2.55	−1	28.41
	Rel_{2000}	Increment of AGB relative to AGB_{2000}	−0.03	1.33	−17.00	0.96
Structure complexity	D_{MAD}	Median absolute deviation of D (1990)	4.59	3.93	0	8.00
	H_{MAD}	Median absolute deviation of H (1990)	1.81	1.18	0	24.35

Field data from the Spanish National Forest Inventory (NFI) and other plot-based local management inventories were used at various stages of the research. The NFI measures a network of permanent plots (1 × 1 km grid) with four concentric subplots of radius 5, 10, 15, and 25 m. The assessment and reporting unit of the NFI is the province (with ~10^6 ha on average). Data from plots dominated by pine species (with pine basal area ≥ 75 % total basal area) in five provincial databases (Ávila, Segovia, Madrid, Guadalajara, and Toledo), acquired by NFI2 (ca. 1990) and NFI3 (ca. 2000) were used in this work. Per tree measures of diameter at breast height (D), total height (H), and per plot number of trees (N), were used for calculation of biomass in an area where NFI plots were de facto measured during the 1992–1994 and 2000–2004 campaigns of the NFI2 and NFI3 respectively. The capacity to obtain synchronous measures over extensive areas is limited in field campaigns, and can affect the calibration of remote sensing based models. Live AGB was calculated with the species specific allometric equations of Montero et al. (2005) and Ruiz-Peinado et al. (2011) for all trees with D ≥ 7.5 cm. These equations account the dry biomass fraction of stem, roots, and branches of various dimensions, but we did not consider the root portion in our analysis. Absolute and relative change of AGB between the rounds of NFI was calculated (Table 7.2). NFI2 intra- plot structural complexity was evaluated as in Gómez et al. (2011) calculating the median absolute deviation (MAD) of measured D (D_{MAD}) and H (H_{MAD}) in each plot: increasing values of the MAD indicate higher structural complexity, and a zero MAD value is possible but unlikely to occur if all trees in a plot have exactly the same dimension. Plots subject to complete resource extraction between the two NFI rounds were disqualified in support of our assumption of near to natural successional conditions, leaving 573 plots for further analysis.

There were more eligible trees (D > 7.5 cm) for measurement in the NFI3 (ca. 2000) than in the previous rotation of the inventory in this area (NFI2, ca. 1990). Increments are particularly marked for the larger diameter classes, resulting in an expected increase in AGB during the period 1990–2000. However, the distribution of AGB (t ha^{-1}), as derived from field measures in NFI plots is similar at both dates, unimodal and positively skewed, with a majority of plots around 50 t ha^{-1} and median values of 59 and 77 t ha^{-1} in NFI2 and NFI3 respectively. The indicators of structural complexity point to potentially more relevance of D_{MAD} than H_{MAD}.

Supporting data, in the form of cartographic maps, were required at some stages of the study, particularly for verification purposes. We used the Mapa Forestal Español (MFE50), the digital version of Ruiz de la Torre forest map of Spain for the year 2000. MFE50 GIS database encompasses 68 attributes to characterize vegetation units. Some relevant attributes for identification of pine forest areas are dominant species and crown cover (that is, the proportion of area covered by the horizontal projection of the canopy (in percentage)).

7.3.1.4 Image Pre-processing

In order to prepare the Landsat images for analysis they were subject to strict pre-processing operations. Working with image time series a robust radiometric correction is essential, for change detection applications (Lu et al. 2004; Coppin et al. 2004) and when image values are related with biophysical phenomena (Gong and Xu 2003). Pre-processing included atmospheric correction of a reference image (year 2000) with the COST method (Chávez 1988), relative radiometric normalization of the whole series with IR-MAD (Canty et al. 2004), ortho-correction with a 30 m digital elevation model, and geometric co-registration (RMSE < 0.5 pixel), following a processing flow recommended for detection of vegetation dynamics (Vicente-Serrano et al. 2008). Image normalization transforms images to a common radiometric scale, minimizing sun, sensor and view angles, as well as atmospheric differences among images. The process of normalization reduces the amount of artefacts due to illumination or atmospheric variations, enabling more reliable detection of true change (Song et al. 2001).

7.3.1.5 Vegetation Indices

Spectral vegetation indices (VI) are combinations of spectral bands that indicate relative abundance and activity of green vegetation (Jensen 2005). VI are dimensionless attributes that attenuate the effects of different viewing angles and sun orientations and help reduce data dimensionality. The Normalized Difference

Vegetation Index (NDVI) (Rouse 1973) and the Tasseled Cap Transformation (TCT) (Kauth and Thomas 1976; Crist and Cicone 1984; Crist 1985; Huang et al. 2002) were derived from normalized Landsat reflectances. NDVI is essentially a contrast between the reflective red and infrared bands, where vegetation is distinctive from any other material. The TCT transforms Landsat original spectral bands into a number of components, some of which have direct physical interpretation. The Tasseled Cap Brightness (TCB) component is by definition (Crist and Cicone 1984) a positive value, whereas the Greenness component (TCG) depends on the contrast between visible and near-infrared bands, with exposed soil having negative values (Gillanders et al. 2008b; Price and Jakubauskas 1998) and vegetated areas positive values. TCG and TCB components define the so called by Crist and Cicone (1984) vegetation plane. Studying the spectral behaviour of forest stands in the vegetation plane provides insights into forest cover densities and forest development stages. A range of studies in coniferous forests have confirmed higher values of TCG and lower values of TCB in forest dense cover classes when compared to open stands or clearcuts (Cohen et al. 1995; Healey et al. 2005). The Tasseled Cap Angle (TCA) index, defined as the angle formed by TCG and TCB in the vegetation plane (Eq. 7.1) and first used by Powell et al. (2010) for modelling biomass in coniferous and mixed forests of Arizona and Minnesota (USA), condenses in a single value the information of the relation TCG/TCB: dense forest stands exhibit higher values of TCA than open stands or bare soil (Fig. 7.3). The TCA was calculated as the angle between normalized TCG and TCB components for the time series of images. The eleven TCA layers combined in a single image are noted hereafter as TCA image. The temporal sequence of spectral values, (temporal trajectory), were obtained from consecutive images and used in the rest of the work.

$$TCA = \arctan\left(TCG / TCB\right) \tag{7.1}$$

7.3.1.6 Relevance of Tasseled Cap Angle and Process Indicator in the Area

The relationship of the TCA with forest density variables at the stand level in the Mediterranean pines of the study area was explored. Data from plot-based management field inventories established in ~100 m grid networks were krigged to 30 m spatial resolution (with an ordinary krigging based on the spheric variogram model) and regressed with values of the TCA at the stand level. The entire range of basal area (BA) representative of the study area was included in the correlation analysis, and as expected, the TCA and BA were found linearly related, with a high and positive value of correlation ($R^2 = 0.80$). Based on the strong relation between TCA and density variables in the study area it was posited that analyzing the TCA values over a time series of images provides information of relative changes in the density of forest stands: the TCA is stable if there is no change in density (constant BA); an increment in BA (e.g., natural regeneration or plantation, stand maturity or increase

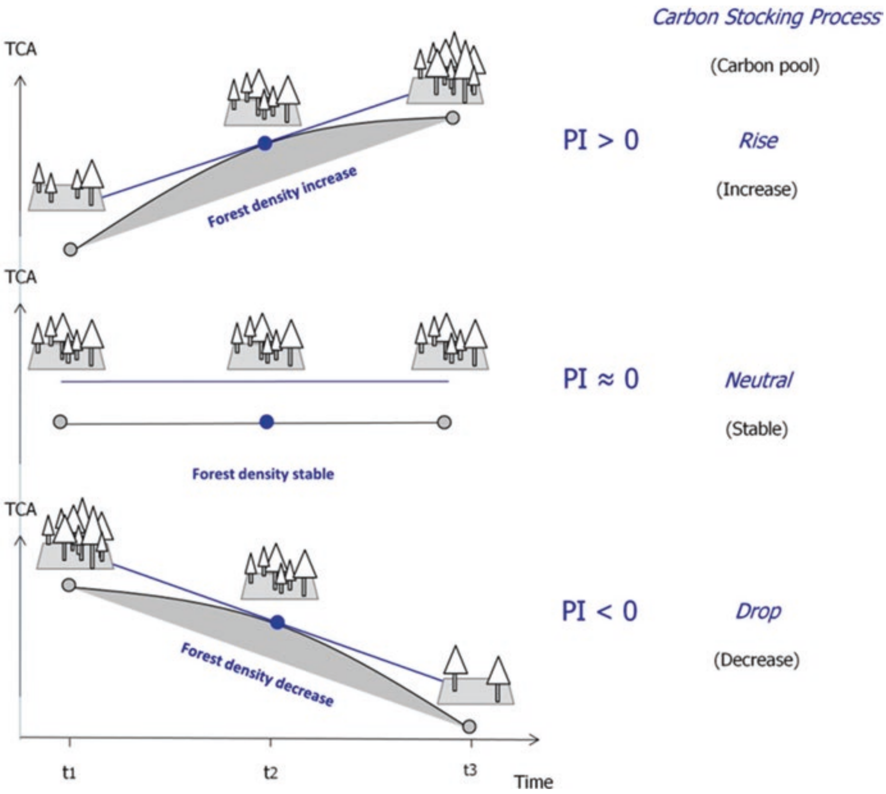

Fig. 7.3 TCA and PI interpreted in relation with carbon pooling processes (After Gómez et al. 2012)

of crown closure) results in a concomitant increase in the TCA and conversely, when the BA diminish (e.g., after a partial harvest or thinning operation, or after a disturbance such as a fire), the TCA value decreases (Fig. 7.3).

Each pixel TCA profile was approximated with a Lagrange second order polynomial (which enables interpolation with uneven intervals among occurrences), and its derivative with respect to time (years) was calculated. The result is a multi-layer spectral image with the same number of bands as the original TCA image, which we define the Process Indicator (PI) image (Gómez et al. 2011). The PI image illustrates at each pixel the rate of TCA change over time. As the TCA informs on relative forest density at each date, the PI similarly informs of the instantaneous rate of change in forest density at each time. For example, a high positive value of PI indicates a relatively fast rise of TCA (e.g., a stand rapidly augments density by rapid growth or quickly developing towards crown closure); high negative value of PI indicates a relatively fast drop of the TCA value (and stand density) (e.g., after a stand replacing disturbance or a strong thinning). Moderate values of PI refer to slow and slight changes in the TCA value, such as lowered density after partial harvest (negative PI) or increasing density with slow natural growth or development (positive PI). Relative changes in carbon pools associated with changes of forest density can similarly be assessed (Fig. 7.3). PI values are direct indicators of processes of change and constitute a practical tool to monitor temporal relative changes; for estimation of absolute values of change, a thorough calibration of the index would be required.

7.3.2 Assessment of Pine Dominated Areas Over Time

To evaluate changes in the extent and distribution of the pine dominated area over time, pine areas were identified at four dates that divide the 25-year period in three epochs of similar duration (i.e., 1984–1990, 1990–2000 and 2000–2009). Radiometrically normalized bands 3, 4, 5, and TCA (indicative of forest density) of four images were independently segmented with a three level hierarchy, into spectrally homogeneous multi-pixel objects of 2, 6.5, and 33.5 ha on average. Three levels of segments per image were classified with a supervised nearest neighbour rule. The multilevel classification aimed to simultaneously identify larger stands with the required species and density characteristics, and smaller objects in patchy areas. This technique is of particular interest to distinguish small changes in distribution that would otherwise blur into larger objects or be rejected as a speckle effect in a pixel-based classification. All four dates' classifications were trained with the spectral signatures of samples acquired for the reference year 2000. A seven class scheme was considered, to reduce the error in change detection (Fuller et al. 2003), but only the pine class was to be retained. Objects classified as pine in any of the three hierarchical levels were merged and the resulting areas at each date (i.e., 1984, 1990, 2000 and 2009) were compared in a GIS for assessment of change. An exhaustive account of changes in the area occupied by pines at each sub-period was possible applying these area change concepts:

Type of change	Description
Stable	area classified as pine in the initial and final date of the period
Increment	area classified as pine at final date, which was a different land cover class at initial date of the period
Reduction	area classified as pine at initial date and not pine at final date of the period
Net change	Increment - Reduction (>0 or <0)
Changed	Increment + Reduction. Area subject to change
Potential area	Stable + Changed. Area occupied by pine at initial and/or final date of the period

The aim was to identify patterns in the distribution and change in extent of the pine dominated area during the 25 year period (1984–2009). From 1984 to 2009 there was a 40 % increment in the area dominated by pine species, mainly attributed to agricultural land abandonment. There was abundant transformation during the first sub-period (1984–1990): an extent equivalent to 57 % of the original pine area changed, producing a net increment similar to 36 % of the original extent. In the second sub-period (1990–2000) the amount of area changed was less notable, equivalent to 33 % of the initial date (1990) pine extent and with a net loss of 17 % of the pine dominated area. This decade maintained the most extensive stable area of the three sub-periods. In the course of the last sub-period (2000–2009) the Increment was 3.6 times the Reduction of the pine area, resulting a Net Change equivalent to 25 % the area occupied by pines in 2000. Spatiotemporal patterns of change indicated that as a general rule, changes in the distribution of pines have occurred at the boundaries of permanently covered areas. However, the central region (B in Fig. 7.4) and eastern region (C in Fig. 7.4) maintained relatively high and permanent coverage, whereas in the southern region (A in Fig.7. 4) coverage was discontinuous in space and time. Increments in pine area, motivated by natural colonization or by plantation of agriculture abandoned lands were common in the three regions during period 1984–1990, mostly located in region C during the intermediate period (1990–2000), and particularly frequent in region A during period 2000–2009. On the other side, reductions of the pine area were more frequent in region C during the initial period, distributed across regions A and B during the intermediate stage and similarly distributed across regions A and C in the last period (2000–2009). Clear cutting is a forestry technique in disuse in the Central Range and all wood extractions are now of low intensity; however, a few stand replacing disturbances due to fire have been identified. Fig. 7.4 and Table 7.3 illustrate these results.

The accuracy of the thematic map (>90 %) was assessed for year 2000, for which field data from the NFI3 were available, with an ad hoc design analysis for estimation of the omission and commission errors (Gómez et al. 2012). Confidence on the validity of other classifications is based on the robustness of the radiometric calibration and normalization and the transference of spectral signatures.

7.3.3 Assessment of Carbon Stocks

The Maximum Potential Pine Area (MPPA) for the period 1984–2009 was defined as the overall union of pine areas at any of the four dates considered. The MPPA represents the maximum extent occupied by pines at any time during this period, and it encompasses a region persistently occupied by pines (Permanent) and other areas that have only been intermittently covered with pines during period 1984–2009 (Intermittent). Change in carbon stocks and spatiotemporal patterns of carbon pooling trends were evaluated over the MPPA.

Relative rates of carbon stock change were examined through the rates of change in TCA, a surrogate of forest stand density, with the Process Indicator (PI) value at each date. The overall neutral quality of these pines as a carbon pool was indicated by a low average PI during the 25 year period: the rate of change of carbon stocks is slow on average. Values of PI over time indicate carbon stock change: prior to

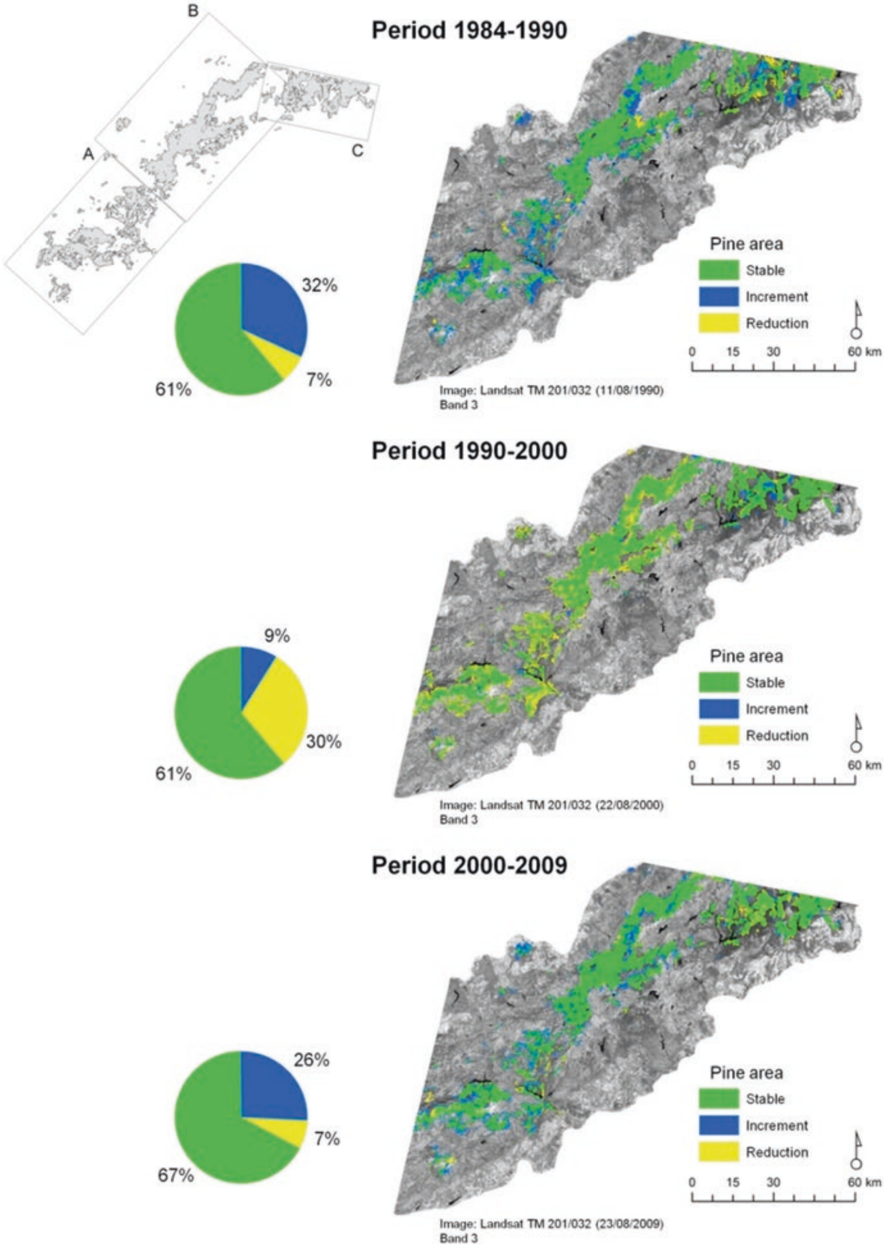

Fig. 7.4 Maps of stable and changed pine dominated area in three sub-periods (1984–1990, 1990–2000, and 2000–2009). *Green* is the area that remains as pine during the period, *yellow* shows the reduced pine area and *blue* shows the increased pine area. The study area (top-left inset) is divided in three sections (A, B, and C) to facilitate description of change over the three periods of interest

Table 7.3 Pine area and changes during three sub-periods

Period	Initial area (ha)	Increment *(%initial)	Reduction *(%initial)	Net change *(%initial)	Changed *(%initial)	Potential *(%initial)	Stable *(%initial)	Final Area (ha)
1984–1990	121,144	56,496	12,858	43,638	69,354	177,365	108,011	164,622
		(46.6)	*(10.6)*	*(36.0)*	*(57.2)*	*(146.4)*	*(89.2)*	
1990–2000	164,622	12,502	41,306	-28,804	53,808	177,023	123,215	135,980
		(7.6)	*(25.1)*	*(−17.5)*	*(32.7)*	*(107.5)*	*(74.8)*	
2000–2009	135,980	47,149	13,001	34,148	60,150	182,757	122,607	169,825
		(34.7)	*(9.6)*	*(25.1)*	*(44.2)*	*(134.4)*	*(90.2)*	

Overall total area identified as "stable": 91,349 ha
Overall total area identified as "potential": 197,144 ha
*Equivalent to the area at initial date of the sub-period

1990 the overall rate of change was relatively low and steady, but from 1990 to 2000 there is a tended increase in carbon stocking rate; the highest overall rate of change occurred in 2000 and in the last decade there is a tendency towards lower rates of change. Interestingly, an increased complexity of the landscape carbon pools during the last period is indicated by higher values of the PI standard deviation. It is worth emphasizing that the PI is an indicator of processes and does not enable estimation of absolute carbon stocks, but indicates relative rates of change in carbon stocks. Positive PI values indicate the forest is in a process of augmenting its carbon storage (e.g., density increment by natural growth); negative PI values indicate the forest is in a process of reducing its carbon storage (e.g., diminution of density in a thinning operation). There is no difference in trends of change over time between Permanent and Intermittent pine coverage. However, fluctuations of the PI average are notably more accentuated in the Intermittent area, while the PI standard deviations and range of values are lower in Permanent areas, confirming the more stable character of the persistent pines.

To facilitate analysis and visualization of patterns, spatial units with homogeneous forest density (34.5 ha on average) were automatically generated by segmenting the TCA 2000 image and these units' PI values were evaluated. At initial dates most objects indicate a similar low rate of carbon stock change (low PI) but the global stability of the landscape carbon stocks decreases progressively (Fig. 7.5): areas with relatively steady carbon stocks (i.e., not modifying forest density) at initial dates develop towards higher carbon stocks (e.g., density increment) or lower carbon stocks (e.g., density drop). Five categories of PI values (fast drop, low drop, neutral, slow rise, fast rise) were established to produce more detailed information of the spatial distribution of carbon pooling changes over time. The carbon stock of areas in the neutral group is not in a process of change; the slow drop and slow rise groups are in a slow process of changing their carbon stock towards lower or higher levels respectively and the fast rise and fast drop groups are in a relatively rapid process of changing their carbon stock towards higher or lower levels respectively. These results are always relative to the area of analysis and comparison with results in other areas would require thorough calibration of values. On average 64 % of the area was in neutral carbon pooling process over the entire period (Fig. 7.5a).

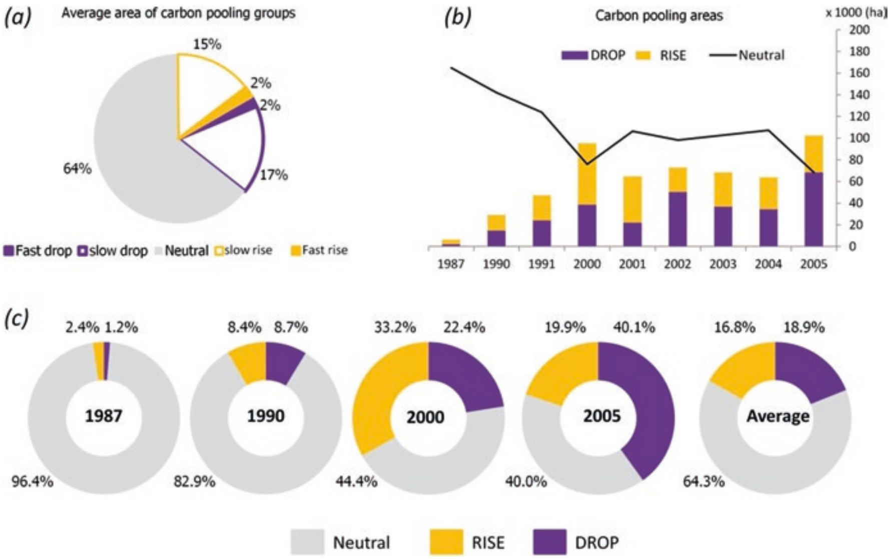

Fig. 7.5 Proportion of carbon stocking groups. (*a*) Average percentage over the whole period; (*b*) Evolution along the period; (*c*) Proportions on various dates and the overall average proportion

The neutral area followed a consistent lowering trend over time from a maximum area in 1987 (96 % of total area) to a minimum in 2005 (40 % of total area) (Fig. 7.5b). On average, only 2 % of the MPPA area was in fast rise and a similar 2 % of the area was in fast drop carbon stocking processes during the period. Slow rise and slow drop carbon stocking processes represent equivalent areas along this period, with an overall average of 17 % and 15 % respectively. Before 2000 more than 70 % of the MPPA area was Neutral and after that it fluctuated between 40 and 64 %. The area in process of rising carbon stock reached a minimum proportion in 1987 (2 %) and a maximum proportion in 2000 (33 %). The area in process of dropping carbon stock reached a maximum proportion in 2005 (40 %) and a minimum in 1987 (2 %). Permanent and Intermittent areas both follow a similar trend of decreasing the proportion with neutral carbon character, although the trend is more pronounced in the Intermittent area. The Permanent pines have a more balanced distribution of areas in process of rising and dropping carbon stocks. Observing the enlargement of area ongoing processes of rising or dropping carbon stock, it is clear that a carbon pooling activation occurred in the period 1984–2009.

The PI continuous scale of values provides versatility in change detection capacity and enables the characterization of rapid (high PI values) and slow (low PI values) rates of change. Subtle changes in forest density can be detected, which is of particular interest in the Mediterranean area, where the majority of forests are subject to some drought and consequently are relatively slow growing when compared with other temperate areas (Merlo and Croitoru 2005). In managed forests, partial harvest or thinning operations might be detected (low negative PI value) and later recovery of density tracked (positive PI value). If the silvicultural goal is to maintain

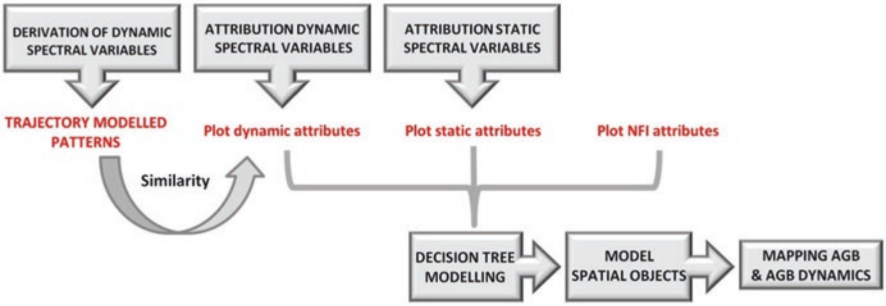

Fig. 7.6 Main stages of the process to estimate and map historical AGB and change

a constant value of basal area, a time series of PI values would remain close to zero. Historic trends of relative carbon stocking can be assessed, and the effect of management practices monitored with detailed spatial information. A PI based approach is especially informative for locations characterized by subtle, non-stand-replacing disturbances.

7.3.4 Estimation and Mapping of Historical AGB and AGB Change

Historical estimations of AGB are necessary as baseline for comparison with more recent estimates and to support monitoring and reporting commitments. Models generated at the plot level were applied at similar multi-pixel object size level, with time series of Landsat images for assessment of AGB at specific dates coincident with NFI rotations (i.e., 1990 and 2000) (Fig. 7.6). To identify vegetation indices suitable for modelling AGB in the area, the relationship between AGB and spectral properties at the plot level was explored through an ordered system of spatially coincident field measures and calibrated spectral trajectories captured over 25 years, transformed by wavelet analysis (Gómez et al. 2014). This exploration revealed strong underlying connections, influenced by forest structural complexity and affected by exogenous factors. Level six of the Meyer based discrete wavelet transform decomposition showed the highest explanatory power: at this level the NDVI and TCA approximation of the wavelet functions follow an almost coincident trend. The relationship of Tasseled Cap Distance (TCD) (where TCD = sqrt (TCB^2 + TCG^2) with AGB is strongly influenced by forest structure, in particular by local diversity of diameter and height. Analysing a sub-sample of plots regularly distributed over the range 1–310 t ha^{-1}, the detail function of level five wavelet decomposition revealed sections of maximal variation at regular intervals, suggesting AGB categories equivalent to ~50 t ha^{-1}.

Dynamic spectral variables (i.e., variables with an inherent temporal component) associated with forest successional development were defined through wavelet

Table 7.4 Dynamic variables derived by transformation of spectral trajectories with a 2D wavelet smoothing of an ordered system

Group of dynamic variables	Spectral index	Variable construction	
		Model ordering attribute	Variable id. (#patterns)
State	TCA	AGB_{1990}	M11 (6)
Original trajectory		AGB_{2000}	M12 (8)
25-year (1984–2009)			
15-year (1990–2004)		ΔAGB	M13 (8)
	NDVI	AGB_{1990}	M14 (7)
		AGB_{2000}	M15 (7)
		ΔAGB	M16 (7)
Process	TCA	ΔAGB	M1 (7)
Derivative trajectory		Rel_{1990}	M2 (7)
25-year (1984–2009)		Rel_{2000}	M3 (7)
15-year (1990–2004)		AGB_{1990}	M4 (8)
		AGB_{2000}	M5 (8)
	NDVI	ΔAGB	M6 (7)
		Rel_{1990}	M7 (8)
		Rel_{2000}	M8 (8)
		AGB_{1990}	M9 (8)
		AGB_{2000}	M10 (7)

transformations of ordered (by increasing AGB) spectral trajectory systems. Six *state* variables (derived from original spectral trajectories) and ten *process* variables (derived from temporal derivatives) were generated with NDVI and TCA temporal trajectories. Since the effectiveness of spectral trajectories as predictor variables is likely to be related to duration and starting position, the performance of a complete 25-year (1984–2009) version, and a 15-year (1990–2004) version spanning the time lapse between NFI2 and NFI3 measurements, were tested (Table 7.4). The domains of these dynamic variables were semi-automatically identified as six to eight modelled curves representing distinctive successional patterns (Gómez et al. 2014). Ideally, if endogenous factors (e.g., structural complexity) and exogenous factors (e.g., topography, sensor limitations) were controlled or suppressed, *state* variables could describe the evolution of forest biophysical parameters related with spectral indices, and *process* variables would represent the rate at which those processes of change occurred. In reality, only approximations can be interpreted, as no single *state* or *process* variable is capable of completely explaining the biophysical development of forests. In order to attribute image multi-pixel objects with a value of each dynamic spectral variable, a time series similarity approach was implemented with the Dynamic Time Warping (DTW) algorithm (Giorgino 2009), whereby the most similar reference pattern is determined by the minimum global distance and assigned to each object.

Table 7.5 Summary of modelling results when including different sets of variables

		Fitting	Validation			
	Variable	R	R^2	RMSE (t ha^{-1})	% Mean error	Bias
AGB_{1990}	All	0.95	0.90	32.2	0.34	0.99
	25-year pattern	0.84	0.68	58.7	0.62	1.02
	15-year pattern	0.76	0.54	70.9	0.74	1.01
	NDVI pattern	0.94	0.89	32.2	0.36	0.99
	TCA pattern	0.29	–	–	–	–
	State trajectory	0.22	–	–	–	–
	Static indices	0.18	–	–	–	–
AGB_{2000}	All	0.73	0.53	71.6	0.65	0.96
	25-year pattern	0.65	0.40	79.1	0.72	0.93
	15-year pattern	0.58	0.26	87.9	0.80	0.95
	NDVI pattern	0.65	0.41	78.0	0.71	0.92
	TCA pattern	0.18	–	–	–	–
	State trajectory	0.22	–	–	–	–
	Static indices	0.17	–	–	–	–

NDVI process patterns are best predictors. The entire 25-year pattern variables yield more accurate and precise results than the 15-year pattern variables, but a combination of both yields best results

Decision trees (Breiman et al. 1984) were then generated, learning from plots characterized by dynamic (*state* and *process*) and static (single date) spectral variables as well as field derived AGB data (Fig. 7.6). Samples were split into calibration (50 %) and validation (50 %) sets, assuring both sets covered the entire range of AGB (1 to 350 t ha^{-1}). To fit the model, a cross-validation process with ten iterations was performed, and to avoid over-fitting we considered the establishment of a minimum number of cases in terminal nodes and pruning with the 1 standard error rule (Breiman et al. 1984). Different combinations of predictor variables were tested and models were cross-validated with half of the sample. The best binary models (highest R^2, and lowest RMSE, ME, and bias) for AGB in 1990 and 2000 were selected (Table 7.5) and kept for later application.

The best trees include decision rules based both on *process* and *state* patterns. Moreover, the best fitted tree (R = 0.95) combines 25- and 15-year NDVI *process* variables. When validated, this model shows high R^2 and a small bias towards under-predictions; with a RMSE of 32 t ha^{-1} it produces errors of 34 % on average. Interestingly NDVI *process* patterns were found more relevant than the analogous TCA patterns in describing historical AGB, despite a similar relationship of either static index with biomass. Static indices alone or together with *state* trajectories did not model biomass satisfactorily. All AGB_{1990} decision trees have a first split of plots with AGB > 100 t ha^{-1} (31 % of the sample) into one branch and plots with AGB < 100 t ha^{-1} (69 % of the sample) into the other branch, determined by a rule based on an NDVI *process* pattern: in other words, the rate of stand development is the most relevant factor for identifying plots with large amounts of AGB (which are presumably more mature) from low AGB (and frequently younger) plots.

AGB_{2000} is necessarily modelled with a different temporal configuration of spectral trajectories (a collection of past and prospective changing patterns). Results of modelling AGB_{2000} are more restricted, with statistically significant (p-value < 0.001) best fitting trees having a correlation with the sample of R = 0.73. Models show a general tendency to underestimate AGB, and have limited predictive power ($R^2 = 0.53$). With high ME (70 %) these models are limited for estimation of biomass in the area. Still, the variables related with changing processes have stronger predictive power than those related with state, which reinforces the importance of the rate of change to model development.

To map and assess the biomass and carbon content of pines over the entire area, AGB models derived at the plot level were applied to spatial units of 3.1 and 2.5 ha on average (similar to the 5 × 5 pixels window used for modelling). Spatial units were defined on the Landsat image date 1990 and 2000 respectively, as small multi-pixel objects, within which the spectral trajectory is averaged. Objects were characterized with static and dynamic predictors and classified following the best decision tree model rules. Through this process each spatial object acquired an AGB (t ha^{-1}) value and its related standard deviation as a measure of uncertainty. The consistency of up-scaling models from plot-level to object-level was supported by the high number of objects (95 %) achieving terminal nodes in the decision tree: all combinations of the relevant spectral trajectories characterizing AGB at the plot level correspond with combinations of averaged spectral trajectories of spatial objects with the same AGB (t ha^{-1}).

Pixel-based mapping facilitates comparisons and evaluation of change with direct algebraic calculations, and it offers the option to define aggregation units for particular applications. Pixel-based maps were derived from the object maps initially produced: each pixel was assigned the AGB (t ha^{-1}) and uncertainty (E_{1990}, E_{2000}) values of the object it belongs to, and from these maps the AGB change (ΔAGB) map was derived. The ΔAGB map consists of three layers: a main map evaluated by differences in pixel nominal values and two other layers depicting scenarios of maximum added uncertainty (Eq. 7.2). Detailed maps of carbon stock and carbon flux due to biomass change over the period 1990–2000 can then be derived from the biomass maps using a 0.5 multiplier of AGB to carbon (Kollmann 1959; Penman et al. 2003).

$$\Delta AGB = \left(AGB_{2000} - AGB_{1990}\right) \pm \left(E_{1990} + E_{2000}\right) \tag{7.2}$$

The biomass allocated in the aboveground fraction of trees in year 1990 was on average 77.6 t ha^{-1} (total in the area amounts 6295 × 103 t), and it was estimated to be 91.5 t ha^{-1} (7415 × 103 t) 10 years later (2000). The difference of calculated AGB represents an increment of 1.3 t $ha^{-1}\cdot year^{-1}$ on average and the total increase in the area is equivalent to 560 × 10^3 t of C. Considering individual objects imprecision, global values of AGB_{1990} and AGB_{2000} range between 5.5 and 7.1 × 10^6 t and between 3.4 and 11.3 × 10^6 t respectively, and as expected from the modelling results, the later date contributes notably more to the uncertainty of estimated change. By means of the multilayer raster maps total change was evaluated in the case scenarios

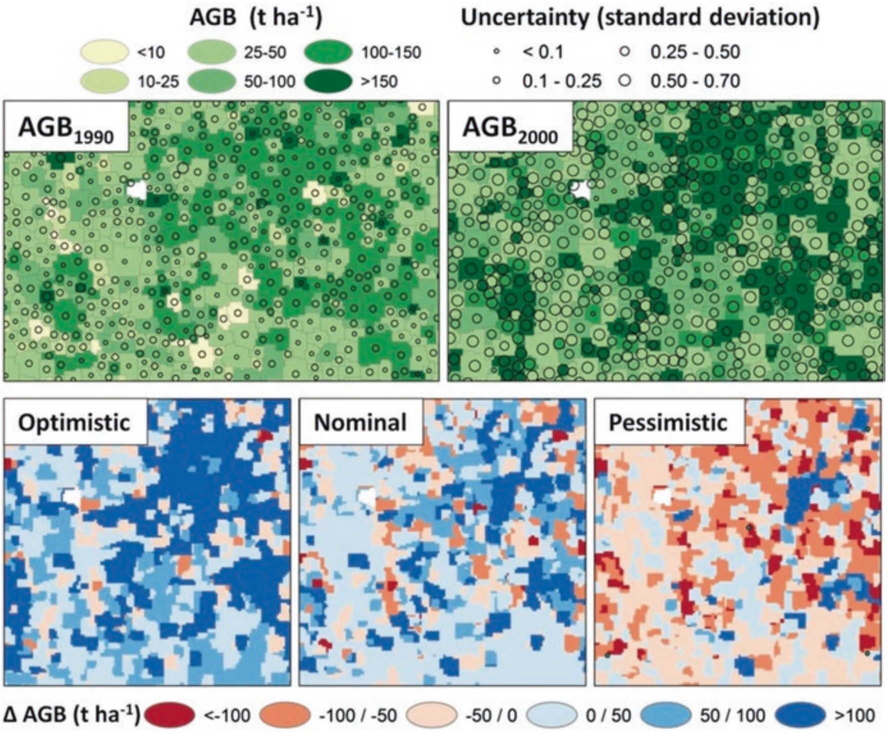

Fig. 7.7 Top: Detail of the object-based biomass maps in 1990 (*left*) and 2000 (*right*). Bottom: Detail of multi-layer ΔAGB maps showing spatial distribution and variability of change derived from uncertainty: optimistic result (*left*), nominal change result (*centre*), and pessimistic result (*right*)

of additive uncertainties derived from modelling and mapping, obtaining values of 2.9 t ha^{-1}·year^{-1} loss and 8.5 t ha^{-1}·year^{-1} gain on average. The spatial distribution of biomass and variability is information of crucial value for management (Fig. 7.7).

In order to validate the raster map of change, it was cross checked with the original values of plot ΔAGB distributed into six categories. To identify sources of confusion we used an error matrix with an expanded diagonal, deemed adequate for this continuum classification established with artificial hard class breaks (Congalton and Green 2009) (Table 7.6). Low producer's (21–27 %) and user's (16–23 %) errors are recorded in intermediate categories (Table 7.6). However, the distribution of values in the matrix point to a slight tendency to overestimate incremental biomass. Overall, 70 % of checked points were classified into the correct category of AGB change.

Previous attempts to model biomass in the Central Range of Spain with single date optical data were limited, characterized by moderate fitting correlation (R = 0.7) and mean error of 0.78 (Gómez 2006). Also in the same area, Vázquez de la Cueva (2008) found structural parameters insufficiently explained by the multispectral predictors selected to derive empirical models; however, the Tasseled Cap

Table 7.6 Accuracy matrix of the raster map of biomass change categories

	Plot change ($t\ ha^{-1}$) (reference)						
Point ΔAGB ($t\ ha^{-1}$)	<-100	-100/-50	-50/0	0/50	50/100	>100	*User's metrics*
<-100	0	0	2	1	0	0	0
-100/-50	3	1	10	33	8	0	0.25
-50/0	3	2	11	28	9	0	0.77
0/50	14	15	50	143	53	17	0.84
50/100	3	7	16	61	8	5	0.74
>100	2	3	8	26	8	1	0.19
Producer's metrics	0.12	0.11	0.73	0.79	0.80	0.26	0.70

Wetness had a stronger relation with forest density than NDVI or any TM/ETM+ band. Interestingly, in the present work the TCA was found significant as static variable, while patterns associated with NDVI were relevant as dynamic process variables. The Tasseled Cap Distance (TCD), more related to age and associated structural complexity than other Tasseled Cap related indices in coniferous forests of Oregon, USA (Duane et al. 2010) was also found linked to forest complexity in these Mediterranean pines. Local difficulties precluded direct modelling of forest attributes with Landsat data, and mathematical transformations based on 2D wavelet algorithms were applied to a data-system created with information from two rounds of field measures and eight repetitions of calibrated spectral data. This technique helped filtering fundamental relations from environmental and endogenous noise. Dynamic variables (i.e., variables with an inherent temporal component) associated with patterns of change, including rate and shape, characterized ground plots, and together with static variables served to model AGB and calculate AGB dynamics. This approach significantly improved previous results, but no single predictor was able to accurately classify biomass.

The temporal configuration (i.e., the duration, starting point, and position relative to the target date) of the dynamic variables presumably affects the capacity to predict structural and successional forest attributes, as suggest different results in modelling AGB in 1990 and 2000. AGB_{1990} corresponds with the initial stages of a trajectory to resemble one of a series of temporal patterns, with possible deviations or delays of key features. Alternatively, AGB_{2000} corresponds with an intermediate position of the available spectral trajectories, with which processes are not aligned. The duration of spectral trajectory necessary to identify significant temporal patterns in AGB is presumably variable and site dependent. Liu et al. (2008) demonstrated that a series of images covering a longer period predicts forest age more accurately, but in some cases a shorter time series of imagery may suffice. In this work a combination of 25-year and 15-year trajectories was the best option for estimating retrospective AGB. Longer-term patterns may potentially explain the variability of AGB more precisely, but they may also introduce irregularities outside the time lapse between data used for calibration of the trajectory models; on the other hand, shorter-term patterns are more explicit and less prone to variations out of the reference period.

The uncertainty remaining in maps of AGB dynamics originates from possible imprecision in modelling, but also from the various stages in the overall approach, including location of plots, field measures, allometric equations, image capture, and image processing (Lu et al. 2012). To minimize the impact of these factors, a representative sample acquired to consistent specifications, such as NFI plots, is recommended for modelling (Duane et al. 2010), and necessary to obtain a comprehensive domain of trajectory patterns for accurate identification by the similarity algorithm. Our estimates of AGB dynamics between 1990 and 2000 are in agreement with complementary regional studies. For instance, pines in the Central Range were found to be more dense and mature in year 2000 than during the previous decade, and – as could be expected – accounted a net increment of biomass and carbon stock. Analysing inputs and outputs recorded by NFI measures, Herrero and Bravo (2012) corroborated a net carbon sinking character between NFI2 and NFI3 rotations, with AGB allocated in pines of 85 t ha^{-1}, while Montero et al. (2004) estimated an annual increment of 0.9 t ha^{-1} of pine biomass between 1993 and 2003.

7.4 Final Remarks and Conclusions

Assessment of forest aboveground biomass (AGB) and its dynamics over time at the landscape level involves evaluation of diverse yet related aspects. The location, extent, distribution, and change in forest area over time need to be identified, as well as ongoing successional processes and spatiotemporal patterns of change. Models calibrated with available field data require up-scaling and extension to large areas, and maps accounting for uncertainties in model estimates are needed for informed interpretation and decision making. Remote sensing technology is well suited to support these activities and has become the primary data source for biomass estimation over large spatial extents. Synoptic, repeat and consistent observations of the landscape provide information at a range of electromagnetic wavelengths associated with forest traits at minimal or no economic cost to assist in the process of regional and global forest biomass assessment and monitoring. Ground observations acquired through local and national forest inventories are crucial as a supporting source of data for model calibration and validation.

Historical series of Landsat images enable retrospective studies of change with increased detailed and precision. Twenty five years of change in the area and distribution of pine dominated areas in the Spanish Central Range were characterized with the Tasselled Cap Angle (TCA), an index that bridges between all Landsat sensors, derived from normalized reflectance data captured at regular intervals. A key strength of methods based on remote sensing technologies is the capacity to readily incorporate data at intermediate dates for generation of more detailed reports. Changing trends in relative carbon stocks were assessed with the Process Indicator (PI), the TCA temporal derivative, and further characterized by sub-periods of time, with subtle change detection enabled and demonstrated. Combined, the TCA and PI described state and processes of change in these forests, capitalizing on the local relation of these spectral indices with forest variables of interest. The TCA is strongly correlated with stand density in the study area, whilst the PI characterizes rates and directionality of change, enabling description of processes. Biomass accrual occurs naturally in the absence of disturbance, unless there is depletion in cases where removals are evident, and carbon equivalents generally follow the same logic. Our analysis and interpretation of the spectral dynamics of pines indicates that the carbon stocking pools of the study area have been activated in the second half of the analysis period (1984–2009), when larger areas show faster rates of increasing and decreasing carbon stocks. The spatial definition of sources and sinks as well as changing trends over time are a valuable contribution for the global issue of carbon budgeting reports and for evaluation of management strategies.

Time series techniques are progressively developing assets in image processing and interpretation and have demonstrated effectiveness for a range of environmental applications. Approaches for image time series analysis are diverse; local characteristics and specific information needs should be considered prior to the choice and application of methods. In our study area, which is characterized by the absence of major perturbations and moderate human intervention during successional stages, a continuous approach for time series analysis was found to be useful for modelling AGB. Frequency and regularity of ground measurements or remotely sensed observations can be critical in providing an accurate understanding of ecological processes. Likewise, gaps in a series of images and irregular data frequencies leave intervals of uncertainty in explaining continuous processes that might be notable in ecosystems prone to rapid changes related to disturbance. Successional patterns are more predictable in undisturbed forests than in areas with unexpected perturbations and the rate of spectral variation is typically greater in immature stands when compared to more mature stands in similar environments. Wavelet transform analysis is particularly suited to detect anomalies in data series (Mallat and Hwang 1992) and does not require periodic sampling (Daubechies et al. 1999), making this approach adaptable for analysis of data in a wide range of environments. Dynamic process features such as pattern and rate of change were more relevant than static variables in the retrospective estimation of AGB in the Mediterranean pines of central Spain. Pines in these forests were found to have accrued biomass over the decadal monitoring period (1990–2000), representing a net carbon sink.

Remote sensing technologies support and enhance the value of national forest inventories for the assessment of biomass and carbon balance, and the Landsat archive in particular is a unique source of spectro-temporal data for modelling and mapping forest attributes. The approaches presented herein allow for characterization of forest areas and trends in carbon stocks, and the retrospective estimation and mapping of AGB, in order to establish a historical baseline and enable change reporting.

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Chapter 8
REDD+ and Carbon Markets: The Ethiopian Process

Julian Gonzalo, Solomon Zewdie, Eyob Tenkir, and Yitebitu Moges

Abstract In 1997, the Parties to the UNFCCC (United Nations Framework Convention on Climate Change) adopted the Kyoto Protocol to operationalise the UNFCCC by committing countries included in its Annex 1 (industrialised countries and countries with economies in transition) to emissions reductions targets and establishing mechanisms to achieve those targets. A new commodity was created in the form of emission reductions or removals. Carbon (carbon dioxide) is now tracked and traded like any other commodity. Carbon markets are means by which contracts of sale of emission reductions of greenhouse gases (GHGs) are exchanged. Complexity in the operation of these carbon markets has been increasing day by day, with different mechanisms, programs and assets.

REDD + mechanism (Reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries) incentivizes a trend change from historic increasing deforestation rates and greenhouse gases emissions. Through REDD+ mechanism developing countries are financially granted for any emissions reductions achieved associated with a decrease in the conversion of forests to alternative land uses.

With more than 18 million hectares of forests (including woodlands, Ethiopian Mapping Agency 2013) covering approximately 16 % of its land area, Ethiopia has a huge potential and is among the countries partnered with FCPF (Forest Carbon Partnership Facility) to implement REDD+.

An approach to the complexity of carbon markets and to the key mitigation program in the forestry sector through the process initiated in Ethiopia.

J. Gonzalo (✉)
Ministry of Environment, Forest and Climate Change, 12767, Addis Ababa, Ethiopia

Sustainable Forest Management Research Institute, Universidad de Valladolid & INIA, Valladolid, Spain
e-mail: jgonzalo.jg@gmail.com; jgonzalo@pvs.uva.es

S. Zewdie • E. Tenkir • Y. Moges
Ministry of Environment, Forest and Climate Change, 12767, Addis Ababa, Ethiopia
e-mail: zew172@yahoo.com; eyobtenkir1@yahoo.com

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The REDD+ process is framed in the Ethiopia's Climate Resilience Green Economy (CRGE) Strategy that sets out that by 2025, Ethiopia will become a middle-income country, resilient to climate change impacts and with a zero net increase in greenhouse gas (GHG) emissions over 2010 levels. REDD+ strategy contributes to the achievement of the CRGE targets through improved management of forests and agricultural areas. This strategy will be followed by a comprehensive action plan that elaborates the required activities to address the drivers of deforestation and forest degradation. The action plan will be incorporated into the Growth and Transformation Plan (GTP II) and implemented by the relevant sectors in line with existing sectoral policies and strategies.

8.1 The Origin of Carbon Markets

The UNFCCC (United Nations Framework Convention on Climate Change) was adopted at the 1992 United Nations Conference on Environment and Development in Rio de Janeiro ('Rio Earth Summit'). The main objective of the UNFCCC is to achieve stabilisation of GHG concentrations in the atmosphere at a level that would 'prevent dangerous anthropogenic interference with the climate system'. The UNFCCC entered into force on 21 March 1994, providing the guiding principles and architecture to assist Parties to meet that objective. There are now 196 parties to the UNFCCC.

The COP (Conference of Parties) is the supreme decision-making body of the Convention. All States that are Parties to the Convention are represented at the COP, and among its tasks highlights to promote the effective implementation of the Convention (institutional and administrative arrangements), and to review the national communications and emission inventories submitted by Parties. The COP meets every year from 1995 (COP1-Berlin) and COP Presidency rotates among the five recognized UN regions (Africa, Asia, Latin America and the Caribbean, Central and Eastern Europe and Western Europe and Others), so there is a tendency for the venue of the COP to also shift among these groups.

In 1997, the Parties to the UNFCCC adopted the Kyoto Protocol (signed at the third Conference of Parties to the UNFCCC, COP-3, in Kyoto, Japan, in December 1997), which entered into force on 16 February 2005. Currently, there are 192 Parties (191 States and 1 regional economic integration organization) to the Kyoto Protocol to the UNFCCC. The rules for the implementation of the Protocol were adopted at COP-7 in Marrakesh, Morocco, in 2001 ('Marrakesh Accords), and its first commitment period started in 2008 and ended in 2012. In Doha, Qatar, (COP-18) on 8 December 2012, the 'Doha Amendment to the Kyoto Protocol' was adopted, including new commitments in a second commitment period from 1 January 2013 to 31 December 2020 for Annex I Parties to the Kyoto Protocol. The Kyoto Protocol operationalised the UNFCCC by committing countries included in its Annex 1 (industrialised countries that were members of the Organisation for Economic Co-operation and Development at 1992 plus countries with economies in transition) to emissions reductions targets and establishing mechanisms to achieve those targets. Parties with commitments under the Kyoto Protocol (Annex B Parties) accepted targets for limiting or reducing emissions. These targets were expressed as levels of allowed emissions, or 'assigned amounts' over the 2008–2012 commitment period. These allowed emissions were divided into 'assigned amount units' (AAUs). During the first commitment period, 37 industrialized countries and the European Community committed to reduce GHG emissions to an average of five percent against 1990 levels. During the second commitment period, Parties committed to reduce GHG emissions by at least 18 percent below 1990 levels in the 8-year period from 2013 to 2020; however, the composition of Parties in the second commitment period is different from the first (only nine countries have ratified to date second commitment period of the Kyoto Protocol and all eyes are on the UNFCCC-COP21, 2015 in Paris, which offers an opportunity for convergence on concerted international climate action).

Emissions trading, as set out in Article 17 of the Kyoto Protocol, allows countries that have emission units to spare - emissions permitted but not used - to sell this excess capacity to countries that are over their targets. Thus, Annex B Parties, could participate in emissions trading (supplemental to domestic actions) for the purposes of fulfilling their reduction commitments. Thereby, a new commodity was created in the form of emission reductions or removals. Since carbon dioxide is the principal greenhouse gas, people speak simply of trading in carbon.[1] Carbon is now tracked and traded like any other commodity. This is known as the 'carbon market'. Carbon markets are means by which contracts of sale of emission reductions of greenhouse gases (GHGs) are exchanged.

Carbon markets exhibited rapid growth since its creation (the total value traded grew from USD 11 billion in 2005 to about USD 176 billion in 2011) but it began to decline because of the economic crisis starting in 2008/09. The fall in industrial production in these countries and the consequent reduction of GHG emissions associated, lowered demand for carbon assets necessary to prove compliance with emission targets, impacting heavily on prices. Currently, prices of carbon assets remain at historic lows and the prospects for global markets are reactivated are highly uncertain.

Nevertheless, in parallel, from 2011 began to proliferate initiatives to create domestic markets at regional, national and sub-national level, to promote the transition to low- emissions development in both developed and developing countries. These actions at domestic level have the potential to collectively overcome the international regulatory gap by fostering targeted low-carbon investments at regional and national level. Today, about 40 countries and over 20 sub-national jurisdictions are putting a price on carbon. Together, these carbon pricing instruments cover almost 6 gigatons of carbon dioxide equivalent (GtCO2e) or about 12 % of the annual global GHG emissions.

[1] 6 greenhouse gases are included in the Annex A to the Kyoto Protocol; Carbon dioxide (C02), Methane (CH4), Nitrous oxide (N20), Hydrofluorocarbons (HFCs), Perfluorocarbons (PFCs), and Sulphur hexafluoride (SF6)

8.2 Carbon Market's Assets

Three types of assets are traded in these markets: *Emission permits, Certificates of Emission Reductions originated in projects and Certificates of Voluntary emission Reductions:*

8.2.1 Emission Permits

These are the 'assigned amounts' over the 2008–2012 commitment period. These allowed emissions were divided into 'assigned amount units' (AAUs) and assigned by the governments of Annex I countries to companies that are emitting GHGs, depending on the emissions reduction commitments assumed under the Kyoto Protocol.

The emissions trading systems, also called 'cap-and-trade systems' is a flexible approach to address GHGs emissions based on the market. The basic principle of these schemes is setting a ceiling on the total amount of allowable emissions of certain greenhouse gases that can be emitted by the factories, power plants and other installations in the system for a period of time (cap). The cap is reduced over time so that total emissions fall. Each participant then receives a certain amount of allowances, which can then be traded in a market. The allocation of permits among the participants can be done in different ways, for example in terms of their historical emissions or by an auction process (companies must buy the permits they need; higher finance charges). After each year a company must surrender enough allowances to cover all its emissions, otherwise heavy fines are imposed. If a company reduces its emissions, it can keep the spare allowances to cover its future needs or else sell them to another company that is short of allowances. They can also buy limited amounts of international credits from emission-saving projects around the world. The limit on the total number of allowances available ensures that they have a value.

The main market is EU ETS – European Union Emission Trading Scheme, where EUAs – European Union Allowances are traded. Launched in 2005, the EU ETS is now in its third phase, running from 2013 to 2020 with significant modifications from the first stage; now there is a single EU-wide cap on emissions (instead national caps) and auctioning (instead free allocation) is now the default method for allocating allowances. Three hundred million allowances set aside in the New Entrants Reserve to fund the deployment of innovative renewable energy technologies as well as carbon capture and storage through the NER 300 programme (one of the world's largest funding programmes for innovative low-carbon energy demonstration projects; environmentally safe carbon capture and storage –CCS – and innovative renewable energy –RES- technologies).

8.2.2 *Certificates of Emission Reductions Originated in Projects*

These are generated when a specific mitigation project conducted in a developing country (CERs – Certified Emission Reductions, under the Clean Development Mechanism – CDM) or in Eastern Europe (ERUs - Emission Reduction Units under the Joint Implementation Mechanism -JI) proves that reduces the GHG emissions. CDM and JI are mechanisms under the Kyoto Protocol.

8.2.2.1 Clean Development Mechanism (CDM)

The main objective of the Clean Development Mechanism (CDM)[2] is to assist Parties included in Annex I (of the UNFCCC; the 40 countries plus the European Economic Community that agreed to try to limit their GHG emissions) (Annex B of the Kyoto Protocol of 38 countries plus the European Community that agreed to QELRCs, Quantified emission limitation or reduction commitment, emission targets), along with the QELRCs they accepted. The list is nearly identical to the Annex I Parties listed in the Convention except that it does not include Belarus or Turkey) in achieving compliance with their quantified emission limitation and reduction commitments and to assist Parties not included in Annex I in achieving sustainable development.

The CDM allows emission-reduction projects in developing countries to earn certified emission reduction (CER) credits, each equivalent to one tonne of CO2. These CERs can be traded and sold, and used by industrialized countries to a meet a part of their emission reduction targets under the Kyoto Protocol. In CDM projects, the Annex B country fund the project and provides any necessary know-how and technology transfer to the non-annex B country where the project is implemented. CDM works because emission reductions are many times more expensive to achieve in Annex B countries than in non-Annex B countries (the opportunities for emission reduction are bigger there).A CDM project must provide emission reductions that are additional to what would otherwise have occurred. The projects must qualify through a rigorous and public registration and issuance process. Approval is given by the Designated National Authorities. Public funding for CDM project activities must not result in the diversion of official development assistance. The mechanism is overseen by the CDM Executive Board, answerable ultimately to the countries that have ratified the Kyoto Protocol.

The market-based projects generated under the CDM mechanism is split into two: the primary market and the secondary market. In the primary market, the mitigation projects developers in the developing world sell their certified emission reductions (CERs) (issued by the CDM Executive Board) to a buyer in the developed

[2]Art. 12 KYOTO PROTOCOL TO THE UNITED NATIONS FRAMEWORK CONVENTION ON CLIMATE CHANGE, 1998.

world, by signing an ‘Emissions Reduction Purchase Agreement’ (ERPA). The transactions are conducted mostly through forward contracts, ie, before licenses are actually issued. Therefore, the prices of primary CERs depend largely on the intrinsic risk of each project. The risks associated with primary CERs are multiple; risk that the volume actually delivered is less than the agreed volume in the ERPA, no registration risk, risk of delays, risk of a specific type of project is not accepted in the EU ETS and uncertainty about what will occur after Kyoto first commitment period. In the secondary market, issued CERs are traded. Here trade occurs between financial operators and do not involve directly to project developers in developing countries, so a strong activity in the secondary CDM market does not imply that effectively being financed and implemented numerous mitigation projects in developing countries, as it does in the primary market. The prices of secondary CERs are highly linked to the activity of the European market (EU ETS).

Anyway, there is an important link between the primary and secondary CERs market: if there are delays in certification and delivery of primary certified, numerous problems for brokers and sellers in the secondary CERs market are generated as traders do not have sufficient assets to meet its delivery commitments and are forced to buy CERs on the spot market (a higher price) to meet their obligations agreed. To hedge against these risks, it is increasingly widespread the use of financial derivatives, primarily calls (options), which shows the increasing sophistication of carbon markets, especially the CDM market.

Operational since the beginning of 2006, the mechanism has already registered (31/03/2015) 7622 project activities, with a potential supply of 2,193,121,580 tonnes of CO2 equivalent in the first commitment period of the Kyoto Protocol, 2008–2012, or 4,723,080,924 CERs to the end of 2015. The number of CDM project activities that have issued CERs is 2582 with 1,545,952,932 to the date (31/03/2015).

The CDM allows crediting from afforestation and reforestation (A/R) projects, but excludes REDD+ and other forest carbon activities. A/R projects hold a very minor share of the CDM market, in part due to the temporary credits issued for these project types, but also because of their exclusion in the European Union Emissions Trading System. A/R credits through June 2013 comprise only 0.5 % of total credits issued under the CDM. There are to the date (31/03/2015) 55 registered projects; 19 in Central and South America, 17 in Africa, 16 in Asia and 3 in Eastern Europe, with 2,107,349 estimated emission reductions in metric tonnes of CO2 equivalent per annum (as stated by the project participants).

8.2.2.2 Joint Implementation Mechanism

The objective of the Kyoto Protocol’s Joint Implementation (JI) is to allow Annex I countries a more flexible and potentially cost-effective means to fulfill their Kyoto commitments. JI allows a country with an emission reduction commitment (i.e. Annex B Parties) to generate emission reduction units (ERUs) from domestic projects and sell them to another Annex I country, which then uses them to help meet its Kyoto target.

Joint Implementation is a project-based mechanism like the Clean Development Mechanism but some of the accounting is different because JI projects are nested within countries that have emission reduction commitments under the Kyoto Protocol. Emission Reduction Units (ERUs) generated by non-land-use JI activities are created through a conversion of Assigned Amount Units (AAUs), while for land-use (including forestry) projects this is through a conversion of Removal Units (RMUs) generated by the country as a result of national level net sequestration. A JI project must provide a reduction in emissions by sources, or an enhancement of removals by sinks, that is additional to what would otherwise have occurred. Projects must have approval of the host Party and participants have to be authorized to participate by a Party involved in the project. Projects starting as from the year 2000 may be eligible as JI projects if they meet the relevant requirements, but ERUs may only be issued for a crediting period starting after the beginning of 2008.

Joint Implementation projects may be conducted under either of two tracks. Track 1 is for Parties to the Kyoto Protocol that have an assigned amount calculated and recorded, a national system for the estimation of emissions, a national registry, and have submitted annual inventories of GHG emissions and supplementary information on its assigned amount. Track 1 allows the host country more control and imposes fewer external requirements. Track 2 applies to countries that don't meet Track 1 eligibility and projects therefore require additional approvals (an independent entity accredited by the Joint Implementation Supervisory Committee, JISC, that has to determine whether the relevant requirements have been met before the host Party can issue and transfer ERUs).

To date 597 projects were conducted under track1 and 51 under track2. Two projects in LULUCF of all of them under track1 (afforestation projects in Romania, Total Emission reductions expected 410046.0 t CO2 equivalent, and Russian Federation, Total Emission reductions expected 1768894.0 t CO2 equivalent) and one project under track2; the Bikin Tiger Carbon Project - Permanent protection of otherwise logged Bikin Forest, in Primorye Russia. (Emission reductions/enhancements of removals annual average in metric tonnes of CO2 equivalent 156,438 t CO2 equivalent and Verified emission reductions/enhancements of removals: 519,512 t CO2 equivalent).

8.2.3 Certificates of Voluntary Emission Reductions

These are certificates traded in the voluntary carbon markets. The so-called 'voluntary carbon market' includes all transactions of carbon credits that are not governed by a regulatory obligation to comply with a goal of reducing GHG emissions. This includes both; credit transactions created especially for the voluntary markets (such as VERs – Verified Emission Reductions) and transactions in which credits from regulated markets (such as CERs from CDM) are sold to buyers looking to voluntarily offset their emissions.

Until 2010, transactions in the so-called 'voluntary carbon market' could be divided mainly into two segments: those made under the Chicago Climate Exchange

(CCX), North America's program to reduce GHG emissions, and transactions "over the counter" (OTC), ie direct transactions between two parts by a financial intermediary (broker). Between 2003 and 2010, the CCX operated as a cap-and-trade voluntary, but legally binding, with a compensatory component. In 2011, however, the CCX ceased operations and voluntary transactions are carried out since then through OTC segment (97 %) and through some private platforms.

The OTC market was initially characterized by a lack of rules and regulations, until a group of organizations developed a series of voluntary standards and methodologies by which project developers could certify their GHG emission reductions and ensure thoroughness of their baselines.

Specifically, the OTC market demand can be divided into buyers "purely voluntary" and buyers "pre-compliance". The first buy credits to offset their own emissions and are primarily guided by ethical and / or corporate social responsibility motivations. Therefore, its demand curve does not have much relation to the regulated carbon markets. Instead, buyers "pre-compliance" acquire VERs with two objectives: to buy credits at low prices in order to use them in the future to demonstrate achievement of goals or to sell at a higher price to entities which are regulated in future schemes cap-and-trade required. Companies pursuing the first of these objectives are usually entities likely to be regulated in the future, while companies with the second goal are usually financial institutions.

At this point it is noteworthy that, unlike what happens in compliance markets as demand in the MVC does not depend on the obligation to comply with a given reduction of GHG emissions, the market is fragmented and there is no information impartial and centralized. This lack of obligation, consistency, transparency and centralized register, in the voluntary market make prices usually lower than those in regulated markets and demand is low, erratic and volatile.

However, the voluntary market does not suffer the bottlenecks that occur in the CDM and includes types of projects that the CDM does not provide, such as projects to reduce emissions from deforestation and forest degradation (REDD +, for its acronym in English). Also, for some environmentalists the voluntary market is an important tool to educate the public about the threat of climate change and the relevance of individual mitigation action.

Various programs and standards have been developed under the voluntary carbon market: the most relevant ones so far are the Verified Carbon Standard (VCS), Gold Standard (GS) and Climate Action Reserve (CAR).

The VCS (Verified Carbon Standard, originally called Voluntary Carbon Standard) (http://www.v-c-s.org/) became operational in March 2006 in order to provide uniformity to the voluntary market and credibility to certificates of voluntary emission reductions (VERs - Voluntary Emission Reductions). After a consultation process involving multiple actors, a new version of the standard known as 'VCS 2007' was released in late 2007, which became one of the voluntary standards most currently used internationally. A third version[3] was launched in March 2011,

[3] This new version also included an important change in the nomenclature of the standard: VCS was renamed as 'Verified Carbon Standard' instead of 'Voluntary Carbon Standard'.

with relevant modifications that included a better functionality and clarifications on rules and procedures.

The objectives behind the creation of VCS were (i) standardize and provide transparency and credibility to the voluntary carbon market, (ii) increase the confidence of businesses, buyers and governments in voluntary reductions, (iii) create a unit of voluntary emissions reduction that is credible and marketable: VCU (Voluntary Carbon Unit), (iv) attract additional funding for projects that reduce emissions, (v) stimulate innovation in mitigation technologies and (vi) provide a transparent system preventing the double accounting (a public central database for projects and VCS Records was created).

Gold Standard (GS) (http://www.goldstandard.org/) is a non-profit organization funded by 60 NGOs operating a certification scheme to ensure quality carbon credits. This foundation registers projects that reduce emissions of greenhouse gases and certifies the actual emission reduction by issuing credits called 'GS VERs' (Gold Standard Voluntary Emission Reductions). These credits can then be sold both in the voluntary and compliance markets.

The project registration takes place in the Gold Standard Registry, a system to track all certified projects and commercialize GS VERs credits in the world. The Gold Standard focuses on Energy Efficiency, Renewable Energy, Waste Management, Land Use & Forest, and Sustainable Water projects.

Regarding Land Use & Forest projects, various sustainable activities can be included for certification; Afforestation/Reforestation, supporting sustainable agriculture practices, and Improved Forest Management (IFM); focused through a landscape approach on ecosystem services.

The total number of Gold Standard projects as of 1st January 2015 is 262 registered projects (107 validated) with a ratio 35/65 of compliance to voluntary market projects and 40 m + GS tonnes CO2e issued.

The Climate Action Reserve (CAR) is an American program that seeks to ensure the integrity, transparency and financial value of the carbon market through the development of regulatory and quality standards for development, quantification and verification projects to reduce GHG emissions in United States. The initiative also seeks to provide track credit (Climate Reserve Tonnes issued, CRTs) transactions through a transparent and publicly accessible system.The Reserve is a 501(c)3 non-profit organization under the Internal Revenue Code.

At the moment, the following are only eligible projects for which the program has protocols: Coal Mine Methane Projects, Forest Projects (Improved Forest Management, Reforestation, Avoided Conversion), Landfill Projects, Livestock Projects, Nitric Acid Production Projects, Nitrogen Management Projects, Organic Waste Composting Projects, Organic Waste Digestion Projects, Ozone Depleting Substances Projects, Rice Cultivation Projects and Urban Forest Projects. The Reserve's offsets have been used by small businesses, international corporations, high profile events, universities and professional sports teams to balance their emissions.

The numbers so far are 306 Account holders and 220 Projects registered with 53,081,480 CRTs issued (http://www.climateactionreserve.org/).

Other programs and standards developed under the voluntary carbon market are: American Carbon Registry Standard (ACRS, http://americancarbonregistry.org/), Climate, Community, and Biodiversity Standards (CCB, http://www.climate-standards.org), ISO 14064/65 Standards (http://www.iso.org/iso/home.html), Panda Standard (the first voluntary carbon standard designed specifically for China CBEEX, *China Beijing Environment Exchange, and* BlueNext, Forestry and Agriculture in China, http://www.pandastandard.org/), Plan Vivo (Forestry and Agroforestry projects, http://www.planvivo.org/), SOCIALCARBON Standard (http://www.socialcarbon.org/).

8.3 State and Trends of Carbon Markets

At a time when the international climate negotiations trust on an agreement in Paris (COP21) to revive private sector confidence to invest in carbon markets, the actions at domestic level have the potential to collectively overcome the international regulatory gap by fostering targeted low-carbon investments at regional and national level. There are about 40 countries and over 20 sub-national jurisdictions putting a price on carbon, through various carbon pricing instruments (covering almost 6 gigatons of carbon dioxide, equivalent (GtCO2e), 12 % of the annual global GHG emissions).The challenges for these domestic initiatives are scaling up GHG emission reductions and lowering the cost of mitigation. Carbon pricing instruments such as carbon taxes emissions trading schemes, and crediting mechanisms are basic to internalize the external cost of climate/ change.

Meanwhile other countries repeal their carbon pricing mechanism legislation (Australia) or pull out of the international agreements on Climate Change (Japan, New Zealand and Russia, officially left the second commitment period of the Kyoto Protocol). On the other hand many players have either exited the market-based mechanisms under the Kyoto Protocol or substantially reduced their activities (the short demand for international credits has led to an intensified exodus of private sector players). The potential demobilization of the Clean Development Mechanism (CDM) market infrastructure and the lack of confidence in the EU ETS due to its inability to cope with the economic downturn seriously impacted in the prices: EU ETS remained in the range of about US$5–9 in 2014 contrasting with US$18 showed 3 years ago and CERs worth just US$0.51.

Carbon prices between schemes occupy a significant range, from under US$1/tCO2 in the Mexican carbon tax up to US$168/tCO2 in the Swedish carbon tax, most clustered under US$12/tCO2, but the main reason for these lower prices in emissions trading is probably due to taxes often exempt industry and put the tax burden on private households avoiding competitiveness and carbon leakage.

In carbon taxation schemes the prices are set and therefore reflect a range of political realities and goals, explaining the variety between countries. However, the majority of prices in existing systems are below $35/tCO2. Prices should stimulate low-carbon investments at scale and maximize mitigation in support of the transformation required to address the climate challenge.

From the perspective of economic efficiency, the desirable design features for carbon pricing schemes should include (Ian Parry in World Bank 2014): a comprehensive coverage of emissions, which can be achieved through implementing pricing in proportion to carbon content on the supply of petroleum products, coal, and natural gas; a uniform price applied to all emissions, stable and predictable emissions prices, emissions prices aligned with environmental damages or climate stabilization goals (estimates of future climate change damages suggest CO2 should be priced in the order of $35 per ton), maximizing the fiscal dividend, which means raising revenues and using the revenues productively and carefully targeted compensation schemes for vulnerable households and firms.

8.4 REDD+ Mechanism

According to the conclusions reached by IPCC Working Group I in its contribution to the IPCC Fifth Assessment Report (IPCC 2013), the atmospheric concentrations of the greenhouse gases; carbon dioxide (CO2), methane(CH4), and nitrous oxide (N2O) have all increased since 1750 due to human activity, in the case of carbon dioxide, exceeding the pre-industrial level by about 40 %.

The contribution of the GHG emissions to the global mean surface warming is likely in the range of 0.5 °C to 1.3 °C over the period 1951–2010 (with further contributions from other anthropogenic forcings, including the cooling effect of aerosols, natural forcings and from natural internal variability). Together these assessed contributions are consistent with the observed warming of approximately 0.6–0.7 °C over this period.

Annual net CO2 emissions from anthropogenic land use change (including deforestation and forest degradation) were 0.9 [±0.8] GtC/yr.[4] on average during 2002 to 2011 (medium confidence); that is a 9.8 % of the annual anthropogenic CO2 emissions (considering also fossil fuel combustion and cement production: 8.3 [±0.7] GtC/yr. averaged over 2002–2011, high confidence). But the cumulative anthropogenic emissions of CO2 from anthropogenic land use change from 1750 to 2011 are estimated as 180 [±80] GtC, a 32.4 % of the total anthropogenic CO2 emissions, 555 [±85] GtC (a 28.8 % of this total has accumulated in natural terrestrial ecosystems; cumulative residual land sink).

It is clear that forests play a relevant role in the control of GHG emissions because they can absorb carbon dioxide (CO2) from the atmosphere (sink), but also release this CO2 back into the atmosphere (source) when they are damaged (deforestation and forest degradation). Forests cover around 31 % percent of total land area (over four billion hectares, FRA 2010). Between 2000 and 2010 there was a net loss of 5.2 million hectares per year, mostly concentrated in tropical regions (South America suffered the largest net loss of forests, about 4.0 million hectares per year, followed by Africa, which lost 3.4 million hectares annually), being commodity

[4] 1 GtC = 1 Gigatonne of carbon = 10^{15} grams of carbon and it corresponds to 3.667 GtCO2.

production responsible for over 50 % of deforestation and 60 % of forest degradation in tropical and subtropical areas.

In 2010, the estimated total carbon stock in forests was 652 GtC (or 161.8 tC/ha), as the sum of carbon stock in biomass, above-ground (AGB) and below-ground biomass (BGB) (289C Gt, or 71.6 tC/ha, 44 % of the total; most countries used the default carbon fraction from the 2006 IPCC guidelines of 0.47), carbon stock in dead wood and litter ((72 GtC, or 17.8 tC/ha, 11 % of the total) and the carbon stock in soil (292 GtC or 72.3 tC/ha, 45 % of the total). The trends from 2000 to 2010 showed a decrease in all carbon stocks (5.0 GtC, 0.03 GtC, 3.4 GtC), 8.5 GtC in total as a result of the loss of forest area; however the carbon stock per hectare has remained almost constant. According to these estimates, the world's forest is therefore a net source of emissions due to the decrease in total forest area. Aside from the release of heat-trapping carbon dioxide (CO2) into the atmosphere, the tropical forest loss has direct devastating effects on biodiversity and forest-dependent communities.

Solving the problem of deforestation is not only a prerequisite for any effective response to climate change but also the action needed for biodiversity conservation, protection of forest-dependent communities, watershed protection, etc., in brief, conservation and promotion of all ecosystem services associated with forest.

REDD+ has emerged in response to this, as an international initiative that aims to provide financial incentives to developing countries for 'reducing emissions from deforestation and forest degradation'. This initiative could soon be included as a key climate change mitigation mechanism within a new global climate change agreement, to be negotiated by 2015 under the United Nations Framework Convention on Climate Change (UNFCCC).

For forest countries preparing to implement REDD+ and receive results-based payments, a clear domestic legal framework of enabling policies and legislation is needed to ensure that national systems not only deliver permanent emission reductions, but can also guard against the social and environmental risks created by REDD+, while also delivering co-benefits. A variety of international requirements and guidance exist for forest countries looking to participate in REDD+, including those adopted by Parties to the UNFCCC, as well as those defined by a number of multilateral and bilateral REDD+ initiatives, which are already delivering finance for 'REDD+ readiness'.

8.4.1 Origin and Development of REDD+ Mechanism

Decisions made by the COP since 2005 have designed and built the REDD+ regime as a global mechanism to be implemented in phases through the elaboration or improvement of specific national legal frameworks and policies in developing countries. Robust REDD+ regimes will be more likely to attract the financial support that will consolidate the role of REDD+ as an important element of sustainable development. Regarding the effective work of the COP, there are two subsidiary bodies to the Convention established by the COP/CMP (CMP is the Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocol); the Subsidiary Body for Scientific and Technological Advice (SBSTA) that supports the work of the COP/CMP through the provision of scientific and technological advice, and the Subsidiary Body for Implementation (SBI) that assess and reviews of the effective implementation of the Convention and its Kyoto Protocol and advises COP/CMP on budgetary and administrative matters.

In 2005, at COP 11 in Montreal, two countries, Papua New Guinea (PNG) and Costa Rica, made a submission to the COP to include the reduction of emissions from deforestation in developing countries on the agenda. The submission to the SBSTA called for economic incentives to reduce emissions from deforestation in developing countries (RED) under a global approach to involve industrialized and developing countries in its implementation. It was suggested to create a new protocol under the UNFCCC for the implementation of RED demonstration activities that allow parties to the Kyoto Protocol to use these RED credits to meet their emissions reductions commitments.

In the following 2 year discussions about the role of forests in reducing emissions in developing countries a consensus was reached to consider a broader scope to RED, in particular to include forest degradation as a large source of emissions.

In 2007, the 'Bali Action Plan' agreed at COP 13, included reducing deforestation and forest degradation (REDD) and also proposed to consider positive incentives on activities as conservation, forest's sustainable management and enhanced of forest carbon stocks in developing countries. It was also called on industrialised countries to make financing available for REDD technical implementation, and on developing countries to undertake REDD demonstration activities. Two negotiating tracks resulted under the 'Bali Road Map'; the Ad Hoc Working Group on Long-term Cooperative Action (AWG-LCA) and the Ad Hoc Working Group on Further Commitments for Annex I Parties under the Kyoto Protocol (AWG-KP established in Montreal, 2005). Their purposes were to conduct an effective process to enable the full implementation of the Convention through long-term cooperative action, and to negotiate emissions reductions commitments for Non-Annex-I countries and the United States, which failed to ratify the Kyoto Protocol, respectively.

In 2009, in the 'Copenhagen Accord' (decision 2/CP.15) the concept of REDD+ was formally established, and the 5 activities were agreed in the context of reducing emissions from forests in developing countries (deforestation, forest degradation, conservation, sustainable management of forests and enhancement of forest carbon stocks).

'Cancun agreements' (COP16) were very relevant for REDD+ definition, development and implementation; (i) the list of eligible REDD+ activities was finally settled, to include avoiding deforestation and degradation, conservation and the sustainable management of forests and enhancement of forest carbon stocks, (ii) it was established the policy framework for REDD+ negotiations (developing countries were requested to develop a national strategy or action plan, national or subnational forest reference levels, FRL , and a national forest monitoring system, NFMS) and (iii) a list of safeguards mechanisms was created to ensure environmental and social outcomes would be protected through REDD+ interventions.

In 2011 in Durban at COP 17, parties advanced in the elaboration on the mechanism for reporting on safeguards and allowed for FRELs/FRLs to be developed at a subnational level as an interim measure as national systems evolved (including always all significant activities and carbon pools). Also some progress on REDD+ financing was made including an agreement that developed countries could provide 'new, additional and predictable' results-based finance from 'a wide variety of sources, public and private, bilateral and multilateral'; both a market-based approach to REDD+ and a non-market approach (mitigation and adaptation) could be developed.

In 2012 in Doha at COP18, it was established a work programme on results-based finance for the development of an international REDD+ mechanism. The work programme aimed to contribute to the ongoing efforts to scale up and improve the effectiveness of REDD+ finance, to specifically consider institutional and governance arrangements, non-market based approaches to REDD+ finance and methodological issues related to non-carbon benefits.

In 2013 in Warsaw at COP 19, most of the key elements of REDD+ were stablished through relevant decisions on operational issues and financing configuring the 'Warsaw REDD+ Framework'. On operational/technical issues, it was agreed a set of (i) 'Modalities for National Forest Monitoring Systems' and a set of (ii) 'Modalities for Measuring, Reporting and Verifying' (consistent with previous UNFCCC guidance, maintaining transparency, completeness, consistency and accuracy, developing national capacities and engaging local communities in monitoring and reporting), a proposal to estimate (iii) FRLs/FRELs, (iv) advances on Parties' reporting on safeguard compliance (v) and a process and guidelines for assessing Parties. Regarding finance and coordination, it was requested interested Parties to designate a focal point or national entity to serve as a liaison for coordination with the UNFCCC and related bodies.

Further decisions have recognised the importance of scaling up predictable financial and technological support, and also the key role of the new Green Climate Fund, GCF (Decision 1/COP16); created in the framework of the UNFCCC as a mechanism to assist the developing countries in adaptation and mitigation practices to counter climate change. On this financial regard, the COP has provided guidance for the development, submission and technical assessment of FREL/FRLs,[5] necessary to access international finance linked to positive results from implementing REDD actions. There is currently no operational financing mechanism under the UNFCCC that provides payments for REDD+ results although the COP has advanced on defining a number of requirements to design and implement this mechanism in the future.

[5] Forest Reference Emission Levels or Forest Reference Levels (if emissions by sources and removals by sinks are both considered) are benchmarks for assessing country's performance in implementing REDD+ activities. Through the analysis of historic emissions during a reference period, a linear trend extrapolation or a historic average (e.g.) could be projected and used as benchmark for measuring emission reductions from REDD+ activities during a specified period (Decision 12/COP17).

At COP 20, in Lima 2014, a successful and timely initial resource mobilization process of the GCF was announced. The mobilization of USD 10.2 billion by contributing Parties, enables the GCF to start its activities in supporting developing country Parties of the Convention, and making it the largest dedicated climate fund.

It was also requested during COP 20 to accelerate the operationalization of the adaptation and mitigation windows, to ensure adequate resources for capacity-building and technology development and transfer, to accelerate the operationalization of the private sector facility, and to complete its work related to policies and procedures to accept financial inputs from non-public and alternative sources.

Parties prepare for the challenges of reaching a new international climate agreement in Paris 2015 at COP21, and the potential contribution that REDD+ can make to an effective outcome will ensure its own future. GCF may provide results-based payments for REDD+ in the future, but also a new financing mechanism associated with this new international climate agreement could be negotiated and agreed before GCF becomes operational on this regard. At the same time, implementation of REDD+, funded through non-UNFCCC REDD+ mechanisms, is occurring and taking a number of different approaches from REDD+ project level to the implementation of national REDD+ schemes. In addition to bilateral agreements, specific programmes focused on finance results-based REDD+ actions has been developed.

8.4.2 REDD+ Under Other Non-UNFCCC Mechanisms

Several initiatives have been launched to facilitate implementation of the UNFCCC REDD+ program: multilateral initiatives; such as the World Bank's Forest Carbon Partnership Facility (FCPF), bilateral support programs; such as Norway's International Climate and Forest Initiative (NICFI), and private and not for profit entities; such as the Verified Carbon Standard (VCS) Association). These mechanisms basically support the first phase of domestic REDD+ implementation ('Readiness') and in some cases provide methodological frameworks or standards for the implementation of REDD+ project activity. The non-UNFCCC REDD+ mechanisms (e.g. FCPF, VCS) provide countries with detailed prescriptions for implementing REDD+ that are consistent with the very flexible principles set out in the UNFCCC REDD+ rules.

The FCPF Methodological Framework is a set of 37 criteria and related indicators (C&I), associated with five major aspects of Emission Reductions Programs: level of ambition, carbon accounting, safeguards, sustainable program design and implementation, and ER Program transactions. ER Programs proposed by REDD+ Countries to the Carbon Fund are expected to demonstrate conformity with the Framework's criteria. The Methodological Framework was approved by Carbon Fund Participants at the 8th meeting of the Carbon Fund on December 9, 2013.[6]

[6] Carbon Fund. Methodological Framework. Forest Carbon Partnership Facility. December 20, 2013.

In 2013, the VCS launched the Jurisdictional and Nested REDD+ (VCS-JNR) framework, which provides guidance on accounting and crediting REDD+ programs implemented by national and/or subnational governments. Within the Agriculture, Forestry and Other Land Use (AFOLU) sector (sectoral scope 14) various methodologies have been approved for several categories, including Agricultural Land Management (ALM), Improved Forest Management (IFM) and Reduced Emissions from Deforestation and Degradation (REDD). The last version of VCS REDD+ Methodology Framework (REDD-MF), v1.5, March, 2015,[7] provides a set of modules for quantifying GHG emission reductions and removals from deforestation and forest degradation (extraction of wood for fuel).

8.4.3 Scope and Scale of REDD+ Activities

8.4.3.1 Scope

The results-based activities within the scope of REDD+ (Decision 1/CP.16) should be consistent with the national policies to promote the sustainable management of forests and in line with the mitigation activities that the country wish to finance and implement: (i) reducing emissions from deforestation, (ii) reducing emissions from forest degradation, (iii) conservation of forest carbon stocks, (iv) sustainable management of forests, (v) enhancement of forest carbon stocks.

8.4.3.2 Scale

UNFCCC REDD+ rules offer countries some flexibility for subnational interim implementation of some of their elements (RELs/RLs, M&MRV) and also encourage REDD+ demonstration activities at project level, but focus on creating a system implemented at the national level. At national scale the risk of reversal, additionality and leakage, is easier to address than at the project level.

The non-UNFCCC REDD+ mechanisms consider three different approaches; jurisdictional ('jurisdiction' is either at the national or subnational level), project-level or multi-scale nested approach. At national scale incentives flow to the national government, based on performance against a national reference level. At subnational scale incentives would typically flow to the subnational governmental entity (e.g. a state, province, municipality, etc.), based on performance on performance against a subnational reference level, but also the national government could be the beneficiary of the emissions reductions generated allocating or not some of these incentives to subnational governments. At project-level scales, the incentives flow directly to project developers based on performance against a project baseline. At multi-scale nested project-level activities are integrated into an accounting and

[7] VCS Methodology VM0007 REDD+ Methodology Framework (REDD-MF). March 9, 2015.

incentive scheme of a larger jurisdiction (sub-national or/and national). Incentives flow directly to subnational entities and project developers in addition to national governments.

8.4.4 Phased Implementation Process

The phased implementation approach for REDD+ program established in Cancun Agreement was developed to provide countries with the confidence to begin work on REDD+ and obtain funding for REDD+ readiness-type activities while allowing institutional and technical capacities to be developed. Phased implementation process of REDD+ program considers the following steps:

1. Phase one: the national strategies, the policies needed and the capacity building are developed. Supported through multilateral and bilateral channels.
2. Phase two: REDD+ policies, national strategies involving further capacity-building, technology development and transfer relating to REDD+, and results-based REDD+ demonstration activities are implemented. Supported through multilateral and bilateral channels.
3. Phase three: results-based actions fully measured, reported and verified (MRV) are implemented.

8.4.5 The National REDD+ Strategy

The Cancun Agreement and the Warsaw Decisions set out a number of institutional policies and processes that developing countries are requested to develop as part of their REDD+ programs. These national action plans are requested to address at least five relevant issues: the drivers of deforestation and forest degradation, land tenure issues, forest governance issues, gender considerations and safeguards, ensuring the effective participation of relevant stakeholders and local communities. REDD+ countries are requested to develop and implement:

1. A national strategy or action plan,
2. A national forest reference emission level and/or forest reference level, or as an interim measure, a subnational forest reference emission levels and/or forest reference levels,
3. A robust and transparent national forest monitoring system (or a subnational monitoring and reporting as an interim measure), for the monitoring and reporting of the activities referred under the scope,
4. A system to report on safeguards,
5. A focal point to communicate with the UNFCCC Secretariat regarding REDD+ implementation and other matters.

Regarding the institutional arrangements for REDD+, parties have to designate a national REDD+ entity to serve as a focal point for communications and also may be.

nominated to obtain and receive results-based payments. These REDD+ entities should meet regularly with the relevant financing entities to discuss about REDD+ implementation (sharing information, identifying needs and gaps in coordination of support, providing recommendations).

8.4.6 Forest Reference Emission Levels and/or Forest Reference Levels (FRELS/FRLS) and M&MRV Systems

FREL/FRLs are benchmarks for assessing each country's performance in implementing REDD+ activities (FREL includes only emissions from deforestation and degradation and FRL includes both emission by sources and removals by sinks) and M&MRV refers to the functions that the required National Forest Monitoring System must fulfil for forest monitoring and for estimation and international reporting of forest emission and removals.

FRELs/FRLs have to be transparently established and consistently with the anthropogenic forest-related GHG emissions by sources and removals by sinks contained in each country's GHG inventories. Parties usually follow a 'step-wise' approach to develop them , updating their FRELs/FRLs periodically to incorporate better or new data, additional pools and improved methodologies (also subnational forest FRELs and/or FRLs may be established as an interim measure).

The main reason for developing FREL/FRL is to access international finance linked to positive results from implementing REDD+ actions, and their requirements (guidance for submissions and technical assessment), included also the M&MRV system's specifications, depend on the specific program focus on finance for results-based REDD+ actions. Waiting for an UNFCCC initiative in this regard (more detailed guidance framework) and in addition to bilateral Agreements there are four specific programmes: the Forest Carbon Partnership Facility (FCPF) Carbon Fund (the only program that has explicit guidance in its Methodological Framework for the development of Reference Levels), Germany's REDD Early Movers program (REM), Biocarbon Fund's Initiative for Sustainable Forest Landscapes (ISFL) and the Verified Carbon Standard's Jurisdictional and Nested REDD+ (VCS-JNR).

In the UNFCCC framework parties can voluntarily submit the proposed FRELs/FRLs for technical assessment together with complete information (transparent, complete and accurate and be guided by the most recent IPCC guidance) regarding their rationale for the development. Submissions should include information on forest definition used, scale and scope, historical data sets, methods and models used, and pools and gases considered in the FREL/FRL.

On the other hand parties are requested to establish a National Forest Monitoring Systems (NFMS), for the purposes of monitoring and reporting of REDD+ activities and estimating anthropogenic forest-related GHG emissions by sources and removals by sinks, forest carbon stocks and forest area changes. If appropriate subnational forest monitoring and reporting may be used as an interim measure.

The NFMS should follow the most recent IPCC guideline for estimating anthropogenic forest related GHG emissions, and it should have two basic pillars according the considered functions: monitoring function to allow assess a broad range of forest information and MRV (measurement, reporting and verification) function for REDD+ refers to the estimation and international reporting of forest emission and removals.

The NFMS should provide transparent data and information, consistent over time, and suitable for measuring, reporting and verifying GHG emissions resulting from REDD+ activities and for NAMAs (Nationally Appropriate Mitigation Actions; defined in two contexts, National level and Individual Action level, to reduce emissions in developing countries as national governmental initiatives, relative to ‘business as usual’ emissions in 2020).

The MRV system should be, as required, a robust and transparent national forest monitoring system for the monitoring and reporting REDD+ activities, providing estimates that must be transparent, consistent (over time and with the established FRELs/FRLs) and accurate (taking into account national capabilities and capacities). All REDD+ results based actions should be fully measured, reported and verified. It will use a combination of remote sensing and ground-based forest carbon inventory – National Forest Inventory - approaches for estimating anthropogenic forest-related greenhouse gas emissions by sources and removals by sinks, forest carbon stocks and forest area changes.

The MRV system will be designed to be able to accommodate multiple stakeholders. At the national level, a coordination mechanism will be put in place to provide a link between policy and practice at different scales. Additionally MRV-related activities and arrangements will be linked to existing relevant structures, including higher education and research institutions and ongoing monitoring activities at the local level.

As the REDD+ scheme is expected to deliver emission reductions and other co-benefits, the MRV system will be designed to help track a range of other indicators such as biodiversity and social benefits. The national MRV system should also consider the development of innovative participatory approaches aimed at engaging forest-dependent communities in monitoring and verification work build understanding and local ownership.

REDD+ countries may submit a technical annex for technical assessment with their biennial update reports, including the results achieved from REDD+ activities expressed in tonnes of CO2 eq. Two LULUCF experts from the UNFCCC roster of experts will analyse the technical annex for consistency in methodologies, definitions, comprehensiveness and the information provided between the assessed REL/RL and the results of the implementation of REDD+ activities and will publish a technical report on the UNFCCC website identifying areas for technical improvement.

8.4.7 Safeguards

The safeguards are a set of agreed policies and procedures that seek to minimise or mitigate (or avoid) adverse environmental and social impacts of REDD+ activities. These are the basis for achieving 'co-benefits' from REDD+ activities (improving participation, forest governance and natural resources management), but also critical to achieve the direct carbon benefits from REDD+ (providing both governments and project developers a social and environmental warranty to operate, minimising for conflicts and assisting REDD+ to operate at a larger scale).

REDD+ Parties should implement REDD+ activities promoting and supporting:

1. National forest programmes under relevant international conventions and agreements,
2. Transparent and effective forest governance structures,
3. Respect for indigenous and local community knowledge and rights (UN Declaration on the Rights of Indigenous Peoples, UNDRIP),
4. Effective stakeholder participation (particularly indigenous peoples and local communities),
5. Conservation of natural forests and biological diversity,
6. Methods to address the risks of reversals,
7. Methods to reduce displacement of emissions (or leakage that occurs when deforestation and/or forest degradation avoided in one forested area is 'displaced' to another forested area).

These safeguards should support national strategies and be included in all phases of implementation. On this regard REDD+ parties are requested to develop a system for providing information on how safeguards are being addressed and respected (Safeguard Information System or SIS).

SIS should provide transparent, consistent, updated and accessible information on how all of the mentioned safeguards are being addressed and respected. SIS should be flexible to allow for improvements over time, country-driven and implemented at the national level and build upon existing systems, as appropriate. Also the NFMS may provide relevant information on how safeguards are being addressed and respected.

A Safeguard Summary (explaining how safeguards are being addressed and respected throughout the REDD+ implementation) should be provided by REDD+ countries after the start of the implementation of REDD+ activities. This document should be provided periodically and be included (at least) in the information hub (information on results-based actions and payments, assessed FRELs/FRLs, safeguards information, National REDD+ Strategy and NFMS) and national communications (which is on average every 4 years), and on a voluntary basis, via the UNFCCC REDD Web Platform.

SIS and Safeguard Summary are prerequisites for the receipt of results-based finance.

8.4.8 Addressing Drivers of Deforestation and Forest Degradation

REDD+ parties should identify options and actions to address drivers of deforestation and forest degradation as part of their national strategies. Drivers of deforestation are usually classified as proximate/direct causes (e.g. human activities that directly impact forest cover and loss of carbon, such as agricultural expansion, infrastructure extension and wood extraction) and underlying/indirect causes (interactions of social, economic, political, cultural and technological processes).

The analysis of causes of deforestation and forest degradation and the identification and prioritization of strategic options to address those in a REDD+ party and in the framework of REDD Readiness process is the best way to ensure that REDD+ activities will address these phenomena.

REDD+ Countries, relevant organisations, private sector and other stakeholders should take action to address and reduce the drivers of deforestation and forest degradation, sharing the information and the results of that work.

8.4.9 REDD+ Finance

Finance for REDD+ implementation may come from a variety of sources (public and private, bilateral and multilateral, market or non-market based) but in all cases should be new, additional, predictable and results-based. The financing entities are encouraged (for Green Climate Fund is requested) to apply the methodological guidance of the COP.

Regarding the requirements for receiving finance, and regardless of the source or type of financing, the activities funded should be consistent with the safeguards (explained in the safeguards section), should be fully measured, reported and verified in accordance with all relevant decisions of the COP (explain in the MRV section), and Parties have to elaborate and submit a national REDD+ strategy or action plan, a national FREL/FRL (or an interim subnational FREL/FRL), a national forest monitoring system (NFMS), a system for providing information on how the safeguards are being addressed and respected (SIS) and a Safeguard Summary.

The national focal points may nominate any domestic entities to receive results-based payments from financing entities, consistent with any specific operational modalities of the financing entities.

An information hub has been established to increase the transparency of information on results-based actions and payments. The hub includes assessed FRELs/FRLs, Safeguards summary, the national REDD+ strategy, information on the national forest monitoring system, as well as results from REDD+ activities expressed as tonnes of CO2 eq. and information regarding payments for those results.

8.5 The Ethiopian Process

8.5.1 Climate Change Impacts and National Climate Policies

8.5.1.1 Climate Change Impacts

Natural resources (water, soil, land, forests and biodiversity) are the foundations of Ethiopia's economic development, food security and livelihood sustenance. Ethiopia's heavy dependence on agriculture, coupled with a high population growth rate, makes the country particularly susceptible to the adverse effects of climate change. Agriculture, primarily rain-fed and highly sensitive to fluctuations forms the basis of the economy providing approximately 46 % of GDP and provides jobs for 80 % of the working population. Chronic food insecurity affects 10 % of the population and even in average rainfall years these households cannot meet their food needs, and rely partly on food assistance. Ethiopia's economy and social wellbeing are already exposed to climate variability and extremes and each face additional pressures through climate shocks and stresses.

There has been evidence of climate change in Ethiopia over the last 50 years. National level, temperatures have increased by an average of around 1 °C since the 1960s.Yearly variation around mean rainfall levels is 25 % and can increase to 50 % in some regions. There is evidence of a 20 % decrease in rainfall in the south central region of the country. The observed weather variability may lead to extreme events and hazards. There is a suggestion that the incidence of droughts and floods may have increased in the last 10 years relative to the decade before. Forest fires have become increasingly frequent in the last decade and an estimated 200,000 ha of forest are being affected. Historic weather variability, extreme events and hazards have resulted in a substantial negative impact on economic growth in agriculture. Floods and droughts have resulted in severe loss to agricultural crops and livestock resulting in food security implications. National estimates holds that although the economic impact depends on the extent of the variability and extreme events, droughts alone can reduce total GDP by 1 % to 4 %. Soil erosion has been estimated to reduce agricultural GDP by 2 % to 3 % (around 1 % of total GDP).

Several factors aggravate the impacts of climate change in a country like Ethiopia. The ever increasing population will result in land use changes which in effect results in the conversion of forested land to agricultural land. The high density of livestock combined with poor management will not only increase GHG emissions but also land degradation through overgrazing. The huge dependency on biomass fuel together with the increased demand for agricultural land will increase deforestation. On the other hand, the fact that Ethiopia's agriculture is primarily rain-fed implies the country's increased vulnerability to climate change impacts. On top of this, inadequate infrastructure will affect the country's resilience to climate change impacts while low level of awareness does also affect strong community participation and successful adaptation to climate change.

Future climate change in Ethiopia is uncertain, although scenarios of change show the range of possible outcomes. A range of projections indicate that a temperature increase between 0.5 °C to 2 °C by the 2050s relative to today. Moreover, current rainfall variability is expected to continue (projections of the change in future annual rainfall range from −25 % to +30 % by the 2050s). The future impacts and costs of climate change on agriculture are potentially very significant, which could put the country's ambition of reaching middle-income status by 2025 at risk. In this regard, in a hotter drier scenario, with increased incidences of droughts, the negative impact on GDP could be 10 % or more by 2050. In addition to economic impacts, agricultural livelihoods are also vulnerable to weather variability and stresses.

8.5.1.2 Climate Policies of Ethiopia

Recognizant of the country's vulnerability to climate change impacts, the Government of Ethiopia has been engaged in designing a national adaptive response. In the global stage, Ethiopia has been actively participating in international climate negotiations and recently led the African group on international climate forums. Apart from these engagements, the country has initiated and implemented different climate related policies over the last two decades. Ethiopia has ratified the UNFCCC in 1994 and the UNCCD in 1997 and submitted its initial national communications to the UNFCCC in 2001. Since then, a number of climate change related policies and institutional arrangements has been introduced in Ethiopia.

Ethiopia's commitment to a healthy environment is enshrined in the Constitution (Art 44) and the Environmental Policy of 1997 guide Ethiopia's response to Climate Change. The policy specifically covers soil husbandry; forestry; genetics; biodiversity; water resources; energy; minerals; human settlement; sanitation; industry and trade; culture and natural heritage and atmospheric pollution and climate change. Across these focus areas, the policy promotes a climate monitoring programme and recognizes the need for control measures for green house gases and use of renewable energy. It also calls for maximizing standing biomass and seeking financial support from industrialized countries.

Different national policies directly related to climate change have been formulated through the years and are under implementation. These include the National Adaptation Program of Action, NAPA, in 2007, Energy Policy & Bio-fuel Strategy in 2007, Nationally Appropriate Mitigation Action, NAMA, in 2010, Ethiopia's Program of Adaptation to Climate Change, EPACC, in 2011, which replaced the NAPA of 2007, and the Growth and Transformation Plan, (GTP, 2010/11) which was complemented by its Climate Resilient Green Economy Strategy (FDRE 2011a, b). Other national initiatives and sectoral programs include Agriculture and Rural Development Strategy (1994), Water Resources Management Policy, Health Policy, National Policy on Disaster Risk Management, Food Security Strategy, National Biodiversity Strategy and Action Plan (2005),

Adaptation to the negative impacts of climate change has been the principal focus in Ethiopia. Since Ethiopia's economy and the wellbeing of its people are closely linked to agriculture and the use of natural resources, adaptation and action towards climate resilience will come in part through focusing on improving performance and management in these areas in the future. Accordingly, the NAPA (2007) broadly focused on the areas of human and institutional capacity building, improving natural resource management, enhancing irrigation agriculture and water harvesting, strengthening early warning systems and awareness raising. EPACC which replaced the project-based and fund-constrained NAPA, has an overarching objective of contributing to the elimination of poverty and laying the foundation for a climate resilient path towards sustainable development (FDRE 2011b). To this end, it aims at mainstreaming climate change so that it is embedded within government policies and plans through Sectoral Climate Programmes and Action Plans. EPACC is a program of action to build a climate resilient economy through support for adaptation at sectoral, regional and community levels. EPACC identified adaptation strategies and priority options in the various socioeconomic sectors including improving agricultural productivity and livelihood diversification, crop and livestock insurance mechanisms, grain storage, renewable energy, gender equality, factoring disability, climate change adaptation education, capacity building, research and development, and enhancing institutional capacity.

Notwithstanding the threats, climate change also represents a huge opportunity for Ethiopia. The opportunity lies in the broader global agenda on climate change. Developing countries like Ethiopia stand to gain from both adaptation and carbon finance. In the succeeding years, with the objective of integrating both adaptation and mitigation activities, a Nationally Appropriate Mitigation Action (NAMA) was prepared in 2010. The NAMA outlined multi-sectoral projects aimed at reducing green house gas emissions (GHG) through the use of renewable energy resources, increasing soil carbon retention through the use of compost and implementation of agro-forestry practices for livelihood improvement and increased carbon sequestration. Forestry projects in the NAMA targeted at reducing deforestation and forest degradation and increasing carbon sequestration through reforestation of degraded area and sustainable management of existing forests. Unilateral/voluntary NAMAs under implementation include the massive watershed management programs in different parts of the country, sustainable land management projects (SLMP) in several regional states and afforestation/reforestation of degraded areas largely in the northern parts of the country.

Ethiopia was the first African country to join the Climate Neutral Network (CN-Net) initiative of UNEP that aims at mobilizing global support towards carbon free nations (Karen Ellis et al. 2009). In this line, the government initiated the Climate-Resilient Green Economy (FDRE 2011a) strategy, an initiative to mitigate the adverse effects of climate change across economic sectors and to build a green economy that will help realize the country's ambitious 5-year development plan, GTP I for the period 2010/11–2014/15 (FDRE 2010a, b). GTP I outlines the country's vision for a low-carbon development path while building a Climate-Resilient Green Economy (FDRE 2011a) strategy with the aim of reaching middle income status before 2025 (FDRE 2011a). The Strategy identifies eight sectors that play key roles in sustainable development; namely Forestry (Reducing Emissions from Deforestation and Forest Degradation; REDD+), soils, livestock, energy, buildings and cities, industry, transport and health. Owing to its huge abatement potential to reduce emissions not only from the forestry sector but also from other sectors, REDD+ is identified as a key pillar among the four initiatives selected for fast-track implementation (FDRE 2011a). The fact that these national climate change policies are multi-sectoral and with set targets indicate the government of Ethiopia's strong commitment towards improving the country's resilience to the impacts of climate change and a significant reduction in GHG emission across sectors.

8.5.2 Evolution of REDD+ in Ethiopia

8.5.2.1 The REDD+ Discourse and REDD+ Actors

The REDD+ discourse in Ethiopia went as back as 2006. A range of national and international actors shaped climate change policy and the REDD+ discourse in Ethiopia. National actors include government organizations (National Meteorological Agency; Environmental Protection Authority; Ministry of Agriculture; lately Ministry of Environment & Forest); Civil Society organizations and practitioners (Climate Change Forum-Ethiopia; Ethiopian Civil Society Network on Climate Change, Climate, Forum for Environment) and practitioners (FARM Africa/SOS Sahel).

As early as 2006, REDD+ was solely the engagement of two prominent NGOs (FARM Africa and SOS Sahel) in Ethiopia. These two NGOs are most actively involved with REDD+ in Ethiopia due to their many years of experience of Participatory Forest Management (PFM) in the Oromia Region of Ethiopia. In 2006, the Bale REDD+ project was initiated by FARM Africa/SOS Sahel and the Oromia Forest and Wildlife Enterprise (OFWE) with a financial support from Norway. The project area covers a total of 260,000 ha of forested area. The Bale REDD+ project is the first REDD+ project in Ethiopia registered under the Voluntary Carbon Standard (VM0015) and is now ready for validation. As REDD+ pioneer in the country, Farm Africa/SOS Sahel provided advisory services on development of National REDD+ during the R-PP development.

International actors were active in the REDD+ discourse largely through financial and technical support. Countries like Norway and UK were the principal donors of these REDD+ projects. Other actors directly or indirectly influenced the REDD+ discourse in Ethiopia include the World Bank, UNDP, GGGI and Academia.

8.5.2.2 The REDD+ Policy Process

REDD+ has evolved in Ethiopia under a policy framework that encourages land rehabilitation through reforestation/afforestation. This is reflected through the setting of national targets to increase forest cover as in PASDEP (2005/6–2009/10) (FDRE 2006) or in the provision of tax incentives for farmers who plant trees on their land as stipulated in the 2007 Forest Management, Development and Utilization Policy. In the later years, the introduction of the NAMA (2010) underscored the need to reduce GHG emissions through outlined multi-sectoral projects. Under the NAMA, forestry projects aimed at reducing deforestation and forest degradation and increasing carbon sequestration through reforestation of degraded area and sustainable management of existing forests.

In recent years, REDD+ policy seem to get embedded within a bigger Climate Resilient Green Economy (CRGE) Strategy which launched in late 2011 which go together with the Growth and Transformation Plan (GTP) 2010/11–2014/15. The GTP I outlines the country's vision for a low-carbon development path while building a CRGE Strategy with the aim of reaching middle income status before 2025 (FDRE 2011a). The Strategy identifies eight sectors that play key roles in sustainable development: forestry, soils, livestock, energy, buildings and cities, industry, transport and health. Owing to its huge potential to reduce emissions not only from the forestry sector but also from other sectors, REDD+ was selected as one of four initiatives for fast-track implementation (FDRE 2011a). The fact that these national climate change policies are multi-sectoral and have set targets indicate the government's strong commitment toward improving the country's resilience to the impacts of climate change and significantly reducing GHG emissions across sectors.

8.5.2.3 The Road Map (R-PP)

The Government of Ethiopia is interested in adopting REDD+ as a way towards sustainable forest development. Thus, Ethiopia submitted a REDD+ Program Idea Note(RPIN) in 2008 and developed a Readiness Preparation Proposal (R-PP), which was finalized in 2011 and then approved by the Forest Carbon Partnership Facility's participants Committee (FDRE 2011c). Implementation of REDD+ readiness was launched in January 2013.

Ethiopia's R-PP has key milestones which includes the readiness organization and consultation, development of a REDD+ strategy, as well as technical, policy, legal, management, Reference Emission level (REL)/Reference Level(RL) and monitoring arrangements necessary for full participation in the evolving REDD+ mechanism. The implementation of the R-PP covers the period of 2013–2016.

The following milestones and key activities of the R-PP are on the process of execution:

1. The establishment of the National REDD+ management arrangement (Steering Committee, Technical Working Group and three Task forces supporting the development of the national REDD+ Strategy, Safeguards, and MRV) is finalized.
2. Comprehensive and coherent REDD+ implementation strategy development and endorsement. Draft national REDD+ Strategy was prepared and consultation will be conducted on the strategy.

3. Consultation and participation and engagement of stakeholders in the REDD+ process is going on at different level. Effective communication mechanisms on all aspects pertinent to REDD+ is on the process of development and implementing consultation at different level.
4. In depth analysis of drivers of Deforestation and Forest degradation is progressing by selected international consultant firm.
5. Studies to identify options for benefit sharing related to the implementation of REDD+.
6. Preparing enabling institutional and legal environment for REDD+ implementation is one of the key activity in readiness phase and the study is now on the final stage. Under this study legal and institutional gaps for REDD+ implementation have been identified and necessary actions will be planned.
7. Preparing relevant safeguard instruments is on progress. The instrument will help to identify and propose mitigation measures for the potential environmental and social risks of the REDD+ strategy. For this four safeguard instruments are identified to be prepared. The formulation of the instruments is on the progress.
8. Designing MRV system and developing Reference Emission level. REDD+ MRV roadmap has been developed with the support of the MRV task force. Government of Ethiopia and Food and Agricultural Organization (FAO) have signed a Technical Assistance agreement for the implementation at the national level of a MRV system and for developing RELs. A REDD+ Compliant National Forest Inventory has been launched in March 2014, where data collection is going on at the moment.

8.5.3 Management Arrangement

Management arrangement to implement REDD+ readiness phase is started by establishing the REDD+ Secretariat at national level. The process of establishing the REDD+ Secretariat is progressed over 90 %. The REDD+ Secretariat is engaged in implementing the Readiness process on behalf of the MEF. The responsibility of the Secretariat is to coordinate facilitate and lead REDD+ process by engaging relevant stakeholders at different level of actions. At the national level there is a REDD+ Steering Committee (RSC) , REDD+ Technical Working Group (RTWG) and task forces. The State Minister of Forest Sector at the Ministry of Environment and Forest is chairing the National REDD+ Steering committee, a decision making body providing oversight for REDD+ Readiness. Members of the REDD+ Steering Committee are drawn from key REDD+ government institutions such as Ministry of Environment and Forest, Ministry of Agriculture, Ministry of Water, Irrigation and Energy, Ministry of Children, Women and Youth, Regional Forest Enterprises, Ethiopia Wild life Conservation, Representatives of Forest Cooperatives and Academic Institutions, and the Media. The Technical Working Group is providing technical guidance and quality assurance. The RTWG draws members from National and regional government institutions, academia, research institutes, civil societies and donors. Three task forces drawn from the technical working group focused on REDD+ Strategy, MRV system and Safeguards. The three task forces regularly meet and actively engage in providing technical support to REDD+ readiness.

The REDD+ management arrangement is gradually moving to embrace the regional states level REDD units. Regional Steering Committee and Regional Technical Working Group have been functional in Oromia Region, with representatives from the forest-dependent peoples and civil society organizations. Similar arrangements are being followed in other regional States, such REDD+ Coordination Units, playing the role of the REDD+ Secretariat at Regional Level are being organized in SNNRP, Tigray and Amhara regional states.

8.5.4 Consultation and Participation

The Cancun safeguards aim not only to mitigate the risk of adverse social and environmental impacts of REDD+ activities, but also to actively promote benefits beyond carbon emission reductions, such as respect for the rights of local communities, enhancing biodiversity, improving forest governance and empowering relevant stakeholders by ensuring their full and effective participation.

The process of consultation and participation is central part for effective implementation of REDD+ readiness. The national REDD+ Secretariat of the country is now developing Consultation and Participation Plan. This plan will provide a framework that ensures owner ship, transparency, and inclusiveness of effective and informed consultation and participation by relevant stakeholders in the process of implementing the R-PP .

To guide this complex and dynamic process at national level, Consultation and Participation Plan together with Awareness and Communications strategy and a Conflicts and Grievances Management Plan are being developed and shall be operationalized .

8.5.5 Readiness Financing

Ethiopia is participant of Forest Carbon Partnership Facility (FCPF). FCPF is managed by the World Bank as trustee. As a member of the FCPF participants committee in 2008, Ethiopia received an initial grant of US$200,000 to develop a Readiness preparation proposal (R-PP) in 2009. From 2009–2011 the Federal Environmental Protection Authority led the development of the R-PP, which was finalized in May 2011. The Federal Environmental Protection Authority has been restructured as the Ministry of Environment and Forest since June 2013. Implementation costs for Ethiopia's proposed Readiness Plan were estimated to be around US$13.6 million. The required finance for R-PP implementation has already been secured from a number of different donors, including the World Bank's Forest Carbon Partnership

Facility (FCPF), the Royal Norwegian Government and DfID (UK). The FCPF Readiness Fund approved a Readiness grant of US$3.6 million in 2012. Other large donors of REDD+ activities are Norwegian Government and the UK-DfID. The Government of Norway's International Climate and Forest Initiative (NICFI) and UK Department for International Development (DfID) each committed US$five million in Readiness Additional Finance required for the Readiness phase at the end of 2014, bringing total funding for Readiness to US$13.6 million. For Oromia Emission Reduction Program Design three million USD and for Emission Reductions 50 million USD additional resource was mobilized. The Norway Government also pledged eight million USD for performance in REDD+ readiness in 2013 and 2014.

8.5.6 Safeguard

It was established consensus in Cancun Mexico at COP 10 to promote and support a set of seven safeguards issues while executing REDD+ activities. The Cancun Agreements, and the subsequent Durban Agreement, also requested parties implementing REDD+ to provide information on how safeguards are being addressed and respected throughout the implementation of the REDD+ activities. Countries like Ethiopia that carry out REDD+ activities require to prepare country-level safeguard instruments to facilitate the implementation of the safeguards outlined in the Cancun Agreement intended to ensure social and environmental risks are minimized and benefits enhanced. National laws and policies already exist support or are consistent with most of the UNFCCC safeguards requirements, for example, the national environmental impact assessment proclamation.

As the FCPF is a World Bank initiative, countries participating in the FCPF like Ethiopia must complete a Strategic Environmental and Social Assessment to ensure coherence with the relevant World Bank safeguards.

Ethiopia is now developing four safeguard instruments to reduce the potential environmental and social risks and enhance the benefits of REDD+ implementation, expexted to be completed in August 2015. These instruments include Strategic Environmental and Social Assessment, Environmental and Social Management Framework, Resettlement Policy Framework and Process Framework. These safeguard documents will provide clear directions for managing and mitigating the environmental and social risks and impacts of future investments (projects, activities, and/or policies and regulations) associated with implementing a country's REDD+ strategy.

The SESA work facilitates the incorporation of environmental management and socio-economic decisions at the earliest stages of planning activities and investments. It will also provide avenues for the involvement of the public, communities/landowners, proponents, private interest groups and government offices in the assessment and review of any proposed interventions. In accordance with UNFCCC and the World Bank guidelines, special consideration will be given to livelihoods, rights, cultural heritage, gender, vulnerable groups, governance, capacity building and biodiversity.

ESMF supports an assessment of the risks and potential impacts connected with one or more projects or activities that may occur in the future. The Framework sets out the principles, guidelines, and procedures to assess environmental and social risks, and proposes measures to reduce, mitigate, and/or offset potential adverse environmental and social impacts and enhance both enhance environmental and social development benefits of REDD+ actions, policies and /or regulations.

The RPF is an instrument used to compensate or replace lost assets, livelihood, and income; and assistance for relocation, including provision of relocation sites with appropriate facilities and services. Moreover it helps restoration of livelihood to achieve at least the same level of well-being with the Project as without it.

The Process Framework (PF), establishes a process by which members of potentially affected communities can participate in planning of project components, determination of measures necessary to achieve the policy objectives, and implementation and monitoring of relevant project activities. It covers restrictions of access to legally designated forest conservation areas, which result in adverse impacts on livelihoods of the affected people.

The REDD+ Secretariat has two safeguards experts at national level to ensure the considerations of environmental and social issues in the whole REDD+ process and also to play a leading role in the development of safeguard instruments.

For guiding the national REDD+ safeguards process, the SESA and C&P Task Force was established from different relevant government and nongovernment offices and Academia and Research.

8.5.7 Forest Reference Emission Levels and/or Forest Reference Levels (FRELS/FRLS) and M&MRV Systems

Among the agreements achieved in Cancun (16th Conference of the Parties, 2010), it is requested that developing countries aiming to undertake the REDD+ activities, to develop, in accordance with national circumstances and respective capabilities, a robust and transparent national forest monitoring system for the monitoring and reporting REDD+ activities.

The monitoring function of the National Forest Monitoring System allow the country to assess a broad range of forest information, including in the context of REDD+, and the MRV function for REDD+ the estimation and international reporting of forest emissions and removals.

Responding to this Ethiopia is now designing and implementing robust and accurate system for monitoring and measuring carbon emissions and removals to enable the country to report and verify actions on deforestation and forest degradation and other activities aiming to conserve, sustainably manage and increase forest carbon stocks.

The Terms of Reference for Developing Capacities for a national Measuring, Monitoring, Reporting and Verification System to support REDD+ participation of

Ethiopia, including the MRV Roadmap, were prepared by the Ministry of Agriculture and the Environmental Protection Agency, with support from the Norwegian embassy and Wageningen University in June 2013.

Several national and international institutions were identified in this study to be part of the MRV Task Force, a body of counselling, advice and technical participation jointly with the Ministry of Environment and Forest to implement the MRV project in a real participated process.

For the implementation of the MRV Roadmap, the Ministry of Environment and Forest on behalf of the government of Ethiopia and the Food and Agriculture Organization of the United Nations signed an agreement in August 2014 for the provision of technical assistance for the implementation of a national forest monitoring and MRV system for REDD+ Readiness in Ethiopia. This is a major project, but not the only, related to M&MRV System implementation at national and regional level.

The MRV system will be, as required, a robust and transparent national forest monitoring system for the monitoring and reporting REDD+ activities, providing estimates that must be transparent, consistent and accurate (taking into account national capabilities and capacities). It will use a combination of remote sensing and ground-based forest carbon inventory – National Forest Inventory - approaches for estimating anthropogenic forest-related greenhouse gas emissions by sources and removals by sinks, forest carbon stocks and forest area changes. The National Forest Inventory is ongoing and is expected to be finalized in 2015.

The MRV system is being designed to be able to accommodate multiple stakeholders. At the national level, a coordination mechanism will be put in place to provide a link between policy and practice at different scales. Additionally MRV-related activities and arrangements will be linked to existing relevant structures, including higher education and research institutions and ongoing monitoring activities at the local level. The implementation of MRV is coordinated by the REDD+ Secretariat, with support from the REDD Technical Working Group and a number of potential national (MRV Task Force Members) and international partners (WB, FAO).

As the REDD+ scheme in Ethiopia is expected to deliver emission reductions and other co-benefits, the MRV system will be designed to help track a range of other indicators such as biodiversity and social benefits.

The national MRV system will consider the development of innovative participatory approaches aimed at engaging forest-dependent communities in monitoring and verification work build understanding and local ownership. In this regard, a PMRV pilot project is being designed jointly with the involvement and support of the MRV and Safeguards components of REDD+ Secretariat, CIFOR and FAO.

Regarding the development of a FRL/FREL in Ethiopia, the basic elements that have been defined or are being discussed at the moment are:

1. National Forest Definition: the final proposal is: ‘Land spanning more than 0.5 ha covered by trees (including bamboo) (with a minimum width of 20 m over not more than two-thirds of its length) attaining a height of more than 2 m and a canopy cover of more than 20 % or trees with the potential to reach these thresholds in situ in due course’.

2. Scale: National and regional (jurisdictional) and project-level. Finally multi-scale nested project-level activities are integrated into an accounting and incentive scheme of a larger jurisdiction (regional and national). Incentives flow directly to regional governments and project developers in addition to national government. Oromia regional state presents an advanced stage in the implementation of REDD+, with the development of a first draft version of FREL/FRL.
3. Scope: Ethiopia is proposing a step-wise approach for the inclusion of the following REDD+ activities: (i) reducing emissions from deforestation, (ii) reducing emissions from forest degradation, and (iii) enhancement of forest carbon stocks (A/R). Using the requirements from FCPF Carbon Fund the MRV component will monitor as targeted activities; deforestation, forest degradation (if emissions are greater than 10 % of total) and A/R activities.
4. Approach: The establishment of FREL/FRL will involve a nested approach, using both sub-national and national baseline data. It is also being considered a stratified approach, calculating a FREL/FRL by vegetation type, homogenizing so drivers of deforestation and forest degradation. It will be established at least two alternative baseline scenarios; a historical annual average over a 14 year period and a historical trend based on changes over 14 years. The number of points to calculate the historical annual average and the historical trend model have not yet been decided (at least three in the period 2000–2013). The end date for the historical analysis is the most recent date prior to 2013 for which forest cover data is available. At the moment the national LULC map (FAO/MEF) is being completed using Landsat mosaics from 2013.
5. Activity Data: A proposal for statistical sampling analysis will be preferred to a ‘wall to wall’ analysis to calculate the activity data (it should be determined at least twice in the 5 year crediting period, but actually It will be used a spatially explicit tracking of land-use conversions over time). The uncertainties related to activity data will be quantified.
6. Emission Factors: to calculate the emission factors corresponding to changes in LULC all significant pools (representing more than 10 % of total) will be measured. IPCC Tier 2 will be used to establish it (country specific data for key factors), but in the future the national inventory of key C stocks (IPCC Tier 3), considering repeated measurements of key stocks through time and modelling, will be used improving the accuracy. The uncertainties related to emission factors will be also quantified (based on this level of uncertainty – AD and EF - a prescribed amount of emission reductions will be placed in a buffer reserve).

In all cases there will be consistency with UNFCCC submissions of national GHG inventory and with the methodological IPCC guidance.

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Part IV
Economic and Management Impacts

Chapter 9
Influence of Carbon Sequestration in an Optimal Set of Coppice Rotations for Eucalyptus Plantations

Luis Diaz-Balteiro and Luiz C.E. Rodríguez

Abstract The coppice regeneration method used to manage eucalypts leads to a simultaneous optimization problem: the manager has to simultaneously define the optimal age in each coppice rotation and the optimal number of coppice rotations for each plantation full cycle. The dynamic nature of the problem justifies the use of methods like dynamic programming in order to achieve optimal solutions. Expected land value and the duration of the optimal rotation may change significantly when payments for carbon sequestration are added as revenues in the cash flow analysis of the project. In this chapter, we analyze the effects of considering carbon sequestration as a subsidized complementary product when defining the optimal set of coppice rotations. A Monte Carlo simulation technique was used to model the inherent risk of some variables and parameters like pulpwood price, carbon price, and discount rate. The variation in the land expectation value and in the optimal rotation length is reported when these stochastic variables are computed. Two study cases are shown, one with Eucalyptus urophylla S.T. Blake in Brazil, and another with *Eucalyptus globulus* Labill in Galicia, Spain.

9.1 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 and Lyon 1985; Tait 1986). The determination of optimal forest rotation ages when considering both timber production and

L. Diaz-Balteiro (✉)
Department of Forest Economics and Management. ETS Ingenieros de Montes, Ciudad Universitaria, Madrid, Spain
e-mail: luis.diaz.balteiro@upm.es

L.C.E. Rodríguez
Escola Superior de Agricultura "Luiz de Queiroz" University of São Paulo, SP, 13418-900, Piracicaba, Brazil
e-mail: lcer@usp.br

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carbon uptake is a problem that has been addressed in forest literature since the 90s. Several authors have used some variants of the Faustmann formula to devise methods in order to incorporate the benefits of capturing CO_2.

9.1.1 Optimal Coppice Rotation Definition

Medema and 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 and 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) 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. Besides, Langholtz et al. (2005) studied the optimal management of a coppicing species (*Eucalyptus grandis* Hill ex Maiden) for dendroremediation purposes. Finally, Ferreira et al. (2012) determined the coppice cycles and the rotation length that maximizes expected net revenues in *Eucalyptus globulus* Labill plantations in Portugal when the risk of wildfire was taken into account.

9.1.2 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) in a forest level analysis to create a production possibility frontier. The marginal cost, measured as the change in the net present value, was used to rank alternative projects and the concept was used in a case study where forest biomass growth and decay was 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.

In a seminal research paper, 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/metric ton), 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 was 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 evaluated the changes in rotation ages when social optima and efficient allocation of resources are directed by public financial aid, namely, subsidies to encourage tree growth and taxes to discourage short rotations.

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. Surveys covering these studies can be found in Richards and 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 and 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 took 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 and Alavalapati (2005).

The economic aspects of considering carbon sequestration in Eucalyptus plantations when finding the optimal number of multiple rotations have been analysed frequently, although ignoring coppicing. Cacho et al. (2003), for example, tested how the optimal rotation of *Eucalyptus nitens* (Deane & Maiden) Maiden plantations vary when carbon credits are included (tonne-year accounting, ex-ante full crediting and ex-post full crediting). Other authors (e.g., Whittock et al. 2004a, b) considered genetic gains as a mechanism for increasing carbon sequestration in *Eucalyptus globulus* plantations.

In this chapter, the optimal coppice management regime considering carbon sequestration was determined for two fast growing plantation cases in Brazil and Spain using a dynamic programming approach. This technique has been used in forest management for several decades. The first applications focused on defining the optimal scheduling of silvicultural treatments in even-aged stands (Amidon and Akin 1968; Schreuder 1971), including the definition of intermediate treatments and optimal forest rotations (Borges and Falcão 1999; Brodie et al. 1978; Filius and Dul 1992; Haight et al. 1985; Torres-Rojo and Brodie 1990). This procedure has also been used to deal with the problem of harvesting adjacent forest stands (Hoganson and Borges 1998; Borges and Hoganson 1999). Further, Cacho et al. (2003) applied a dynamic programming algorithm in an agroforestry system to determine the optimal area covered with trees and the optimal forest rotation lengths used to provide carbon sequestration services. More recently, Ferreira et al. (2012) developed a stochastic dynamic programming approach in order to evaluate forest management decisions in *Eucalyptus globulus* Labill plantations under different wildfire occurrences and post-fire mortality scenarios.

The analysis in this paper covers two scenarios (deterministic and non-deterministic) and three basic situations (cases) for each scenario and country. The Monte Carlo methodology was employed to deal with the uncertainty associated with three parameters, namely the discount rate and the prices for pulpwood and carbon. The three basic situations are: (i) no payment for carbon sequestration, (ii) payment for carbon sequestration services including the assumption that all sequestered carbon is 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 50 years after harvest. The two scenarios were explained with more details in Diaz-Balteiro and Rodriguez (2006) and Rodriguez and Diaz-Balteiro (2006).

9.2 Materials and Methods

9.2.1 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. Clearcuts and replanting can be accomplished in one or less than one year. Carbon emissions during the first two 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, were 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.

9.2.1.1 Brazil

In Brazil, approximately 5.1 million hectares of eucalypts are intensively managed for wood production and represent 77 % of the total area of planted forests (ABRAF 2013). Among other uses, Brazilian Eucalyptus plantations are the main source of fibre for the pulp industry, which has an installed production capacity of

Table 9.1 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 clearcut age, and k years after clearcut

13.9 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 48.6 % 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 and 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 six to seven 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 clearcutting, inappropriate choice of species/ provenance for given soil and climate conditions, and harvest technology.

Table 9.1 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: (i) highly productive 6-year old *Eucalyptus urophylla* plantations, with an average number of 1660 trees per hectare and a yield of 530m^3/ha at harvest (site index I, with an average height of 34 m at 6 years old); and (ii) less productive plantations with the same number of trees per hectare but a yield at harvest of 179m^3/ha (site index II, with an average height of 24 m at 6 years old). Volume estimates for the coppicing rotations were simply

Table 9.2 Management costs for *Eucalyptus globulus* plantations in Galicia

Year	Silvicultural treatment	Cost (€/ha)
0	Site preparation and plantation costs	1297
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 clearcut age, and k years after clearcut

based on the first rotation previous production level and reduced by a fixed rate (Gonçalves et al. 1997). For the Brazilian plantations, the basic wood density was considered to vary between 0.484 and 0.793 (Scanavaca Junior and Garcia 2004), with an average 0.655 t/m^3.

9.2.1.2 Galicia

In Galicia (Northwest Spain), *Eucalyptus globulus* stands constitute the largest group among all other forest plantations. Eucalyptus plantations cover more than 600,000 ha and produce over 5.5 million cubic meters of roundwood annually, which account for 39 % of timber harvests in Spain (MAGRAMA 2014). These plantations produce mainly pulp for paper industries: 85 % of the eucalypts production is pulpwood, and a large percentage of the land occupied by 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 insect or pest controls, 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 clearcut 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 are shown in Table 9.2.

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).

In order to calculate the carbon content of each species, a 0.25 conversion factor (wood density x proportion of carbon in pulpwood) was used for the Spanish plantations. In the Brazilian plantations, this figure is 0.3275. Only the aboveground

Table 9.3 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 clearcuts allowed	1–4	1–4
Minimum and maximum rotation ages (years)	5–9	13–18
Possible dynamic programming states	780	1554
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

carbon (stem and aboveground biomass carbon) was computed. To compute carbon associated with aboveground biomass, first the stem volume was estimated and then this value was multiplied by a biomass expansion factor. 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 five years that follows clearcutting (Bateman and 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 fifty years after clear-cut (van Kooten et al. 1999). Neither recycling of paper products nor changes in carbon content due to fertilization was considered. A reference price of 10€ for each metric ton of carbon captured was used in the analysis of the deterministic scenario 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 9.3 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€ per metric ton for logs from trees with a minimum diameter greater than 14 cm, and 14€ per metric ton 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 were derived from the series of values taken at the beginning of each of the last 33 months stored in the Point Carbon database.[1]

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.

9.2.2 Methods

To determine the optimal rotation we utilized first 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 clearcutting), the financial optimal rotation is the age at which the land expected value (LEV) reaches a maximum. LEV is calculated as follows:

$$
\begin{aligned}
&LEV = \frac{\sum_{\forall z} NPV_z - \sum_{\forall z} RE_z}{1 - e^{-iT}} \\
&\text{with:} \\
&z \in T \\
&T = \sum t_k
\end{aligned}
\tag{9.1}
$$

where t_k is the harvest age at each rotation z, NPV_z is the net present value for each clearcut of trees grown from seedlings or sprout shoots, and revenues are generated by selling pulpwood and carbon credits (see Eq. 9.2). The number of clearcutting events 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) than 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

[1] http://financial.thomsonreuters.com/en/products/infrastructure/financial-data-feeds/point-connect-energy-market.html

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. 9.1:

$$\begin{aligned} NPV_z &= I\left(t_k\right)\cdot e^{-i\cdot t_k} - K - G\cdot\frac{e^{-i\cdot 1}\cdot\left(e^{-i\cdot t_k}-1\right)}{\left(e^{-i\cdot 1}-1\right)} - \sum_{t_j\in t_k} Y_j\cdot e^{-i\cdot t_j} - \sum_{t_j=1}^{t_j=t_k} I_C\left(t_j\right)\cdot e^{-i\cdot t_j} \\ &\text{with:} \\ I_C\left(t_j\right) &= Pc\cdot \Delta Vb_j \\ \Delta Vb_j &= \rho\cdot Cc\cdot f\cdot \Delta V\left(t_j\right) \end{aligned} \tag{9.2}$$

where $I(t_k)$ corresponds to total income obtained in year t_k as the result of the wood harvested in each clearcut (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 costs 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 biomass factor expansion (f). The discount rate is represented by i and is applied for timber and carbon benefits. Finally, modifications in some terms of the Eq. (9.2) were introduced depending on the growth period (seedling rotation or coppicing rotation).

$$\begin{aligned} RE_z &= \sum_{l=1}^{l=50} RE_l\cdot e^{-i\cdot\left(t_k+l\right)} \\ &\text{with:} \\ RE_l &= Pc\cdot\sum p\cdot BCr \end{aligned} \tag{9.3}$$

The carbon released at harvesting is calculated (Eq. 9.3) 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 fifty years after harvest. Finally, RE_z is the present value of annual carbon released and discounted in the future.

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 clearcut is executed, a choice emerges, independently of the stand age: replant again, or let the sprouts 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 were defined (clearcutting 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 to generate a new rotation. Obviously, this model can be extended to other situations with larger numbers of states, stages and decisions.

According to Hillier and 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:

$$LEV_z^* = \max\left(LEV_z' + \left(LEV_{z+1}^*\right)\cdot e^{(-i\cdot t_k)}\right) \tag{9.4}$$

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, a clearcut was 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 1554 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 were: 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 some empirical data in Galicia (González-Río et al. 1997), for the Spanish plantations 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 equalled the first clearcut volume; and in the third coppice rotation, a drop of 25 % was found when compared to the first production level.

In order to deal with risk and uncertainty, the Monte Carlo method is frequently employed. This methodology has been used in several forest management cases. Carlsson et al. (1998) applied the Monte Carlo simulation method to analyse 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 and Carlsson 2000), and to compute the net present value in some projects (Klemperer 2001). On those lines, Knoke et al. (2002) 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) 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 was used to develop the Monte Carlo simulations.

Two scenarios, deterministic and non-deterministic, and three different cases were established for both plantations. In the first case, carbon sequestration was not included; in the second case, carbon sequestration was included, following IPCC guidelines (Penman et al. 2003), with all carbon in harvested biomass oxidized at the moment of clearcutting; and in the third case, carbon sequestration was also considered, but a percentage of carbon was retained as stored carbon 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.

9.3 Results and Discussion

Table 9.4 shows the optimal coppice regimes considering the deterministic scenario. Initially, we will comment on the results for the Eucalyptus plantations in Spain. When carbon sequestration was not rewarded, the same coppice management regime was optimal regardless of the site index level. Moreover, the optimal regimes had the same rotations. The result of rewarding carbon sequestration can

Table 9.4 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

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 9.4), with three identical six-year rotations. For site index II (lower productivity site index) it is 21 years, with three seven-year rotations. These plantations showed 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 was more responsive to site quality. Thus, a coppice cycle with two rotations is recommended for the higher productivity site in cases 2 and 3, while a regime with three rotations is optimal in the three cases considered for the lower productivity site.

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. 9.1). Conversely, LEVs increased for site index III by 5.7 % and 7.2 %, for cases 2 and 3, 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.

Table 9.5 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 one year in most of the analysed cases. The Brazilian analysis

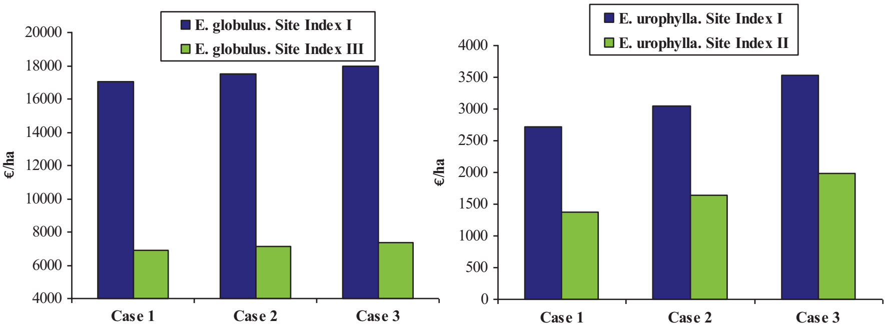

Fig. 9.1 LEV for both plantations. Deterministic scenario

Table 9.5 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

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 were two or three years shorter depending on to site index. In case 3, the optimal cycle was even shorter.

In contrast, LEVs from Spanish plantations were 20 to 30 % less than the values obtained for the deterministic scenario (Fig. 9.2). 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 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 9.4). However, the Brazilian plantations

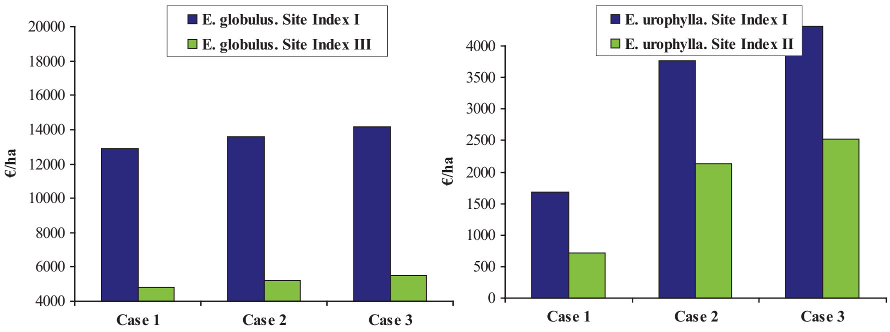

Fig. 9.2 LEV for both plantations. Non-deterministic scenario

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 regarding the duration of the rotations coincide with the results obtained for *Eucalyptus globulus* in Portugal (Cunha-e-Sá and Rosa 2004). Other studies for non-eucalyptus species (van Kooten et al. 1995) established the same trend. One exception was shown by (Stainback and Alavalapati 2005), where increases in management costs to promote carbon sequestration decreased the optimal rotation length.

When randomness is introduced in the analysis (Table 9.5), results were 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 (Rodriguez and Diaz-Balteiro 2006) if a sensitivity analysis for variations of some variables would be applied in the deterministic scenario.

If only based on Medema and 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 carbon payments do not exist. Thus, the lengthening of the rotations (not the optimal coppice cycle) when carbon captured is considered in the analysis with positive prices for permanent credits commonly occurred in numerous studies using other fast-growing species (Olschewski and Benitez 2010).

It is relevant to point out that the methodology used can easily incorporate different states reflecting changes on genetic material and/or silvilcultural 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 allows for including hypothetical scenarios with possible subsidized credit schemes as in Huu-Dung and Yeo-Chang (2012). Future work would be needed to incorporate more accurate yield models to predict production under multi-coppicing rotations, particularly in Spain. With these new models it would possible to justify the adoption of diverse silvicultural regimes that would increase carbon sequestration as in the study of Bussoni Guitart and Estraviz Rodriguez (2010). Finally, this analysis could be extended to include other services besides carbon capture in the evaluation of the optimal rotation in these plantations. For example, in a recent study Nghiem (2014) included a biodiversity conservation objective studying the optimal rotation in *Eucalyptus urophylla* planted forests in Vietnam. In this case, the optimal rotation was longer than without biodiversity component.

9.4 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 showed different results regarding the length of the optimal coppice cycle and the land expectation value. The optimal total cycle length, though, remained 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.

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Chapter 10
Use of Forests and Wood Products to Mitigate Climate Change

L. Valsta, B. Lippke, J. Perez-Garcia, K. Pingoud, J. Pohjola, and B. Solberg

10.1 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 650–1200 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?

L. Valsta (✉) • J. Pohjola
Department of Forest Economics, University of Helsinki, Helsinki, Finland
e-mail: lauri.valsta@helsinki.fi

B. Lippke • J. Perez-Garcia
College of Forest Resources, University of Washington, Seattle, WA, USA

K. Pingoud
VTT Technical Research Centre of Finland, Espoo, Finland

B. Solberg
Department of Ecology and Natural Resource Management, Norwegian University of Life Sciences, Oslo, Norway

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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.

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. 10.1. 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.

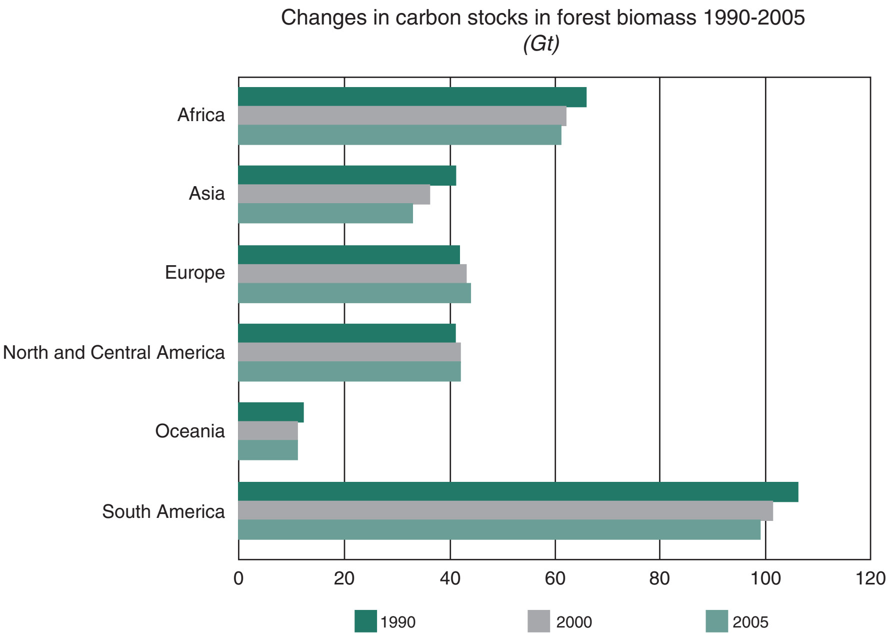

Fig. 10.1 Changes in regional forest biomass (FAO 2006)

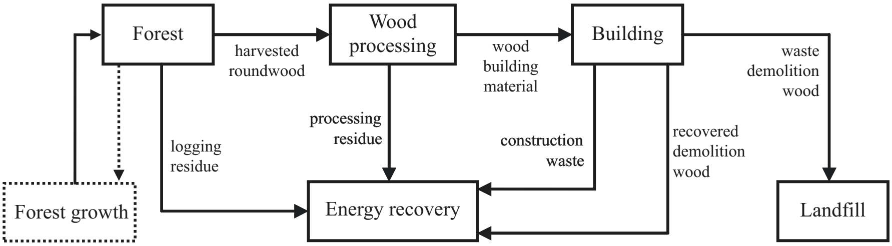

Fig. 10.2 Carbon flows in an integrated analysis covering forest dynamics and the wood product life cycle (Modified from Gustavsson et al. 2006)

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 roundwood and 40% fuelwood. However, FAOSTAT statistics (2006) report a 500 million cubic meters higher fuelwood use and the resulting total roundwood removals became 3490 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. 10.2). Additionally, this relationship is non-linear 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.

10.2 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.

10.2.1 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. 10.3). 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. 10.4).

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

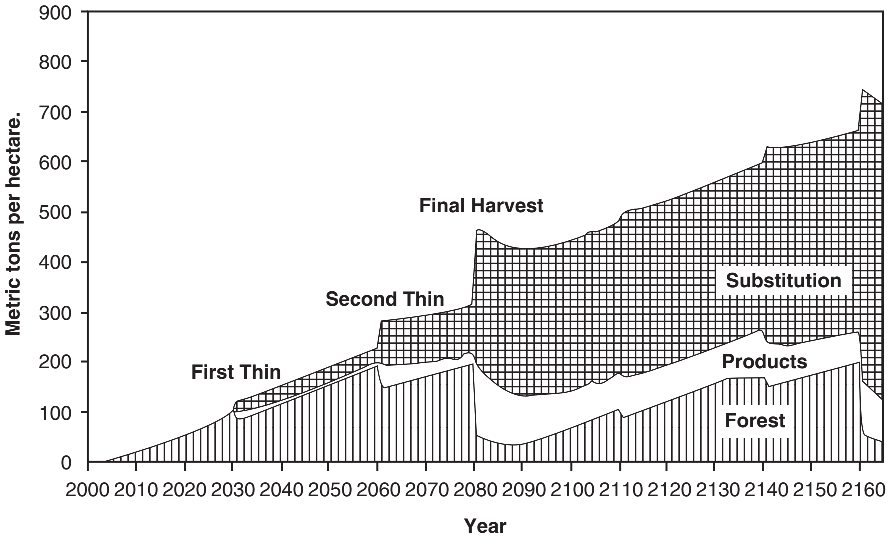

Fig. 10.3 Carbon in the forest and product pools with concrete substitution for the 80-year rotation (From Lippke et al. 2004)

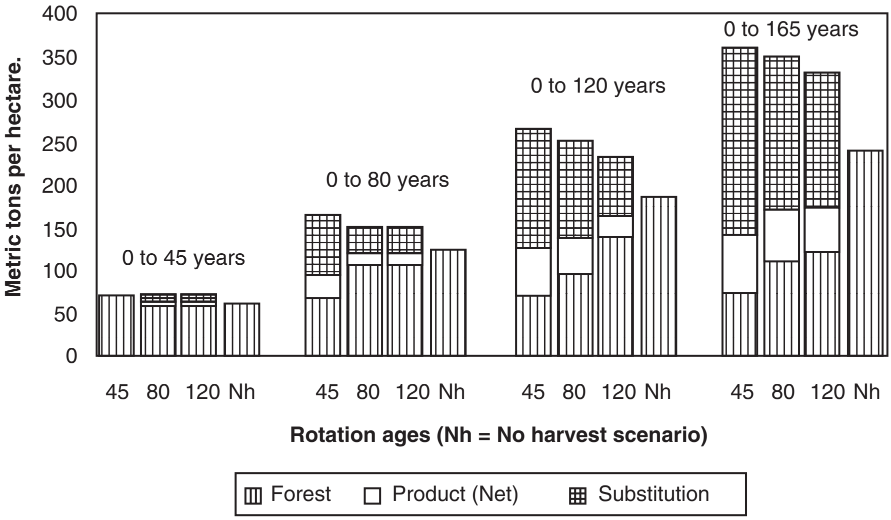

Fig. 10.4 Average carbon per year in forest, product, and concrete substitution pools for different rotations and simulation times

houses, respectively (ratio 1.26) (Lippke et al. 2004). These amounts can be made relative to the carbon in wood employed in houses (6496 and 3298 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.

10.2.2 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 emissionfree 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. 10.5). 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

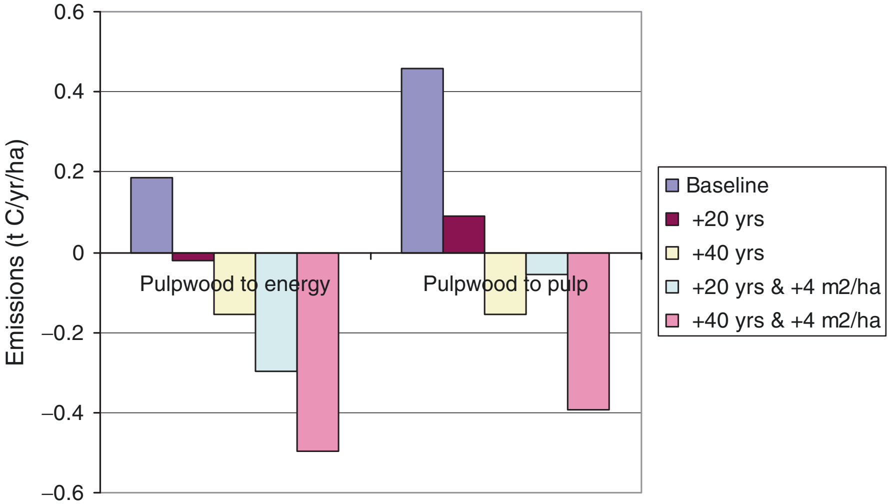

Fig. 10.5 Total emissions from wood utilization based on silvicultural regimes for Norway spruce, Swedish building

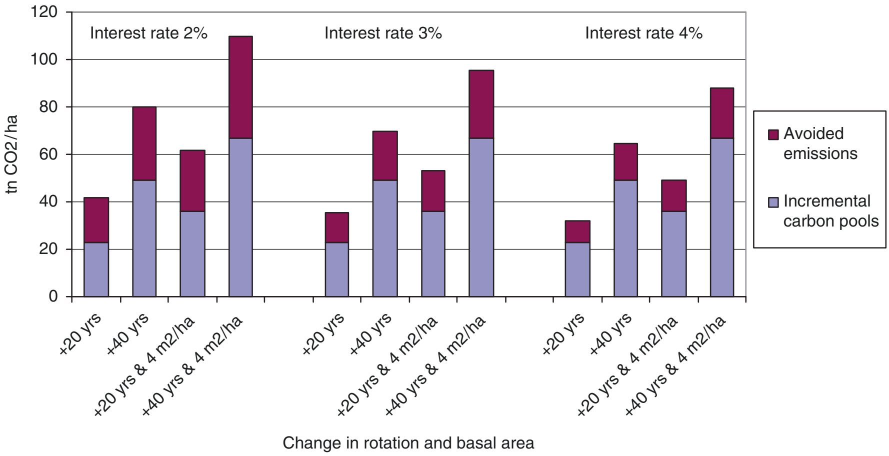

Fig. 10.6 Changes in the present value of discounted emissions and steady state carbon pools due to changes in rotation and stand basal area before and after thinning, Norway spruce

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 and Solberg 1994). Figure 10.6 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.

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.

10.2.3 GAYA-J/C Model (Petersen et al. 2004, 2005)

While the two previous studies were based on stand-level and conceptual fullyregulated 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 2207 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 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. 10.7). 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 1900%). Forest management activities were somewhat intensified, for example planting area was increased. A larger share of wood was harvested as sawtimber.

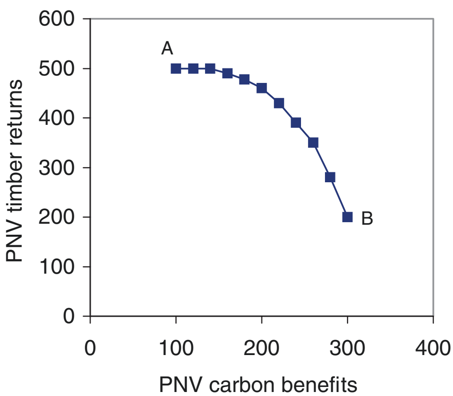

Fig. 10.7 Trade-off between timber returns and carbon benefits

10.3 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.

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Chapter 11
Biomass Forest in Sweden and Carbon Emissions Balance

Tord Johansson

11.1 Introduction

Biofuels have 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 biofuels for fossil fuels; and replacing energy-intensive materials like cement, steel and plastics with wood products (Schlamadinger and Marland 1996; Ericsson 2003). In a study analyzing the relation between forest management and the carbon balance of the forest sector, Lundmark et al. (2014) concluded that about 475 kg of CO^2 emissions are avoided for each cubic meter of biomass harvested, after accounting for carbon stock changes, substitution effects and all emissions related to forest management and industrial processes. In a long-term experiment of Norway spruce (*Picea abies* (L.) Karst.) stands in northern Denmark, Skovsgaard et al. (2006) found that total C stocks of biomass compartments decreased with increasing thinning intensity. Further, increased forest biomass production can reduce net CO_2 emissions by an additional 40 million tons per year. In the Kyoto Protocol, forestry and forest management can be including in the accounting of greenhouse gases reduction programs (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 result in a change of the carbon stocks can be included in carbon accounting. Besides anthropogenic CO_2 emissions originating from the combustion of fossil fuels, land-use changes could also have a considerable impact on the amount of greenhouse gases in the atmosphere. Using wood biomass for biofuels reduces the forest stock causing net reduction in the amount of

T. Johansson (✉)
Department of Energy and Technology, Swedish University of Agricultural Sciences, 7061, Uppsala SE 750 07, Sweden
e-mail: tord.johansson@slu.se

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carbon stored (Kirschbaum 2003a; Cherubini et al. 2011). While land-use changes can cause increased emissions, these could be decreased by increasing the forest area through decreased harvests and through afforestation. For example, abandoned farmland could be used for establishing new forest stands.

11.1.1 Biofuel Use in Sweden

For centuries, trees have been used in households as firewood or for charcoal production.

Conventional forest management combined with biofuel utilization has been practiced for the last 50 years in Sweden. At present, biofuel harvesting is mainly conducted on large forest land areas for economic reasons, where tops and branches from clearcutting are recovered as fuelwood. Further, special equipment for fuelwood harvests is being used and there is an existing market for energy production from biofuels using district heating plants. The infrastructure is well-established and has resulted in an increasing market for energy originating from biofuels. The majority of harvested fuelwood is distributed to the district heating plants close to cities. However, harvested fuelwood are also used as firewood, wood chips and pellets for household use.

The total Swedish energy supply in 2013 was 564 TWh (Anon. 2015). Biofuels provided 23 % of that total (Fig. 11.1). Although the majority of energy systems in Sweden is based on oil and electric power, there has been a dramatic increase in the use of biofuels during the last decade. Using only tops and branches recovered from clearcut areas provides a supply of 14 TWh. However, by using small trees from stand cleaning and thinning operations or by using stumps on clearcut areas, the total fuelwood production could increase by 5 and 5–10 TWh per year, respectively. Further, using more efficient harvesting methods could increase the production by 50 % or 5–10 TWh per year.

Stumps are a promising potential fuelwood source. Among the two most frequent tree species in Sweden, Norway spruce and Scots pine (*Pinus sylvestris* L.), Norway spruce is the most useable species for stump removal. Forest companies are currently working on the most effective ways to harvest the stumps with or without roots. Also, the ecological effects of stump harvesting are being studied, along with ways to store the stumps prior to use. In the 1970s, the harvesting of stumps harvests was researched, but this research was focused using stumps as raw materials for the pulp industry. Experiences and research from that period can be used today in the efforts to harvest stumps for biofuels. For example, when tops, branches and stumps are removed, the pH in the soil will decline and soil nutrients may be depleted. One way to ameliorate these losses is to recover wood ashes from heating plants and spread this over harvest areas.

Authorities have prepared rules and recommendations regarding when tops and the branches on forest land can be harvested to minimize nutrient losses. One controversial issue is afforesting existing farmlands, in that farmers and the general public favor maintaining an open landscape. On the other hand, there is an aware-

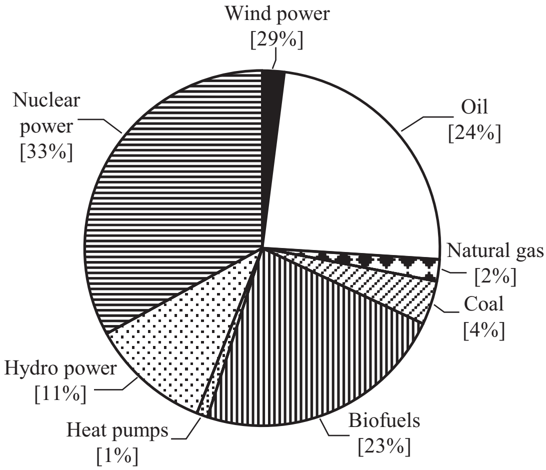


Fig. 11.1 Total energy supply in Sweden in 2013 (Anon 2015)

ness of the need to decrease the use of fossil fuels, and pressure to phase out nuclear power. As a result, biomass utilization for biofuels may be a more ecologically-acceptable way to solve the energy supply problem.

Short rotation plantations of *Salix* on former farmlands have been practiced for the last 30 years (Rytter 1996). However, the annual planted area of 12,000–15,000 ha has not increased during the last 20 years. Currently, the number of plantations of hybrid aspen (*Populus tremula* L. x *Populus tremuloides* Michx.) and other hybrid poplars (*Populus maximowiczii* x *P. trichocarpa* among others) has increased. These species are fast growing and might be managed for fuelwood, pulpwood, and/or timber. Depending on the price of cereals and the political view of the importance of bioenergy, a farmer might choose to plant fast-growing tree species instead. Further, some politicians regard the use of short rotation plantations (*Salix* or other fast growing broad-leaved species) on large areas of farmland as one way to balance CO_2 emissions.

11.1.2 Forest Management May Affect the Amount of Carbon Emissions

Forest and forest management are important factors 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. 2001; Myeni et al. 2001). Some forestry activities that can affect the amount CO_2 in the atmosphere is (Schlamadinger and Marland 1996; Cherubini et al. 2011; Kirschbaum 2003a):

- 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
- Net reduction of carbon captured in forest by use of wood for biofuels

The amount of carbon stored in wood products has been studied sparsely 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. A target in Swedish forest policy has been to maintain a long-term sustainable production of timber with an even aged-class distribution on regional and national level. The selection of rotation length is considered to be an important and an effective strategy to affect the carbon stocks in the forest (Liski et al. 2001; Harmon and 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., 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 can store more carbon in the biomass than the managed forest (Cooper 1982; Thornley and Cannell 2000; Maclaren 2000; Kirschbaum 2003b). If the purpose is to store as much carbon as possible in the forest there should be no thinning operations. However the length of the rotation period is depending on the efforts to optimize forest production.

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 prolonged 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).

Today, most of the biofuel harvested in Sweden consists of logging residues on forest land 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 depending on fewer amounts of nutrients when the residues are removed (Jacobson et al. 2000).

Afforesting broadleaved trees on former farmland could also be one way to increase the amount of biofuel that can replace fossil fuel. A suitable strategy could be to use trees, growing on abandoned farmland, for biofuel as it gives considerable amounts of biofuel (Mitchell et al. 1981; Johansson 2000a). The distance to the power plants are often short (Johansson 1999a). The amount of biofuel that can substitute fossil fuel is affected by the length of 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 2003b). 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 farmland for a long rotation period but without any thinning operations (Johansson 1999b, 2000a). 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). In a review by Rytter et al. (2011) both short and long rotation periods of planted hybrid poplar and hybrid aspen was studied. Depending on number of planted trees per hectare and rotation length the biomass production of hybrid poplar (3–15 years rotation) ranged between 25 and 170. When managing hybrid aspen a combined production of timber and biomass is the normal management. Besides the timber production 70 tons ha^{-1} will be produced during 25 year.

11.2 Biomass Production

An overview of the biomass production for different tree species is given by Canell (1982).

Below three models for biomass production is described. The chosen models are based on results from earlier studies. In practice these models are easy to use, especially on fertile soils with broadleaf species growing in dense stands.

11.2.1 Ingrowths on Former Farmland

A lot of areas are not cultivated in any way when farmland areas are laid down as a result of too high agricultural production combined with the lack of young people interested in farming but also of biological, technical and economic reasons such as low fertility, small areas with an impractical layout. According to Johansson (1999c), 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 1999d, e, 2000b). 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 16 and 90 years old (Johansson 1999a, 2002). Some characteristics for the studied species are given in Table 11.1.

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 15–20 years of age or most of the broadleaves are removed, Fig. 11.2. Then a new generation of trees will be established by sprouts and suckers (Børset and Langhammer 1966; Berry and Stiell 1978; Pastor and Bockheim 1981; Meeuwissen and Rottner 1984; Alban and Perala 1992; Johansson 1999d, e, 2000b).

Table 11.1 Characteristics of ingrowth by 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)

Tree species	No. of stands	Age (years)	No. of stems (ha^{-1})	MAI (tons ha^{-1} year^{-1})
Young stands				
European aspen	6	5–15	6800–46,150	2.9–9.1
Silver birch	4	7–11	4061–45,500	0.7–5,3
Downy birch	6	6–11	35,500–298,000	1,0–4,9
Common alder	8	4–13	9860–40,000	0.5–7.7
Grey alder	10	5–15	7400–94,000	0.7–9.9
Mature stands				
European aspen	43	16–91	245–32,700	1.2–9.0
Silver birch	4	16–32	2280–11,900	4.3–6.74
Downy birch	1	20	2737	2.3
Common alder	34	17–91	431–6200	1.5–6.1
Grey alder	30	17–66	546–17,500	1.5–5.5

After Johansson (1999a, d, e, 2000b, 2002)

Fig. 11.2 Harvested fuel wood from a stand of ingrowth of aspen and birch

11.2.2 Short Rotations of Norway Spruces and Hybrid Larches Planted on Farmland

Another model is to practice short rotation with conifers (Norway spruce or hybrid larch). Short rotation of Norway spruce is not practiced in Sweden. Conifers planted on abandoned farmland areas will grow fast. The problem for forest industry will be a lower basic density with smaller amounts of fiber per cubic meter than for more slowly grown conifers on forest land. Parts of the sawn products cannot be used for constructing purposes. Therefore one solution could be to manage the stand without cleanings and thinnings until the selfthinning phase starts and then clearcut the stands (Johansson 1999b, 2013a). The number of plantations with hybrid larch (*Larix* x *eurolepis* A. Henry) on farmland is increasing. The rotation period will be 15–25 years. However large amounts of hybrid larch are not used as raw material in the pulp industry depending on their chemical contents in the wood. Hybrid larch grows fast and produces high amounts of biomass (Johansson 2013a).

11.2.3 Mixed Forest of Broadleaves and Conifers on Forest Land

A third alternative is mixed forest mostly a conifer and a broadleaf specie. Today mixed stands of birch and Norway spruce is a practiced method for managing young stands. 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 (Tham 1988). The figures are from studies in Nordic countries (Braathe 1998; Johansson 2014).

11.2.4 Short Rotations of Hybrid Aspen and Hybrid Poplar Planted on Farmland

Today results from studies on biomass production from plantations of hybrid aspen and hybrid poplar (mainly *Populus maximowiczii* x *P. trichocarpa*) have been published (Johansson 2013b; Johansson and Karacic 2011). Hybrid aspen is a hybrid between European aspen (*Populus tremula* L.) and trembling aspen (*Populus tremuloides* Michx.). Depending on the purpose with the plantation the range of rotation period is 10–20 years. The hybrid aspen stands produce high levels of biomass in 10 years followed by the second generation with sucker production. In a longer rotation period the aspen produce pulp wood and timber together with biomass residues from harvests.

Poplar hybrid and clone stands produce as high biomass as hybrid aspen. In Sweden most of poplar wood is used for bioenergy purposes as there is no market for timber and a weak market for pulp wood.

11.3 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 localization. The influence on carbon stock is discussed.

11.3.1 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 was made. Especially the difference between short and long rotation periods was 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 11.2.

When managing the young stands there is no conventional way to obtain the potential growth capacity. As shown in Table 11.2 MAI for specie differs between 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

Table 11.2 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)

Tree species	No. of stands	Age (years)	No. of stems ha^{-1}	SI H_{40}, m	MAI t. ha^{-1} $year^{-1}$	Moisture (cont. %)
Young stands						
European aspen	7	10–20	5964–16,500	–	2.9–8.6	46
Silver birch	5	10–17	3301–45,500	–	0.9–8.3	40
Downy birch	4	11–17	2737–32,400	–	0.5–5.4	40
Common alder	4	10–17	3861–21,600	–	1.7–4.5	43
Grey alder	8	10–17	7400–47,600	–	2.0–8.8	45
Mature stands						
European aspen	19	32–64	846–3242	16–20	1.4–6.9	59
Silver birch	8	26–50	950–4061	21–28	1.6–6.7	41
Downy birch	9	30–57	838–3253	19–27	1.7–7.3	41
Common alder	14	31–61	826–2994	15–23	2.4–5.3	53
Grey alder	12	30–66	904–4031	17–24	1.9–4.5	54

After Ericsson and Johansson (2006)

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 be well introduced. The effective heating value (q_v (net)) reported by Nurmi (1993) and moisture content of each species (Table 11.2) 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. 11.3. After three rotation periods (totally 45 years) the young stands of European aspen, silver birch, grey alder and common alder substituted 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 (young stands) with 25 % higher MAI could replace more carbon than mature stands for all species.

As shown in Fig. 11.4 the average carbon stock varied from 32 to 41 tons C ha^{-1} for the mature stands and 9–26 tons C ha^{-1} for the young stands. If the objective is to maximize the average carbon stock the rotation period should be long according

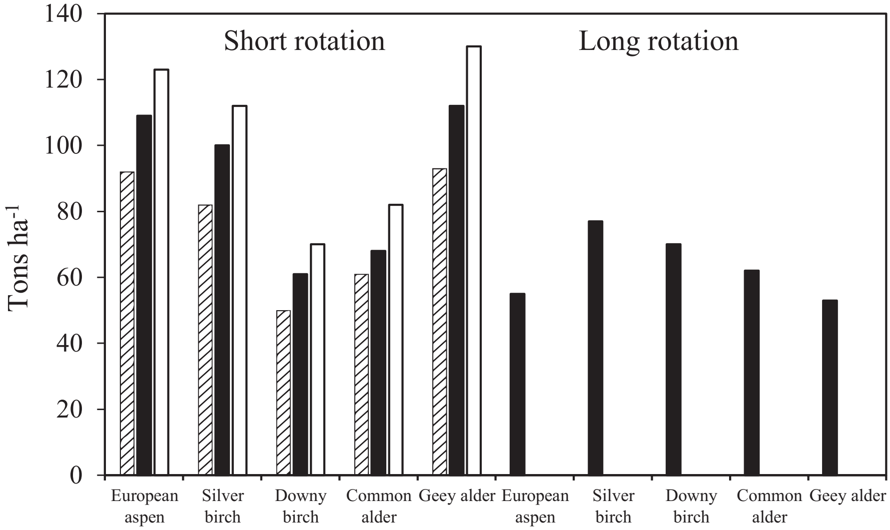


Fig. 11.3 Avoided immersions of CO_2 when replacing coal with biofuel after three rotation periods for the young stands and one rotation period for the mature stands. MAI for the second and third rotation period was equal in all rotations (▨), decreased by 25 % (■) and increased by 25 % (□), respectively, for the short rotation period

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.

11.3.2 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 clearcut at the time for selfthinning. In a study of Norway spruce planted on farmland the selfthinning procedure will start at 40–45 years of age (Johansson 1999b). 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 programs in Norway spruce and Scots pine stands. The stands have been examined every 5–10 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 ten stands had a site index, SI, of > 30 m (H_{100}) and > 1500 stems per hectare when the experiment was initiated, Table 11.3.

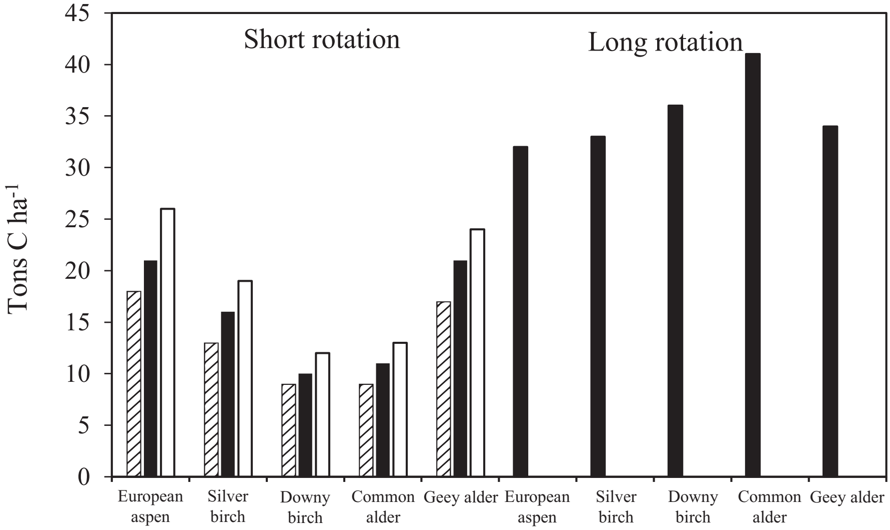


Fig. 11.4 Average carbon stock after three rotation periods for the young stands and one rotation period for the mature. MAI for the second and third rotation period was equal in all rotations (▨), decreased by 25 % (■) and increased by 25 % (□), respectively, for the short rotation period

The biomass production of unthinned stands during the studied period is shown in Fig. 11.5. The amount of biomass at 30, 40 and 50 years was 125, 163 and 204 tons d.w. ha^{-1}. According to the calculation of CO_2 emissions described earlier for ingrowths of broadleaves the emission from the spruce stands were calculated. The average CO_2 emissions were 62, 81 and 103 tons C ha^{-1} after 30, 40 and 50 years rotation.

11.3.3 Mixed Forest of Broadleaves and Conifers

In Sweden mixed forest is the most common structure of forest stands. If no management is done especially colonizing broadleaved species, e.g. the light demanding and fast growing species as *Alnus*, *Betula* and *Populus* will compete with other and make different kinds of mixture (Johansson 2000a), Fig. 11.6. If dense mixed stands are not managed the total growth and the living number of stems will drastically decrease.

When all the birches were harvested in the mixed stands after a rotation of 40 years the total biomass production of Norway spruce in the shelter was 62 and without shelter 65 tons d.w. ha^{-1} (Johansson 2014), Fig. 11.7. The total birch biomass production was 79 tons d.w. ha^{-1}. Based on the mean value the CO_2 emissions by managing birches in a Norway spruces stand was 33 tons C ha^{-1}. As the surplus

Table 11.3 Characteristics of 10 Norway spruce stands growing on former farmland

	SI H_{100}, m	Age (years)	DBH (mm)	No. of stems ha^{-1}	Age (years)	DBH (cm)	No. of stems ha^{-1}
	First examination				*Third examination*		
Mean±SE	34.4±0.6	29±1	117±4	3330±189	45±2	167±4	2293±75
Range	31.5–36.9	23–47	98–155	1820–4533	37–52	148–231	1320–2743

Johansson (1999b)

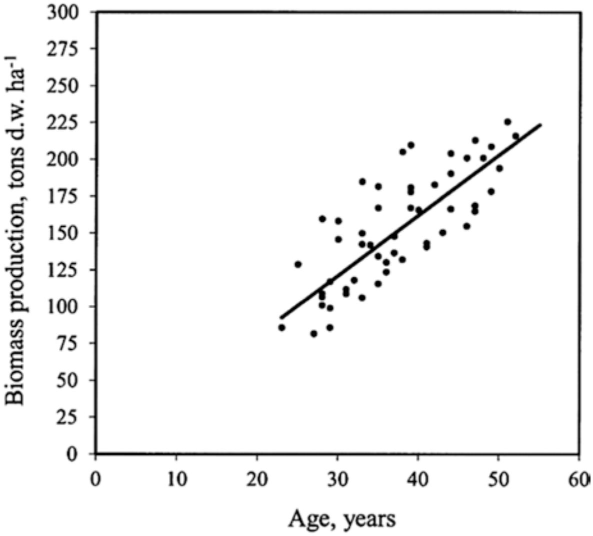


Fig. 11.5 Biomass production (tons d.w. ha^{-1}) in self-thinning Norway spruce stands (After Johansson 1999b)

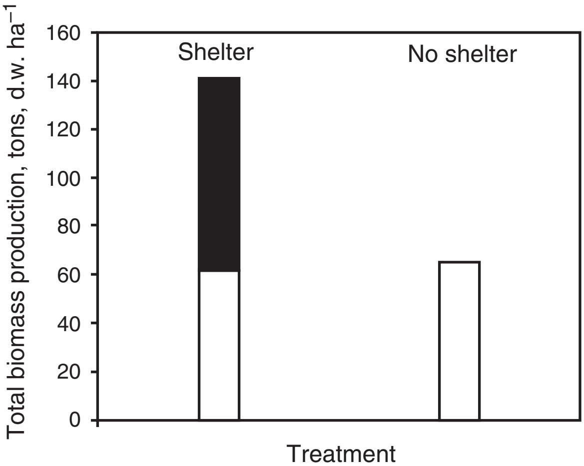
of birches in the Norway spruce stand did not drastically decreased the growth of spruces, carefully managed mixed stands of broadleaves (alder, birch or aspen) and spruces both gives pulp wood or biofuel and less CO_2 emissions than if only a stand of spruces has been practiced. In the conclusions of the referred study it was stated that a correct management of the mixed stands with a cleaning in the birch stand at 10 years of age the biomass production should have been higher than reported (Johansson 2014).

11.4 Conclusions

In Sweden strong efforts are made to reduce the use of oil and nuclear power by utilizing forest land and farmland for harvesting biofuel. An increasing supply of biomass from fast-growing species planted on former farmland is ongoing but

Fig. 11.6 Ingrowth of birch and Norway spruce before (*left*) and after (*right*) cutting of 75 % of birch stem number

Fig. 11.7 Total biomass production (tons d.w. ha^{-1}) of Norway spruce (□) and birch (■) in mixed stands (*left*) and in pure Norway spruce (□) stands (*right*) after a rotation of 40 years



strong efforts are also made by refining management methods and tools for efficient and high production and harvest of biomass from forest land.

By efficient methods the harvest of biomass on forest land today could be increased by 50 %. Harvest technique, level of biomass and ecological consequences of stump harvest are tested today and more studies about the consequences on the sustainability is needed before stump harvest in a larger scale can be recommended. Some of the methods practiced on forest land today are based on optimal rotation periods and adequate management of the stand, including cleaning, and thinning, at the correct time. Severe competition could drastically decrease tree growth.

As most of biofuel harvest in Sweden is based on large scale systems on clearcut areas (5–20 ha) on forest land 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.

Special equipment for biofuel harvests is 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.

Besides the need for the site to be suitable for tree cultivation, the skill of the owners is important. The most important factor, however, is the enthusiasm and curiosity of the owner, without this, most of the methods will not produce the yields suggested in the present study.

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 stands with high stem density both in 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. However the stands should be managed including commonly used methods including final cutting and establishment of a new stand is important and will increase the level of low CO_2 emissions.

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Chapter 12
Innovation in the Value Chain of Wood Products: Data, Equations and Life-Cycle Analysis

I. Lizarralde, F. Rodríguez, and F. Bravo

Abstract This chapter represents a review of the state of the art of the techniques and methodologies described and used in Lizarralde et al. 2008. Several issues have been reviewed and updated, while the work area and most of data remain equal, as an exercise of methodology review. Assessment of CO_2 sequestration on wood products starts to raise importance when measuring the global sequestration in a forest. In this work, a more accurate estimation of CO_2 on wood products is proposed, taking into account not only the volume but the useful life of those products. In order to estimate this carbon uptake and sequestration, innovation in forestry reveals new technologies and methodologies such as forest inventory using LiDAR technology or advanced modelling techniques. Besides, a new accountability and legal framework is added to review the state of the art of this particular issue. With this new information and knowing yield data of transformation of these products in final commercial products, the sequestration rate of each product and its useful life, we will be able to calculate the global sequestration of annual cuttings in an important and well-known forest in Spain.

12.1 Introduction

During the last decade, since the Kyoto Protocol came into effect on February, 2005, thousands of publications have been released related to the accomplishment of the protocol and the ways to measure the carbon fluxes all around the world. From the point of view of forests, it was considered as a great opportunity to enhance the value of our forests, beyond the traditional accounting of wood production and its value.

I. Lizarralde (✉) • F. Rodríguez
Fora Forest Technologies, SLL, Soria, Spain

F. Bravo
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain
e-mail: fbravo@pvs.uva.es

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Although the accounting methodology of the sequestration proposed during the first commitment period (2008–2012) by the UNFCCC (United Nations Framework Convention on Climate Change) through the IPCC (Intergovernmental Panel on Climate Change) did not take into account the forest products as carbon sink (IPCC 2000a), assuming that all carbon in the harvested biomass is oxidised in the removal year (Dias et al. 2005), several countries and the UNFCCC recognized the important role products can play on the global accounting. Thus, it was not until the second commitment period (2013–2020) when new opportunities to wood products (IPCC 2014) were opened.

Besides, in a context where the European Union started defining the inclusion of wood products into the global carbon accounting, especially with the 2012 resolution, positioning wood products in an starting lane as low carbon products becomes an important aim for researchers and the industry. And this aim can only be accomplished with feasible and economically available data for research bodies, consultings and wood industry. Data of life cycle analysis, carbon footprint and useful life of products will help mitigating climate change and enhancing national carbon budgets while allowing the active participation of citizen in the fight against climate change.

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). The use of wood products is one of the few mitigation strategies that do not involve added economical cost. In summary, the calculation and promotion of carbon footprint for wood products in a feasible and reachable way is analysed, allowing both the development of new mitigation strategies and a better situation of national carbon budgets. Forest modeling 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 forest, but the link with the sequestration of CO_2 has not been significantly explored. New approaches of forest model assessment become very useful for carbon accounting purposes.

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.

12.2 Estimation of Carbon Stocks

As we have said, wood products were not included for the first commitment period of the Kyoto Protocol, although the UNFCCC (2003) already assumed different options for the inclusion of wood products in future commitment periods. The three main options were 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 carbon keeps fixed, while accounting refers to the estimation of product and carbon fluxes between and within countries. Quantifying, as it is 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, were the relationships between CO_2-emitting and CO_2-fixing countries stand out. These trends were:

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 o 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 and was the method 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 was rejected by the majority of UE 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 form harvesting while the consuming country does not increase its carbon pool with the imported wood products but must report the emissions when those products decay.

Decay approach

In a very similar way as it was described by Lizarralde et al. (2008), a new mixed approach based on the decay of carbon in products through the useful life has been added to the methodologies of accounting for harvested wood products.

In order to use this useful life, two options were described (Lizarralde et al. 2008): 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 and Nicholson 2000). On the other hand, "average lifetime" is the average time a product is in use, using a linear function of product decaying.

Nowadays, the first option seems likely to be the one used by several countries, assuming a decay equation guided by half-life.

All the methodologies can be used and depending of the consumption, production and import/export balance of each country, one method could work better than others. Nevertheless, the most important issue should be to have good available data (Tier 3). Unfortunately, in Spain, only data for some wood boards are available (Canals et al. 2014). 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, for 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 is demonstrated (EU 2004) that producing a fixed quantity of concrete needs about the double of 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 of this substitution factor is:

$$S = \Delta C / \Delta P = Increment\ \ of\ \ emissions / Increment\ \ of\ \ wood\ \ use \quad (12.1)$$

12.3 Materials and Methods

12.3.1 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. In order to maintain the same example from the previous version of this chapter, no changes were made in data acquisition. Nevertheless, as it will be shown, new inventory technologies could have been used, where accuracy and spatial resolution are the main improvement factors.

12.3.2 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 and Bravo 2005) made it possible to introduce data of the felled trees (Diameter at Breast Height and Total Height) in the cited equation. At the same time, this equation is part of Cubica (Rodriguez and Broto 2003) and CubiFor (Cesefor 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. 12.2.

The taper equation used for this purpose was developed for Scots pine, natural forests in Castile and León (Lizarralde and 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:

$$d = \left(1 + 0.4159\ e^{-16.733h}\right)\left(0.7365\ \ DBH\left(1-h\right)^{0.5869-0.8945(1-h)}\right) \quad (12.2)$$

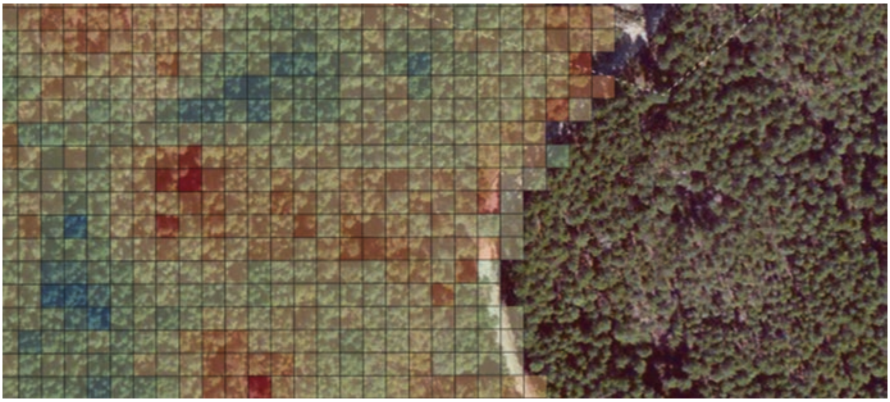

Fig. 12.1 Example of LiDAR inventory cells with volume data and classification

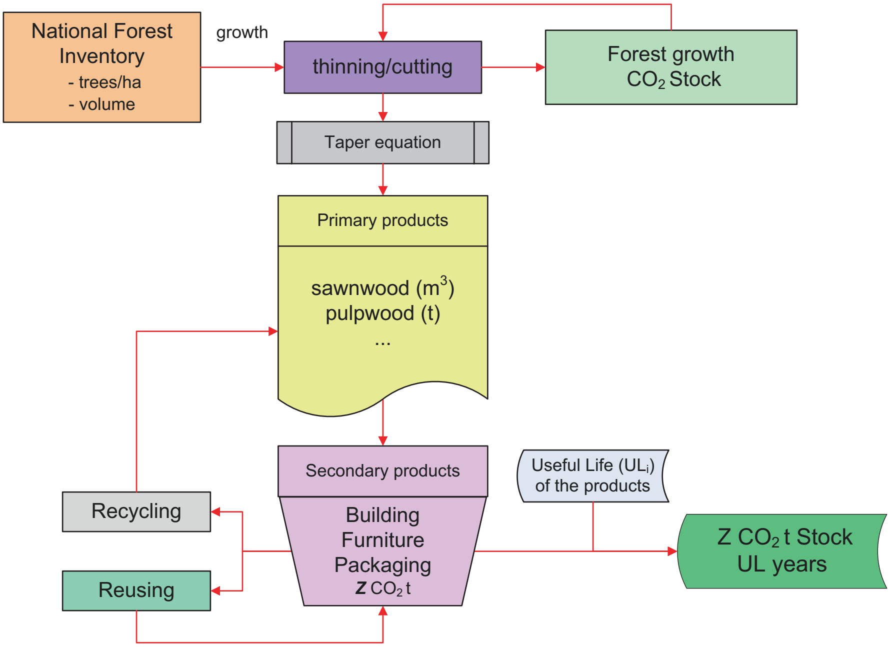

Fig. 12.2 Methodology for the calculation of the CO_2 stored in wood products

where:

d = relative diameter
h = relative height
DBH = Diameter at Breast Height

Although this equation is proved to accurately assess diameter change along the stem and, subsequently, the volume of trees, other equations, such as segmented ones, seem to work as precisely as the one used in the study (Rodríguez et al. 2015)

A way to avoid some assumptions and lead with extensive inventory data is airborne LiDAR technology. LiDAR is a remote sensing active system that allows capturing a bog amount of data of a forest through an inertial system, a LiDAR sensor and gps positioning. This way, a point cloud of laser return of vegetation is created and using forest modelling techniques, it is possible to obtain certain forest variables in a spatially continuous way, For instance, for the purposes of this chapter, the whole area of Pinar Grande has available data and equations to relate the point cloud and volume can be developed. Thus, accurate and economically affordable volume estimation is obtained for the whole area, with data for 625 square meter cells. This technology, that it has been widely used for forest inventories during the last years, is going to play an important role as an accurate estimation of present forest situation and, subsequently, will help proposing new management and cutting plans.

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 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 12.3 shows the flowchart of the production of those products with the recycling fluxes.

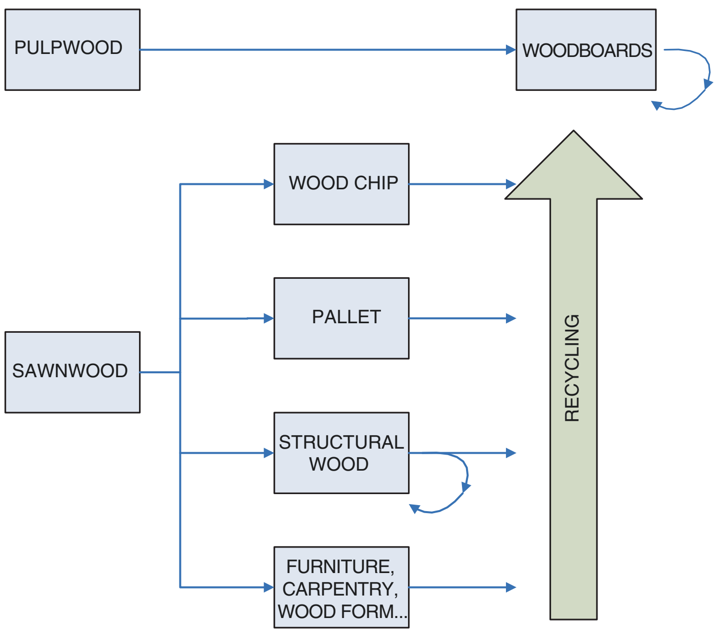

Fig. 12.3 Flowchart of Scots pine wood production in Castilla y León with recycling fluxes

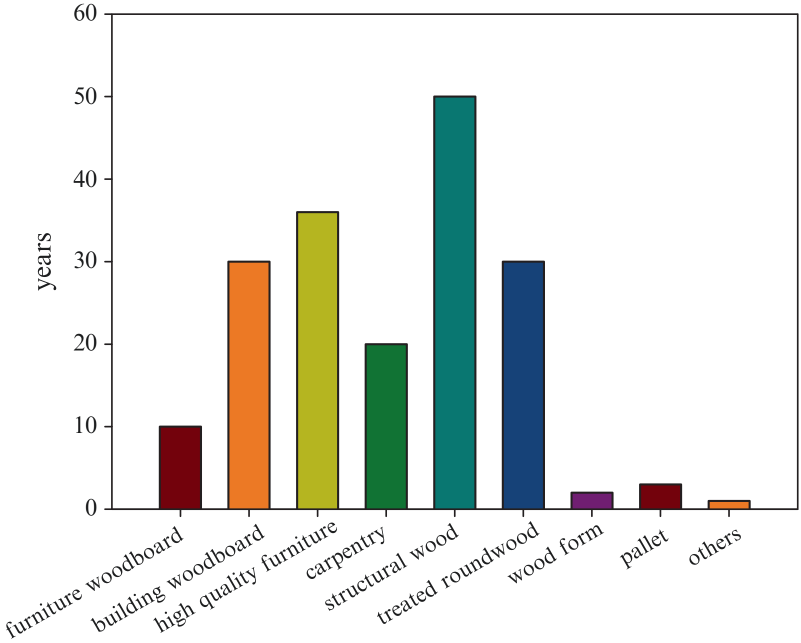

Fig. 12.4 Useful life of the obtained wood products in the process

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 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 and Phelps 1996; Dias et al. 2005). In Fig. 12.4, 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 12.3 shows an example of this methodology:

$$ULi = \%\mathrm{j}\ ULj + \%\mathrm{k}\ ULk + \cdots + \%\mathrm{n}\ ULn \tag{12.3}$$

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.

12.4 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 three 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 (Table 12.1). 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, Path2 (García-Abejon 1981) and the other is a moderate proposal from the yield table for the species in the Guadarrama range, Path3 (Rojo and Montero 1996). The path named "Path1" is the one proposed in this chapter and its objective is to maximize carbon pools based on strong cutting paths.. With these different silvicultural options, a simulation of the evolution of carbon stock (in forest and products) was done for the next 50 years. Table 12.2 shows the evolution of CO_2 with the different silvicultural paths.

Table 12.1 Thinning percentage by diametric class for Scots pine with different management paths

DC	No cutting	Path1	Path2	Path3
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%

Where Path1 is the silvicultural option proposed in this chapter, Path2 is the yield table of García-Abejón (1981) and Path3 is the yield table of Rojo and Montero (1996)

Table 12.2 CO_2 evolution on growth, products and globally for different silvicultural paths. The "Authors" path is the proposed in this chapter

	years				
Paths	10	20	30	40	50
Growth (CO_2 tones)					
No cutting	15,977,417	16,170,680	11,192,515	5,281,162	2,161,514
Path1	1,110,201	762,636	54,506	−874,599	−1,275,781
Path2	7,170,277	7,697,776	8,145,751	8,557,277	8,974,851
Path3	4,643,226	1,259,296	−722,365	−1,596,384	−1,938,542
Products (CO_2 tones)					
No cutting	0	0	0	0	0
Path1	14,867,216	29,801,101	41,846,954	53,616,554	59,333,896
Path2	8,807,140	17,034,699	21,206,620	25,274,917	27,849,496
Path3	11,334,191	24,977,231	32,165,208	37,741,974	39,884,001
Total (CO_2 tones)					
No cutting	15,977,417	16,170,680	11,192,515	5,281,162	2,161,514
Path1	15,977,417	30,563,737	41,901,461	52,741,954	58,058,115
Path2	15,977,417	24,732,476	29,352,370	33,832,193	36,824,347
Path3	15,977,417	26,236,527	31,442,843	36,145,590	37,945,459

Where Path1 is the silvicultural option proposed in this chapter, Path2 is the yield table of García-Abejón (1981) and Path3 is the yield table of Rojo and Montero (1996)

This data produce the following figure (Fig. 12.5), 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 and 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 CO2 accumulation while it avoids carbon emissions from big, old trees if not harvested.

Reducing the scope of the analysis (and in order to clearly show the importance of the management plans and the need of quantifying carbon pools on wood products), a case study from the even-aged Scots pine "Pinar Grande" based on the next ten years 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. 12.6, the proportion of wood assigned to each product by diametric class is shown (Table 12.3).

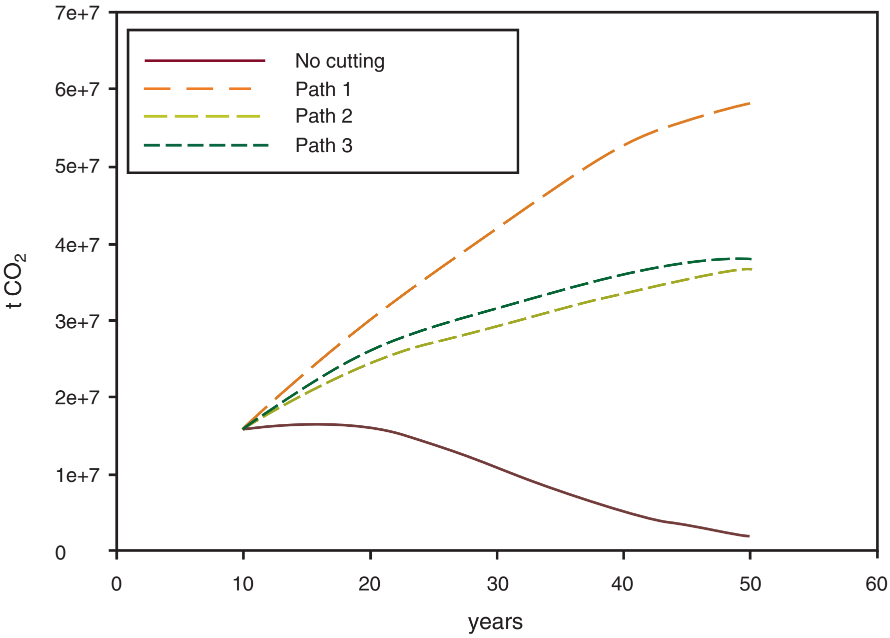

Fig. 12.5 Comparative analysis of CO_2 sequestration from different management paths for Scots pine in Castilla y León

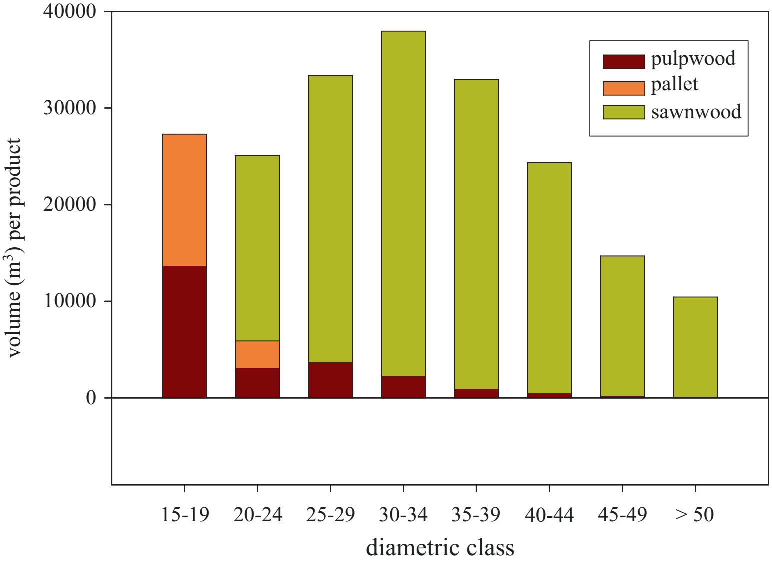

Fig. 12.6 Distribution of volume per product for each diametric class

Table 12.3 Allowable cut in "Pinar Grande" in a 10-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

Data from Forest Management Plan (Junta de Catilla y León, Spain)

Once we have simulated the cutting plan 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 obtained products, a 50-year simulation was performed.

The final result of fixation in the 8750 ha of Scots pine in "Pinar Grande" is almost 600,000 tons of CO_2 which will not be emitted to the atmosphere due to the manufacturing of wood products, as shown in Fig. 12.7. 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 bigger than what is shown.

The sequestration of the forest itself (taking into account only the stem of the trees) reaches some 750,000 tons of CO_2. This means that the proposed harvesting level in the next 50 years will increase a %78.8 the sequestration capacity of the forest and around 81.6% if the harvesting is concentrated in trees with diameter bigger than 35 cm.

Thinning strategies have not a big influence in the carbon sequestration in themselves, but the products that those strategies can create in the future forest will increase the sequestration rates, in a 7.12 % if the diameters reached are around 25 cm and even at 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 is better to produce for a certain forest, rather than the recycling rates this product will have in the future.

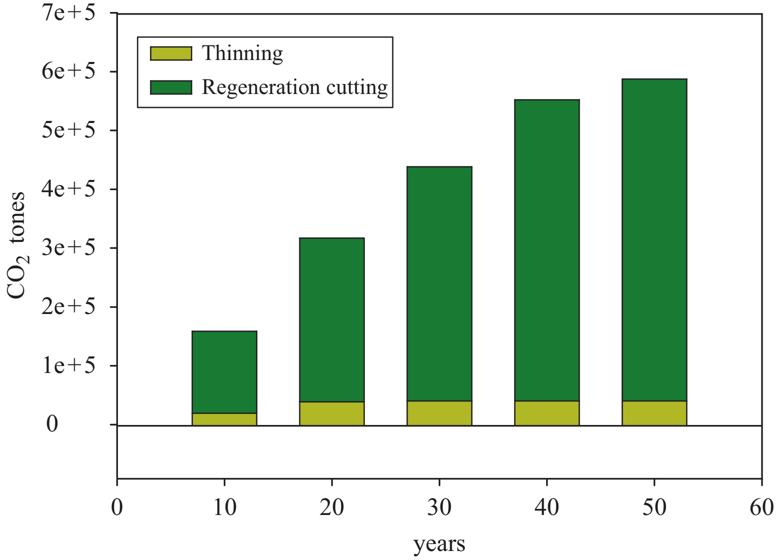

Fig. 12.7 CO_2 sequestration by Scots pine wood products for the next 50 years in "Pinar Grande"

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 analyzed.

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 builded per year, the balance of substituting, for instance, aluminium for wood would represent more than 1.8 million CO_2 tones (Lizarralde et al. 2008), enough to absorb the emissions of 750.000 cars. Although this data is no longer valid, as building capacity in Spain has been largely reduced due to the economic crisis (specially affecting building sectors), the fact of the carbon balance of substitution of aluminium by wood remains true.

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 framework for wood in a single house means 20 t of CO_2.

Finally, producing all the utility and telephone poles in Spain with concrete may represent the emission of more than 3.6 million CO_2 tones. Otherwise, if made with wood, 2.6 million CO_2 tones 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 the 7 % of the energy consumed in Spain.

12.5 Conclusions

The use of taper equations and other forest modelling techniques, 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 use of forest inventory technologies, such as LiDAR technology will help acquiring spatially continuous and precise data. In this way, more and better information will be available in order to develop broader studies of carbon budgets in forests and wood products.

The integration of the wood products in the global accounting of the carbon balances can represent a great advantage towards the accomplishment of the Kyoto Protocol, being as important as the forest growth. The first version of this chapter (Lizarralde et al. 2008) aimed to be a starting point to try to include them for the next commitment period (2013–2017). Nowadays, harvested wood products can be included in national carbon budgets. Nevertheless, few decisions and changes have been done in accounting methodologies. 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 of the development of these kind of policies. The election 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 treatments and plans focused on sustainable management and obtaining long-lived products will optimize the global influence of our forests upon climate change, maintaining at the same time our natural resources.

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Chapter 13
Forest Carbon Sequestration: The Impact of Forest Management

Felipe Bravo, Miren del Río, Andrés Bravo-Oviedo, Ricardo Ruiz-Peinado, Carlos del Peso, and Gregorio Montero

13.1 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: living plant biomass (stem, branches, foliage, roots), plant detritus (fallen branches and cones, forest litter, tree stumps, tree tops, logs) and soil (organic mineral humus, 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 through autotrophic respiration. When leaves or branches die and decompose, they increase soil carbon and also release a small amount into the atmosphere through heterotrophic respiration.

Recent climate changes have resulted in highly variable weather patterns and the general trend of rising temperatures is increasing evapotranspiration in forest

F. Bravo (✉)
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain
e-mail: fbravo@pvs.uva.es

M. del Río • A. Bravo-Oviedo • R. Ruiz-Peinado • G. Montero
iuFOR – Sustainable Forest Management Research Institute, Universidad de Valladolid – INIA, Palencia, Spain
e-mail: delrio@inia.es; bravo@inia.es; ruizpein@inia.es; montero@inia.es

C. del Peso
Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Valladolid, Spain

Department Producción Vegetal y Recursos Forestales, ETS Ingenierías Agrarias, Universidad de Valladolid, Palencia, Spain
e-mail: cdelpeso@pvs.uva.es

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ecosystems. A drop in available water for vegetation growth is expected to accompany this rise in temperature. Plants will respond to reduced water availability by closing stomata, resulting in lower rates of gas exchange. Although higher CO_2 concentrations from burning fossil fuels would be expected to increase photosynthetic rates, closed stomata may mitigate any positive impact in this regard. 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 phase. Additionally, distribution of the energy captured by plants among different functions such as stem increment, branch and leaf formation, flowering, and fructification may be altered under the erratic weather conditions that climate change might generate.

Given that each tree species has an optimum temperature range for development, a widespread rise in temperature will modify the competitive balance between species and may alter species distribution patterns. Though adult trees are very unlikely to suffer generally from sudden death, this is possible in extreme conditions such as the drought affecting holm oaks (*Quercus ilex* L.). For example, Prieto-Recio et al. (2015) found that water deficits and high competition triggered the decline of *Pinus pinaster* Ait. stands in the Iberian Peninsula. Other problems related to regeneration and the initial development of forest species in certain areas will likely occur also. Ruano et al. (2009) identified water availability as a key factor in the germination and early growth of *Pinus pinaster* and later found water stress to be a mediator in seed production (Ruano et al. 2015). Changes in precipitation amounts and regimes will likewise reduce natural regeneration in Mediterranean forests. Fujimori (2001) pointed out that these effects will be especially serious at the margins of plant distributions, where competition for resources is more pronounced. Disease and insect attacks may also become very severe (Melillo et al. 1996). As a result, climate change has been classified as predisposing factor for forest decline (Hennon et al. 2009). Conservation or even promoting greater biodiversity, with regards to both species and genotypes within each species, should constitute a management priority for reducing the effects of climate change. Mixed-species and other complex forests are potentially more adaptable to climate change, so promoting these forest types can help managers to cope with increasing uncertainty (Bolte et al. 2009; Kolström et al. 2011). Greater diversity of tree species can also limit damage from pests (Jactel and Brockerhoff 2007) and diminish the risk of biological invasion, according to the associational resistance hypothesis (Barbosa et al. 2009).

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”. He subsequently defined forest management as the practical application of principles from a variety of disciplines, including biology, ecology, and economics, to the regeneration, density control, use, and conservation of forests. Silviculture and forest management were developed as sciences in Europe in the eighteenth century to satisfy the regular, continuous need for fuel and construction wood. Used together, they can mitigate the impact 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 with biomass-derived fuels; and (4) reducing the use of products that require fossil fuels for manufacturing through use of renewable forest products such as wood, resin, and cork. Maintaining forest areas and using forests as a source of renewable energy will have the greatest impact worldwide.

In this chapter, we describe alternative ways in which forests and forestry can help to mitigate climate change, along with the potential impact of these activities. The three carbon storage compartments should be considered in all impact estimates. Carbon content in living biomass is easily estimated via species-specific equations or by applying factors to oven-dry biomass weights (e.g., Ibañez et al. 2002; Herrero et al. 2011; Castaño and Bravo 2012). Litter carbon content has been analysed in many studies on primary forest productivity, though information regarding the influence of forest management on litter carbon content is less abundant (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 (Lindner and Karjalainen 2007). Hoover (2011), for example, found no difference in forest floor carbon stocks among stands subjected to partial or complete harvest treatments in the United States.

13.2 Forest Management and Carbon Sequestration During the Last Century

In its reports regarding on mitigation, the Intergovernmental Panel on Climate Change (IPCC 2001, 2007, 2014) warned of the temporality of forest carbon deposits and the possibility of extensive tree mortality from large-scale disturbances such as droughts and forest fires, leading to massive emissions. Globally, the more prominent carbon loss in tropical zones due to human-caused deforestation is being offset by expanding forest areas and increasing wood stocks in temperate and boreal forests.

Human activities and land use have historically affected carbon storage, emissions, and sequestration. For example, U.S. forests were carbon sources from 1700 to 1945. Since then, fire suppression and forest renewal in abandoned farmland have reversed the trend and forests have become carbon sinks (Houghton et al. 1999). Woodbury et al. (2007) found that total U.S. carbon stocks had increased since 1990 and were expected to continue increasing through 2010. In contrast, Pacala et al. (2001) reported a stable carbon sink in the continental U.S. (excluding Alaska), with similar values for 1980–1989 and 1990–1994. China offers another example of how national forest management can alter carbon storage trends. After the social revolution in 1949, the carbon content in living biomass decreased due to human pressure on forest resources. From 1970 to 1998, afforestation and reforestation programs were implemented to increase forest land and, subsequently, stored carbon (Fang et al. 2001). This effort continues and its impact on forest carbon stocks will be relevant in future decades (Zhou et al. 2014). Bellassen and Luyssaert (2014) pointed out how assessments of forests as carbon sinks or sources rely on specific

Table 13.1 Annual CO_2 sequestration rates in temperate and Mediterranean forests in Spain

Zone	Years	IFN2 (10^3 Tn)	IFN3 (10^3 Tn)	Annual rate (%)
Cantabrian range	1991–2000	41,696	45,433	0.9583
Castillian plateau	1992–2992	17,623	21,635	2.0723
Basque Mountains	1994–2003	32,608	50,601	5.0035
Catalonian coastal range	1991–2001	15,591	19,532	2.2792
Demanda range	1992–2003	5195	95,244	5.6200
Pyrenees	1994–2003	120,159	185,812	4.9628
Central range	1991–2002	57,164	72,404	3.2069
Toledo Mountains	1992–2001	9619	11,802	2.2986
Sierra Morena	1994–2001	15,518	19,542	2.3324

Adapted from Bravo et al. (2007)
IFN2 and IFN3 are the second and third Spanish National Forest Inventories, respectively

hypotheses about processes, including the carbon-neutral role of mature forests and the life-cycle of wood products. Clarifying the underlying processes involved in forest carbon stocks and cycles will reduce uncertainty and generate more reliable predictions, possibly confirming the persistence of forests as carbon sinks.

As noted earlier, the positive effects of forests as carbon sinks are endangered by large-scale disturbances. Liu et al. (2002) showed that the forests of Ontario, Canada could have been considered carbon sinks between 1920 and 1975, but became carbon sources after that because of large-scale natural and human disturbances including wildfires, pest infestations, and extensive harvesting. Schmid et al. (2006) noted that forests with minimal management serve as carbon sinks in the short term, but accumulated biomass will increase risks of fire and pest occurrences.

In many European countries, forest management did not begin until the nineteenth and twentieth centuries. Once implemented, forest management planning, activities, decisions, and results have since been recorded. Consequently, we can determine how CO_2 fixation as forest biomass has evolved. Current forest management activities are improving silvicultural activities aimed at increasing the quantity of fixed carbon in the forests. For example, (Montero et al. (2004) found that from 1993 to 2003 net carbon fixation increased by 6.28 % in the woodlands of Monte de Valsaín (Segovia, Central Spain) Similarly, in the Pinar Llano woodlands of Valladolid (Northern Plateau, Spain) the amount of fixed carbon was expected to increase by 7.23 % in the next 10 years, based on the rate of gain presented in the Management Project Review (Martín 2005). Bravo et al. (2007) analysed 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 13.1). These results were obtained by comparing outcomes from the second and third Spanish National Forest Inventories (INF2, INF3) using biomass equations by Montero et al. (2005) and converting to equivalent carbon.

Bravo et al. (2010) studied the evolution of fixed carbon dioxide in the Mediterranean maritime pine forests (*Pinus pinaster* Ait.) of Almazán (Soria,

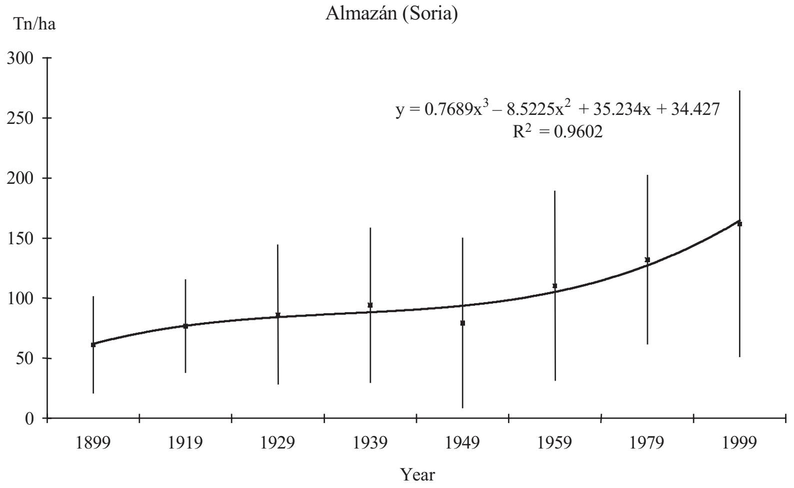


Fig. 13.1 Evolution of CO_2 sequestration in the pine woods (*Pinus pinaster* Ait.) of Almazán (Soria, Northern Spain) during the twentiethcentury

Central Spain). They analysed a century of woodland management in Almazán, from 1899 to 1999, obtaining the numbers of trees by size from the original planning documents and successive planning revisions. This information was combined with the biomass equations developed by Montero et al. (2005) to reconstruct the evolution of CO_2 fixation in these forests. Carbon sequestration increased during the century studied, oscillating between 0.78 and 3.11 Tn/ha per year (Fig. 13.1). The only exception was the period immediately after to the Spanish Civil War (1936–1939), when greater pressure on natural resources due to poverty caused a decrease in biomass (and a reduction of 1.49 Tn of CO_2/ha per year). The CO_2 forest biomass sequestration levels in these pinewoods did not recover until 15 years after the end of the war. Forest management under a sustained yield paradigm that also maintains or increases other forest values and services has been traditionally implemented in the Almazán forests. This management regime maintains or increases forest carbon stocks over the long term, while producing goods and values according to IPCC goals (2007, 2014). The increases in carbon sequestration achieved in the Almazán forest align with evidence by Nabuurs et al. (2003), which showed that European forests increasingly became carbon sinks between 1950 and 1999. Their carbon calculations were based on simulations of carbon storage using net biomass production (i.e., final production including harvest and disturbances). Later, Nabuurs et al. (2007) stated that sustainable forest management strategies designed to increase or maintain forest carbon stocks while producing a constant annual yield of goods (wood, fibre, etc.) and environmental services (water, biodiversity, etc.) will generate the largest long-term sustained mitigation benefit.

13.3 Strategies to Improve Carbon Sequestration

Carbon storage in forests and forest products has been proposed as an appropriate strategy for mitigating the effects of climate change, In spite of this, forest products were excluded from the Kyoto protocol. To a certain extent, carbon storage in forests buys time, until we find more definitive solutions to our dependency on energy from fossil fuel. However, forests can easily become carbon sources rather than carbon sinks (Kurz and Apps 1999; Gracia et al. 2001; Reichstein et al. 2002). Changes in the regimes of natural disturbances such as fire, pests or drought (Fuhrer et al. 2006; Sohngen et al. 2005; Ciais et al. 2005), can affect major forest functions, forestry outputs and forest stability. Metsaranta et al. (2010) reported that Canadian forests will likely be carbon sources until 2030, and become carbon sinks after 2050, based on simulations of the impact of future fire and insect disturbances.

Biomass and carbon accumulation in forest stands can be increased through a variety of management options (Gracia et al. 2005). These include fire, disease, and pest control; increasing rotation lengths (i.e., time to harvest); density regulation; fertilization and other activities to improve soil nutrients; species and genotype selection; management of post-harvest residues and advances in fibre processing and biotechnology. Such activities could increase the carbon accumulation rate by 0.3–0.7 Mg C ha^{-1} $year^{-1}$ (Gracia et al. 2005). Management practices that alter species composition, rotation lengths, and thinning regimes, or that result in forest conservation, increased forest land, and soil conservation can also increase carbon sequestration in forests.

13.3.1 Species Composition

Carbon storage varies according to species composition and site quality (Bravo et al. 2008). For example, the amount of carbon per unit biomass is greater in conifers than in broadleaf trees (Ibáñez et al. 2002). In Mediterranean areas, Scots pine (*Pinus sylvestris*) stands store more CO_2 than pure oak (*Quercus pyrenainca*) stands (Bogino et al. 2006), while mixed oak-pine stands store intermediate levels. Differences among pine species have also been reported (Bravo et al. 2008).

Many studies have reported greater forest productivity with increased species diversity (e.g. Vilà et al. 2007; Paquette and Messier 2011). As a result, mixed-species forests are likely to have higher carbon storage capacity. Similarly, the productivity by space occupancy of many species is frequently higher when admixed with other species due to facilitation and/or complementarity, often resulting in over-yielding or even transgressive over-yielding compared to corresponding monospecific stands (e.g. Río and Sterba 2009; Condés et al. 2013; Pretzsch et al. 2015). Forest productivity is generally evaluated based on volume or aboveground biomass per unit area, calculated via tree volume or tree biomass allometric equations. These were often developed using sample data from monospecific stands, resulting in less accurate biomass estimates (Forrester and Pretzsch 2015). Carbon content per unit of biomass may also vary, depending on whether stands are mixed or monospecific. Additionally, belowground biomass is difficult to estimate and often excluded from estimates, and niche complementarity between species can also occur belowground (Brassard et al. 2013). These issues underscore the need for more research focusing on carbon storage in mixed-species forests, in order to determine whether they fix more CO_2 than monospecific forests.

Species composition can also modify soil carbon storage, as tree species identity mediates soil carbon distribution between the forest floor and mineral soil (Vesterdal et al. 2013). There are indications of a continuum species effect mediated by local conditions rather than a global dichotomy effect such as deciduous vs. conifer (Prescott and Vesterdal 2013). In a study of beech (*Fagus sylvatica* L.) forests and *Pinus nigra laricio* plantations in Calabria (Southern Italy), Scarascia-Mugnozza et al. (2001) found that beech stands stored 1.47 times more soil carbon than pine plantations. Co-existence with species that fix nitrogen can also increase carbon accumulation. Chiti et al. (2003) found that mixed oak-alder (*Quercus robur* L.-*Alnus cordata* Desf.) 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 mix of tree species affects soil fauna assemblages (Chauvat et al. 2011), whereas litter type and initial leaf-litter concentrations of co-existing species affect soil processes such as litter decomposition and nutrient release (Aponte et al. 2013).

The selection of the best species composition for a stand depends on many factors (management objectives, site characteristics, etc.), and CO_2 storage should also be included as an 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 and forests.

13.3.2 Rotation Length

During immature and mature stages of stand development, forests are carbon sinks. In older forests, carbon sequestration may continue to increase slowly or may decrease slightly. Rotation length can be extended in order to maximize or maintain forest carbon sinks while obtaining other goods and services. Different criteria can be used to determine the appropriate rotation for obtaining forest products while achieving forest sustainability. One widely-used criterion in the management of regulated forests is to set the rotation length at biological rotation, or maximum mean annual increment (MAI) of volume per unit area. This practice maximizes longer-term wood production (several rotations), while promoting other products and services that society demands (wild mushrooms, hunting, ecosystem conservation, etc.). Rotation has a two-fold impact on carbon storage in forests (Table 13.2). 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). Similarly,

Table 13.2 The impact of species, site quality and rotation on carbon sequestration in stands of Scots pine (*Pinus sylvestris* L.) and Mediterranean Maritime pine (*Pinus pinaster* Ait.)

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

Adapted from Bravo et al. (2008)

[a]Site index is the dominant height in m at 100 years (*Pinus sylvestris*) or 80 years (*Pinus pinaster*)

since products from harvests are often destined for long-term uses (e.g., furniture, construction etc.), products made from harvests after longer rotations result in greater carbon storage.

If the rotation length is very long, tree mortality rates will increase, resulting in an increase in structural diversity with dead and fallen trees. This, along with regeneration in the gaps created is related to an overall increase in species diversity (Franklin et al. 1997). Tree mortality contributes to coarse woody debris (CWD), a key carbon reservoir in mature and old-growth forests. Differences in CWD stocks have been found among forest types (Herrero et al. 2010, 2014a). The decomposition rates of dead woody materials varies with the species, size, type of material (i.e., bark, sapwood or heartwood), and site conditions (i.e., temperature, humidity, etc.). Dead wood has an important impact on carbon storage in forest systems because it may increase the risk of perturbations, resulting in sudden outbreaks of fires, pests, and pathogens. Other impacts on the amount of soil carbon may also occur due to increased rotation lengths. Using the CO2FIX model, Kaipainen et al. (2004) reported a decrease in soil carbon stocks for some cases in Europe when the rotation length was increased.

There is evidence that biomass allocation among different components varies with tree age. For *Pinus sylvestris* and *Pinus pinaster*, Bravo et al. (2008) found that with age biomass allocation to stems increases, while allocation to branches decreases. Added to the importance of larger stems for carbon storage in stands, as well as in wood products created from these stems, the biomass distribution in trees harvested after longer rotations had a considerable impact on the possible use of harvest debris to generate energy. Bravo et al. (2008) have demonstrated that the percentage of biomass for pinewood branches between 2 and 7 cm in diameter decreases with age.

The proportion of carbon stock in the final harvest relative to total fixed carbon in the stand is higher for longer rotations. However, a shorter rotation is associated with higher carbon MAI values at rotation, regardless of the site index (Bravo et al. 2008, Table 13.2). Longer rotations on poor sites can result in carbon storage similar to that of shorter rotation on good sites, as shown for *P. sylvestris*. Additionally, a longer rotation on a poor site may produce stems large enough to be suitable for lumber and other products. Thus, long rotations could be applied to the poorest sites in order to achieve both carbon sequestration and timber value objectives. Bravo and Diaz-Balteiro (2004) showed that more extensive management systems that involve lengthening rotations result in a loss of economic return compared to traditional management with shorter rotations. However, when carbon sequestration income is included in the analysis, longer rotation alternatives present a positive land expectation value. Increasing harvest rotation length would lead to reduced harvest rate over a landscape. Under such circumstances, 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 pool dynamics on a broad temporal and spatial scale should be included in management planning.

13.3.3 Thinning

Management of tree density by thinning is one of the most important silvicultural interventions for achieving both economic and ecological objectives (Río 1999):

- To reduce competition in order to procure biological stability and improve health.
- To regulate or maintain the specific composition and to prepare the stands for natural regeneration.
- To obtain production yield at early stages, in such a way as to maximize production at the end of the rotation.
- To increase the value and dimensions of the remaining products.

However, thinning will affect the amount of stored carbon. In particular, aboveground tree biomass is reduced immediately after thinning, along with litterfall inputs and accumulation on the forest floor. However, carbon sequestration rates may increase after thinning, as the growth rates of residual trees are altered.

Long-term experiments are essential for obtaining knowledge about the effects of thinning on carbon storage and sequestration rates. A few studies have shown that unthinned stands present higher carbon stocks in tree living biomass than thinned stands, because higher stocking from moderate or heavy thinning results in lower carbon stocks over the long-term (Skovsgaard et al. 2006; Powers et al. 2011; Ruiz-Peinado et al. 2013, 2016). However, in some cases very light thinning has a positive effect on carbon storage, as Keyser and Zarnoch (2012) found. Though maintaining a high tree density could maximize on-site carbon stock, it may also increase the risk of natural disturbances (Jandl et al. 2007). Increasing off-site carbon storage via thinning may prove a better strategy, especially in high risk areas. Tree carbon removed by thinning operations should be included in calculations of total carbon stocks and carbon sequestration rates in order to compare thinning intensities, as is often done for volume production (Assmann 1970).

Along with thinning intensity, the type of thinning may also affect carbon stocks and sequestration rates. Hoover and Stout (2007) showed that thinning from below presented higher carbon stocks in tree biomass than thinning from the middle or from above. With thinning from below, growth is concentrated in the larger trees that are retained: smaller trees with lower net productivity are removed, making resources available to the residual trees. Similarly, D'Amato et al. (2011) found equal amounts of carbon in tree biomass with thinning from below as with a combination of thinning from above and from below (i.e., multiple thinning events) over the long term.

Thinning can also reduce the amount of deadwood, since dying or dead trees are often removed in thinning operations. Other carbon pools such as soil carbon (forest floor and mineral soil) may also be affected. Reduced density from thinning may alter microclimatic soil conditions, thereby affecting soil temperature and moisture. Thinning activities may also result in soil compaction as well as mixing of forest floor litter with upper soil layers. As noted earlier, fewer trees imply lower litterfall inputs and higher carbon losses as a consequence of higher respiration rates (Jonard et al. 2006).

The harvesting method can result in different impacts of thinning on forest carbon. Whole-tree harvesting might have more intense impact than stem-only harvesting, where thinning residues that could reduce soil impacts such as soil compaction are retained on-site (Tarpey et al. 2008; Han et al. 2009). In a meta-analysis, Johnson and Curtis (2001) found that harvesting had no statistically significant effect on soil carbon stock. However, these authors also reported differences depending on the harvest method used: there was a slight reduction in soil carbon stocks when whole-tree harvesting was applied and a moderate increase with sawlog harvesting; but this was restricted to coniferous species. The Nave et al. (2010) meta-analysis found that harvesting reduced soil carbon stocks in a small but significant way: forest floor carbon stocks decreased markedly and no influence was detected in the mineral soil, though great variation was identified between soil orders. This forest floor carbon stock reduction tendency for thinned stands is corroborated by several studies (e.g., Vesterdal et al. 1995; Jonard et al. 2006; Powers et al. 2012; Ruiz-Peinado et al. 2013), though other authors have reported little or no influence (Novák and Slodicák 2004; Chatterjee et al. 2009; Jurgensen et al. 2012).

Analysis of the effects of thinning on total ecosystem carbon should include all pools: tree biomass (above and belowground), understory (shrub, herbaceous…), deadwood, forest floor and mineral soil. To obtain the most complete results, the carbon removed in thinnings should be also incorporated into the analysis, as mentioned earlier.

The works of Ruiz-Peinado et al. (2013) and Bravo-Oviedo et al. (2015) in Spain indicated no significant influence of thinning on total ecosystem carbon stock, when compared to unthinned stands at the end of the rotation period. In another study (Ruiz-Peinado et al. 2016), early thinning in the middle of the rotation period slightly reduced the total carbon stock .

In a chronosequence study on *Pinus resinosa*, Powers et al. (2012) found that thinning did not have a significant effect on total ecosystem carbon stock. However, in a previous study of the same species, Powers et al. (2011) found that thinning reduced the total carbon stock when these treatments were applied at 5–10 year interval. These results suggest that thinning rotation period is another important aspect of forest management to consider in relation to carbon storage.

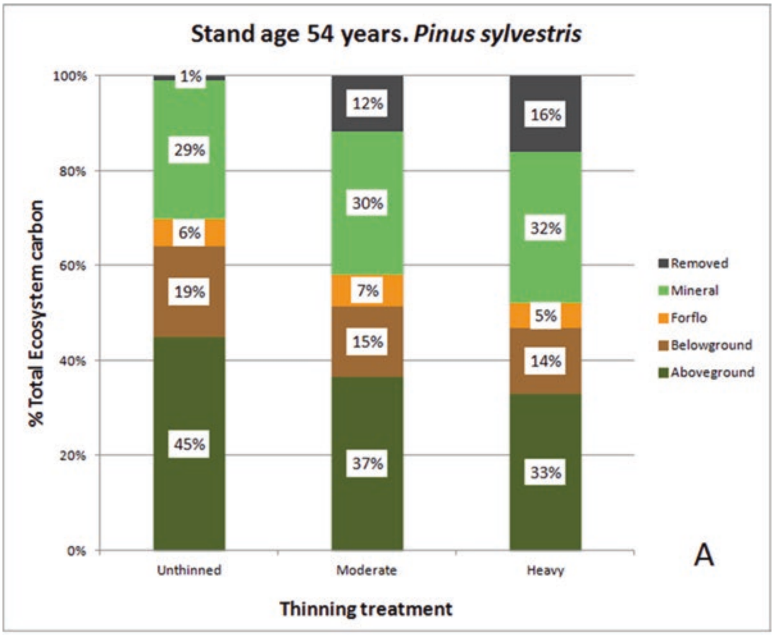
Stand age also influences the carbon amounts that are stored in the different forest compartments. Stands have different carbon sequestration rates at different ages, even when managed under a similar thinning plan. Using data from the studies of Ruiz-Peinado et al. (2016) and Bravo-Oviedo et al. (2015) on *Pinus sylvestris*, Fig. 13.2 shows that the greatest carbon pool was found in the aboveground tree biomass of medium-young stands, whereas in old stands (near the rotation period for this species) the most important pool was located in the mineral soil. Carbon removed by thinning constitutes an important pool in both types of managed stands, especially if we consider that wood products can store carbon for long periods.

Pre-commercial thinning should be applied to very dense stands, (e.g. in natural regeneration after a forest fire), in order to reduce tree density to near the suggested values. Results from a study of *Pinus halepensis* by De las Heras et al. (2013) showed that applications of early and heavy treatments presented carbon amounts similar to those of unthinned plots. However, thinning accelerated cone production and stand maturity (2–5 years after) and tree growth (2–4 years after), thereby increasing stand resilience against perturbations such as forest fires (Ruano et al. 2013). Similarly, Jiménez et al. (2011) found for *P. pinaster* that early and moderate thinnings resulted in no total carbon biomass differences compared to unthinned plots.

In general, the findings and results of the different studies indicate that the effect of thinning on soil carbon stocks is not quite significant, although a high degree of variation exists according to species, harvesting methods, soil types, etc. Living biomass decreased with the thinning interventions in the long-term (reduction of tree density). However, in areas with high risk of fire or other disturbances (winds, heavy snow, etc.) or when pests or disease may endanger the ecosystem, moderate or heavy thinning helps maintain tree cover and improve carbon sequestration on-site as well as off-site, in wood products or as bioenergy.

13.3.4 Conservation of Forests

Although forestry activities have different carbon mitigation potentials depending upon ecosystem features, the short-term carbon mitigation benefits of conserving current forests by reducing deforestation outweigh the benefits of increasing forest area (IPCC, 2007). Between 1990 and 2005, 13 million ha per year of forest land were lost to other uses (FAO 2006), and deforestation rates were highest in South America, South and Southeast Asia and Africa (Table 13.3). Forest loss rates are currently decreasing worldwide (from 7.3 million ha per year in 1990–3.3 million ha per year in 2015), but total forest area declined by 3 % between 1990 and 2015) (Keenan et al. 2015), with differences between climatic domains. While subtropical



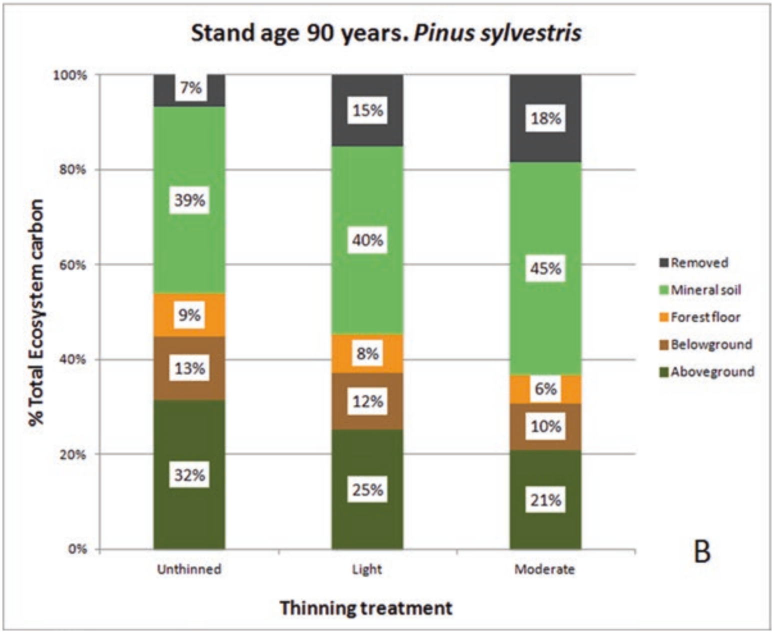


Fig. 13.2 Total carbon proportions found in the different pools of *Pinus sylvestris* stands after 30 years of forest management and 3 thinnings. (**a**) data obtained from a 54 year-old afforested stand (Ruiz-Peinado et al. 2014).(**b**) data obtained from a 90 year-old natural stand (Bravo-Oviedo et al. 2015)

Table 13.3 Forest area by regions between 1990 and 2015

Region	Forest area (1000 ha)				
	1990	2000	2005	2010	2015
Africa					
East and South Africa	319,785	300,273	291,712	282,519	274,886
North Africa	39,374	37,692	37,221	37,055	36,217
West and Central africa	346,581	332,407	325,746	318,708	313,000
Asia					
East Asia	209,198	226,815	241,841	250,504	257,047
South and Southeast Asia	319,615	298,645	296,600	295,958	292,804
West and Central Asia	39,309	40,452	42,427	42,944	43,511
Europe	994,271	1,002,302	1,004,147	1,013,572	1,015,482
America					
Caribbean	5017	5913	6341	6745	7195
Central America	26,995	23,448	22,193	21,010	20,250
North America	720,487	719,197	719,419	722,523	723,207
South America	930,814	890,817	868,611	852,133	842,011
Oceania	176,825	177,641	176,485	172,002	173,524
Global	**4,128,271**	**4,100,602**	**4,032,743**	**4,015,673**	**3,999,134**

Adapted from Keenan et al. (2015) based on FAO (1995, 2006, 2010, 2015)

and boreal forests remain stable, tropical forest area decreased and temperate forest area increased (Keenan et al., Keenan et al. 2015) Between 2000 and 2005, Brazil (3 million ha per year), Indonesia (1.8 million ha per year) and Sudan (0.6 million ha per year) suffered the largest amount of deforestation (FAO 2006). Though forest area loss was controlled between 2010 and 2015 (Keenan et al. 2015), Brazil still presents the highest forest area loss values (984,000 ha per year), followed by Indonesia (684 thousand ha per year), Myanmar (546,000 ha per year) and Nigeria (410,000 ha per year). Conserving forests by reducing deforestation and degradation, especially in threatened areas such as the tropics, is the most effective short-term strategy for carbon stock preservation and may be included in the United Nations Framework Convention on Climate Change (UNFCC) as the official mechanism of the Climate Change Agreement. Effective forest protection will facilitate carbon sequestration while adaptive management of protected areas will result in biodiversity conservation and reduced vulnerability to climate change (Nabuurs et al. 2007).

13.3.5 Increasing Forest Area

In recent decades, reforestation of marginal lands in temperate zones has increased through natural and artificial reforestation of abandoned farmland. Increased forest area in Europe in the last quarter of the twentieth century, prior to similar trends in the US, has led 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 (1.98 million ha per year), Spain (0.39 million ha per year) and Vietnam (0.42 million ha per year) dramatically increased their forest areas between 1990 and 2000 (FAO 2006). From 2000 to 2010 Vietnam and Spain slightly reduce its forest area change (0.21 and 0.12 million ha per year respectively) but China increase its change in forest area (2.99 million ha per year) China still presented the highest rate of forest expansion (1.5 million ha per year between 2010 and 2015), though less than that of previous years (Keenan et al. 2015).

Living biomass carbon stocks decreased steadily from 1990 to 2005, but by 2010 had recovered to year 2000 levels (Table 13.4) (FAO 2006, 2010). In 2010, living biomass carbon stocks increased (compared to 1990 levels) in Europe, East Asia and South America, but decreased in North Africa, West and Central Africa, South and Southeast Asia, Central America and Oceania. Finally, biomass carbon stocks

Table 13.4 Forest biomass carbon by regions between 1990 and 2010

	Biomass carbon (gigatons)			
Region	1990	2000	2005	2010
Africa				
East and South Africa	15.9	14.8	14.4	15.8
North Africa	3.8	3.5	3.4	1.7
West and Central Africa	46.0	43.9	43.1	38.3
Asia				
East Asia	7.2	8.4	9.1	8.8
South and Southeast Asia	32.3	25.5	21.8	25.2
West and Central Asia	1.6	1.7	1.7	1.7
Europe	42.0	43.1	43.9	45.0
America				
Caribbean	0.4	0.5	0.6	0.5
Central America	3.4	2.9	2.7	1.8
North America	37.2	38.5	39.2	37.3
South America	97.7	94.2	91.5	102.2
Oceania	11.6	11.4	11.4	10.5
Global	**299.2**	**288.6**	**282.7**	**288.8**

Adapted from FAO (2006, 2010)

Table 13.5 Forest plantation area by regions between 1990 and 2005

	Productive plantations (1000 ha)			Protective plantations (1000 ha)		
Region	1990	2000	2005	1990	2000	2005
Africa						
East and South Africa	2544	2712	2792	66	66	66
North Africa	6404	6158	6033	1840	2021	2192
West and Central Africa	1099	1453	1853	70	87	112
Asia						
East Asia	17,909	23,028	30,006	11,622	12,490	13,160
South and Southeast Asia	8896	10,750	11,825	3869	4451	4809
West and Central Asia	2120	2428	2583	2175	2518	2505
Europe	16,643	19,818	21,467	4569	5574	6027
America						
Caribe	239	243	280	155	151	170
Central America	51	183	240	32	29	34
North America	10,305	16,285	17,133	-	1047	986
South America	8221	10,547	11,326	10	27	31
Oceania	2447	3456	3812	1	3	21
Global	**76,826**	**97,061**	**109,352**	**24,408**	**28,464**	**30,114**

Adapted from FAO (2006)

remained fairly constant in West and Central Asia, East and South Africa, West and Central Asia and North America. Forests of South America and Africa constitute the largest carbon reservoirs; therefore, conservation of forests in these continents is crucial to mitigating climate change through forest management initiatives.

Forest plantations can be used in the Kyoto protocol for emission reduction accounting, but only in regulated circumstances and 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, productive plantation area increased from 76.8 million ha to 109.3 million ha (Table 13.5) (FAO 2006). China (26 % of global productive plantation area), 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 ha 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 ha in 1990 to 30.1 million ha in 2005 (Table 13.5). Japan (35 %) and Russia (17 %) had the most area planted for protective purposes (FAO 2006).

The effect of plantations on carbon sequestration varies according to plantation type, objectives and management, including whether the plantation is primarily intended as a productive or conservation area. Protective plantations managed for

Table 13.6 Forest area by regions between 1990 and 2010

Region	Planted forests area (1000 ha)			
	1990	2000	2005	2010
Africa				
East and South Africa	3500	3689	3813	4116
North Africa	6794	7315	7692	8091
West and Central Africa	1369	1953	2526	3203
Asia				
East Asia	55,049	67,494	80,308	90,232
South and Southeast Asia	16,531	19,736	23,364	25,552
West and Central Asia	4678	5698	5998	6991
Europe	59,046	65,312	68,502	69,318
America				
Caribbean	391	394	445	548
Central America	445	428	474	584
North America	19,645	29,438	34,867	37,529
South America	8276	10,058	11,123	13,821
Oceania	2583	3323	3851	4101
Global	**178,307**	**214,839**	**242,965**	**264,084**

FAO (2010)

conservation (through long rotation, for example) have a limited impact on carbon sequestration –since carbon sequestration rates decline in very old plantations– but can work as effectively on poor sites as short rotation does on better sites (Bravo et al., 2008). In that sense, plantations for production (e.g., for wood biomass, wood for building material, etc.) represent a better carbon mitigation strategy (Lindner and Karjalainen 2007). At each rotation, substitution with younger trees results in a net carbon emission mitigation effect.

Starting 2010 Forest Resources Assessment (FRA), no information is provided regarding classification of forest plantation areas as productive or protective (FAO 2010). The concept of planted forests (originated by afforestation or reforestation) is introduced in Table 13.6. China (1932 thousand ha per year), United States (805 thousand ha per year), Canada (385 thousand ha per year) and India (251 thousand ha per year) accounted for more than 78.64 % of the new planted forest area between 1990 and 2010 (FAO 2010).

When forest plantation projects are intended to compensate for CO_2 emissions, baseline carbon fixation (situation prior to plantation) must be compared to the expected carbon fixation from the plantation. Also, a reliable monitoring and accounting program should be developed for land within the project boundaries. Carbon monitoring and accounting programs require a large database and fitted biometric models, including volume equations, biomass expansion factors, root-shoot ratios, and other previously fitted models. In some cases, prior models or data are not locally available and substitutions must be made. Guidelines have been developed to indicate the order of priority for use of substitutions: (1) existing local and species-specific models; (2) national and species-specific models; (3) species-specific models from neighbouring countries with similar ecological conditions; or, finally (4) global species-specific models, such as those from IPCC. Uncertainties arising from these biometric models and from sampling have to be considered in accounting (Temesgen et al. 2015; Weiskittel et al. 2015); limitations that affect biomass equations include (1) high variation in sampling areas and stands, (2) data gathering in limited areas, (3) methods that rarely include belowground biomass, (4) use of simple models that do not take into consideration autocorrelation problems or mensuration of key variables and (5) loss specific estimates due to grouping of species to fit robust models. In some cases these problems are solved by including belowground biomass in the equations (Herrero et al. 2014b), including crown size (Goodman et al. 2014) as an independent variable in the models or using light detection and ranging (LiDAR) technology estimates of independent variables (Uzquiano et al. 2014). Although the best solution for biomass estimation is difficult to establish, different alternatives should be explored that improve estimations (Weiskittel et al. 2015) and that incorporate: (1) consistency in biomass data gathering across large geographical scales, (2) generation of open datasets compiling volume, biomass, carbon and wood density data, (3) use of such open datasets to evaluate and compare biomass models, (4) testing of new model forms using data from technology such as airborne and terrestrial LiDAR and (5) the application of appropriate available mathematical and statistical methods. Another source of uncertainty stems from the use of general default values for forest species when specific differences in gravity and carbon contents are widely-known (Herrero et al. 2011; Castaño and Bravo 2012) Specific differences can even occur between tissues for each species, as Herrero et al. (2011) determined for three Mediterranean pines (*Pinus nigra* Arn., *Pinus pinaster* Ait. and *Pinus sylvestris* L.) and Castaño and Bravo (2012) reported for two European oaks (*Quercus petraea* and *Quercus pyrenaica* Wild). Also, CO_2 losses due to plantation activities, such as burning of fossils fuels 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]), which must be followed in order to obtain 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) in generating revenue (Nabuurs et al. 2007). Where forest expansion leads to a reduction of agricultural land area that in turn results in intensive farming practices, the conversion of mature forests to croplands or increased agricultural imports

[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

(McCarl and Schneider 2001), will generate more emissions than potential sinks from plantations will occur globally.

13.3.6 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 in living biomass (1750 vs. 550 PgC). Forests store around 50 % of total soil carbon while representing only 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. Given the critical role of soils in overall carbon storage, conservation measures –including fire prevention and control– must be developed and implemented to conserve carbon pools in soil. Land reclamation via forest plantation on degraded land is another important carbon pool enhancement measure; plantations on formerly eroded soils can store up to 77 % more carbon (Tesfaye et al. 2016). Forest harvesting operations commonly result in short-term carbon losses from the soil (Turner and Lambert 2000). In fact, research has indicated changes in soil carbon related to management intensity (i.e. removal or maintenance of slash, soil compaction or increased radiation due to open canopies), though these 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 contains 20–35 % of total tree carbon content, is crucial for maintaining soil carbon levels.

Although carbon pools in old-growth forests are considered to be in a steady state, Zhou et al. (2006) showed that from 1973 to 2003, soil organic carbon increased at an average rate of 0.035 % each year in old-growth stands (over 400 years old) in China. These results suggest that a longer rotation length may increase soil carbon, even though living biomass accumulation may have reached an asymptote. Soil carbon maintenance was found to be compatible with sustainable forest thinning practices that did not affect soil C stock in natural and planted Scots pine (Ruiz-Peinado et al. 2016; Bravo-Oviedo et al. 2015) and afforested *P. pinaster* (Ruiz-Peinado et al. 2013) in south-west Europe.

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 previous soil carbon levels after 30 years.

13.4 Conclusions

The 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 forest area by reducing deforestation and degradation, and through increasing plantation areas or natural expansion of forest land (e.g., afforestation of abandoned lands).
2. Maintaining or increasing 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 promoting forest-based products to substitute fuel and other materials (e.g., biomass, building materials, etc.)

In the future, climate change may impact forest growth responses dramatically and modify all the scenarios analysed here. In light of such uncertainty, adaptive management holds potential for developing adequate, operational forestry strategies in a world of constant social and ecological change (Nyberg 1998). Increases in the frequenScy of both droughts and floods, or alterations in inter-annual rainfall distribution could have specific impacts. Although several climate forecasts indicate a generally positive effect on future forest growth (Sabaté et al., 2002), local drought and changes in temporal and spatial rainfall distributions may make timber production and carbon storage difficult. Greater efficiency in use of resources has been reported in some studies, such as that of Bogino and Bravo (2014) for water in Mediterranean pines in Spain. The impact of climate change on forest growth,and the interaction of climate changes with silvicultural treatments (Olivar et al. 2014), is differentiated in ecosystems across Europe (i.e., Bogino and Bravo 2008; Bogino et al. 2009; Olivar et al. 2012; Granda et al. 2014; Pretzsch et al. 2014), The combined effects of reducing deforestation and forest degradation while promoting afforestation, forest management, agro-forestry and bio-energy have the potential to increase in the future (IPPC 2007), contributing to climate change mitigation and sustainable development.

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Chapter 14
Effects of Forest Age Structure, Management and Gradual Climate Change on Carbon Sequestration and Timber Production in Finnish Boreal Forests

Jordi Garcia-Gonzalo, Ane Zubizarreta-Gerendiain, Seppo Kellomäki, and Heli Peltola

Abstract In this work, we employed two different ecosystem models in two separate case studies to assess the effects of forest age structure, management and gradual climate change on timber production and carbon sequestration of Finnish boreal forests with Norway spruce as main (dominant) tree species. Our case study examples demonstrated that over the 90 years simulation period the total timber production was from 16 to 30 % lower in southern Finland under the mild (SRES B1) and strong (SRES A2) climate change scenarios. In the northern Finland, the total timber production increased regardless of climate change scenario applied. Furthermore, both total timber production, its economic profitability and carbon stocks of forest ecosystem could be increased simultaneously if higher stocking (growing stock volume) was maintained over the rotation compared to the Finnish baseline forest management. However, timber production and carbon sequestration could not be maximised simultaneously. Thus, any preference of carbon sequestration in forest management would reduce the timber production and would result on opportunity costs.

J. Garcia-Gonzalo (✉)
Forest Research Centre, School of Agricuture, University of Lisbon,
Tapada da Ajuda, 1349-017 Lisbon, Portugal

Forest Sciences Centre of Catalonia (CTFC), Ctra de St. Llorenç de Morunys Km 2,
E25280 Solsona, Spain
e-mail: j.garcia@ctfc.es

A. Zubizarreta-Gerendiain
School of Forest Sciences, University of Eastern Finland, 111, 80101 Joensuu, Finland

Basque Centre for Climate Change (BC3), Scientific Campus of the University
of the Basque Country, Sede Building 1, 1st floor, 48008 Bilbao, Spain

S. Kellomäki • H. Peltola
School of Forest Sciences, University of Eastern Finland, 111, 80101 Joensuu, Finland

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14.1 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). In these conditions, the growth of forests is expected to be largely affected by the projected climate warming, related to the increase of greenhouse gases (GHG) in the atmosphere, particularly carbon dioxide (CO_2). In Finland, the annual mean temperature has been expected to increase up to 2–7 °C and precipitation up to 8–30 % until 2100, depending on the scenario used for the GHG concentrations (Jylhä et al. 2009). The increase in temperature may, concurrently, prolong the growing season and enhance the decomposition of soil organic matter (SOM), and the supply of nitrogen (Melillo et al. 1993; Lloyd and Taylor 1994). These changes may also substantially enhance forest production and carbon (C) sequestration in boreal forests (Giardina and Ryan 2000; Jarvis and Linder 2000; Luo et al. 2001; Strömgren 2001; Kellomäki et al. 2008). On the other hand, expected changes might differ depending on the projected climate change scenario and region concerned (Kellomäki et al. 2008).

Previous ecosystem model-based studies using business as usual management have shown an increase in forest growth, timber yield and C stocks in the boreal forests both at the stand and regional level under the changing climate compared to the current climate (e.g. Briceño-Elizondo et al. 2006a; Garcia-Gonzalo et al. 2007c; Kellomäki et al. 2008). However, the growth of Norway spruce (*Picea abies*) has been found to be very sensitive to drought extremes on soils with low water holding capacity under changing climate, unlike Scots pine (*Pinus sylvestris*) and birch (*Betula* spp.) (Briceño-Elizondo et al. 2006a; Kellomäki et al. 2008). Thus, there may be a need to modify current forest management practices under the changing climate to avoid detrimental impacts and to utilise the opportunities provided by climate change (Fürstenau et al. 2006; Garcia-Gonzalo et al. 2007a, b; Ge et al. 2013a, b; Zubizarreta-Gerendiain et al. 2014).

In addition to environmental conditions (climate, site), the age class structure of forests also affects timber yield and C stocks in managed forests (Garcia-Gonzalo et al. 2007b). Newly regenerated sites are sources of C emissions, whereas young and medium aged stands are C sinks. Old mature stands with low growth rate may again become even sources of C emissions (Jarvis et al. 2005). Both timber production and C sequestration (and stocks) are also affected by tree species and their composition (e.g. Briceño-Elizondo et al. 2006a, b; Kellomäki et al. 2008). Therefore, to maintain sustainable timber production and C sequestration over time a forest region should consist of stands with different development stages and proper tree species composition.

Empirical growth and yield models have been widely used to support practical forestry decision-making in boreal conditions (Matala et al. 2003). Usually, parameterisation of such models is based on forest inventory data, and they are capable of predicting in an accurate way the effects of forest management and site conditions on forest growth and timber yield under the current climate. Recently some empirical models have also been further developed to consider in model predictions also possible effects of gradual climate change (e.g. Matala et al. 2005; Pukkala and Kellomäki 2012).

Unlike empirical growth and yield models, forest ecosystem models (e.g. process-based models) can consider physiological processes (with time step from hour to year), as controlled by climatic and edaphic factors, such as atmospheric CO_2 concentration, temperature, and light, soil moisture and nitrogen availability (e.g. Kellomäki and Väisänen 1997; Briceño-Elizondo et al. 2006b; Ge et al. 2013a, b). In addition to them, also gap type forest ecosystem models can be used to study the sensitivity of forest growth, timber production and carbon sequestration (and stock) to changes in environmental conditions (climate, site) and management. However, their application in forestry decision-making has been limited so far since they have often required input data not typically provided by conventional forest inventories. However, such models can provide the same prediction capacity under practical management situations as empirical growth and yield models (Matala et al. 2003; Routa 2011). Moreover, such models may help to define proper adaptive management strategies needed under the changing climate (Lindner 2000; Pukkala and Kellomäki 2012; Garcia-Gonzalo et al. 2014).

In this work, we employed two different ecosystem models in two separate case studies to assess the effects of forest age structure, management and gradual climate change on timber production, its profitability, and carbon sequestration of Finnish boreal forests with Norway spruce as main (dominant) tree species.

14.2 Material and Methods

14.2.1 Case Study 1: Sensitivity of Timber Production and Carbon Sequestration to Forest Age Class Distribution and Climate Change from Southern to Northern Finland

Outlines for the SIMA Model In this case study, we used in model based analyses the gap-type forest ecosystem model SIMA, in which the regeneration, growth and mortality of trees control the dynamics of tree populations and the consequent uptake and emissions of CO_2 (Kellomäki et al. 1992a, b, 2005, 2008). The growth of trees is based on diameter growth; i.e. $\Delta D = \Delta D_o \times M_1 \times, \ldots, \times M_n$, where ΔD is diameter growth [cm yearr.$^{-1}$]; ΔD_o is diameter growth [cm a^{-1}] in the optimal conditions; and $M_1, \ldots, M_n$ are multipliers representing the temperature sum (TS; +5 °C threshold), prevailing light conditions, soil moisture and nitrogen supply. Optimal conditions refer to growth under no shading and no limitation of soil moisture and nitrogen supply. The values of ΔD_o are further related to maturity of the tree (diameter of tree, D cm) and the atmospheric CO_2. In the model, trees may die for random reasons and for the competition between trees with a consequent reduction in growth, which affects the risk of a tree to die in a given year. Litter from any tree organ and dead trees transfer carbon and nitrogen in soil, where litter is decayed, and finally nitrogen released for the use of growth. Nitrogen supply is further related to the amount of litter and humus specific for each site fertility type. In general, fertile sites with fine soil texture represent higher productivity than medium fertile sites and poor sites with coarse soil texture.

In the model, management options include natural regeneration and/or planting with desired spacing and tree species, control of stand density in pre-commercial and commercial thinnings, nitrogen fertilisation, and choice of rotation length based on tree age or basal area weighted mean diameter of trees at breast height (1.3 m above stem base, DBH). In harvesting, both timber (saw log and pulp wood) and energy biomass (foliage, branches, stumps and small sized tree stem tops) can be considered. Simulations are carried out on an area of 100 m^2 using a one-year time step based on the Monte Carlo technique, because certain events (e.g. birth, death) are stochastic. Therefore, each simulation scenario is repeated several times (e.g. 20–50 times) and the mean tendency of output variables are used in data analyses.

The model has been parameterised for Norway spruce, Scots pine and silver birch and downy birch (*Betula* ssp.) growing throughout Finland (the latitudes N 60° and N 70° and longitudes E 20° and E 32°). The previous simulations with the model have shown a good agreement with the measured values of the volume growth of these species on the permanent sample plots of the National Forest Inventory throughout Finland. Furthermore, the parallel simulations with the empirical growth and yield model Motti (Finnish Forest Research Institute) and the SIMA model have provided good agreement for the predicted volume growth of Finnish forests (e.g. Kellomäki et al. 2008; Routa et al. 2011a, b).

Layout for the Simulations For the simulations, three model forest areas, with 1000 ha each, were generated for southern (Helsinki, 60° N), central (Kuopio, 62° N) and northern Finland (Rovaniemi, 66° N). They consisted of representative Norway spruce stands of different stand ages, i.e. ranging from seedling stands to mature stands with age range from 10 to 90 years on medium fertile sites (*Myrtillus*, MT). The development of each Norway spruce stand (1 ha each) was simulated over 90 years (period 2010–2100). As inputs for model simulations, the location of study site, site fertility, tree species, tree height, diameter at breast height and stand density were used in addition to climate scenario applied. For thinnings and final cut the current species-specific Finnish business-as-usual management recommendations were used (see Äijälä et al. 2014). Dominant height (<12 m) and basal area of stands were used to define the timing and intensity of thinning (Fig. 14.1), which means that the timing of thinning is adjusted for the growth and development of the stand. All thinnings were done from below, mainly removing smaller trees. After clear-cutting at age of 70 years, the site was planted with an initial density of 2500 saplings ha^{-1}.

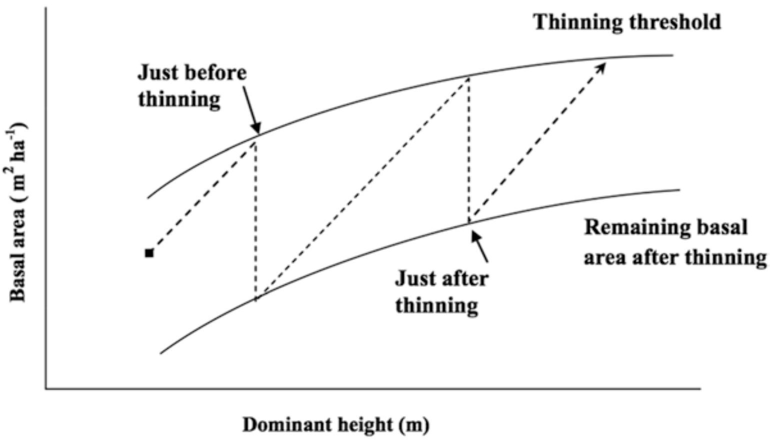

Fig. 14.1 Principles used for thinning: dominant height and basal area together determines the timing and intensity of thinning

In addition to current climate, the two recent climate change scenarios were used, i.e. the SRES B1 and A2 for the period of 2010–2099. The climate data used consisted monthly means for temperature and precipitation (Venäläinen et al. 2005; Jylhä et al. 2009). The climate change projections were derived from nineteen global change model simulations originating from the CMIP3 multimodel dataset (Meehl et al. 2007). According to the SRES B1 projections, in Finland the temperature (T) increased 4–5 °C and 2 °C and precipitation up to 17 and 11 % in winter and summer until 2100 while in the SRES A2 scenario, T increases 6–8 °C and 3 °C and precipitation up to 32 % and 14 % in winter and summer. Expected changes are in relative sense higher in northern than in southern Finland. Compared to the current CO2 concentration and greenhouse gas emissions (360 ppm), the climate change scenario SRES B1 implies a lower increase (up to 540 ppm) than the SRES A2 climate change scenario (up to 840 ppm) by 2100.

Data Analyses Based on these representative stand simulations, we analysed the effects of forest age class structure and climate change projection on total timber yield (saw logs and pulp wood, $m^3\ ha^{-1}$) and annual mean C stocks ($Mg\ C\ ha^{-1}$) in the model forests over the 90 year simulation period, considering four alternative age class distributions (distributions A, B, C, D). Following age classes were used in data analyses: (1) sapling stands, (2) stands at first thinning age, (3) stands at later at later thinning age, (4) stands at final cutting age, Following model age class distributions were created (Table 14.1): 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).

Table 14.1 Percentage of area occupied by each of the age class groups in each age class distribution. Descriptions of the cases are given in the text

Age class groups	Age	Age class distribution			
		A (Normal) (%)	B (Uniform) (%)	C (Left) (%)	D (Right) (%)
Sapling stands	0–10	12.5	12.5	25	5
	11–20	12.5	12.5	25	5
Stands at first thinning age	21–30	15	12.5	12.5	7.5
	31–40	15	12.5	12.5	7.5
Stands at later thinning age	41–50	10	8.3	5	8.3
	51–60	10	8.3	5	8.3
	61–70	10	8.3	5	8.3
Stands at final cutting age	>70 year	15	25	10	50

14.2.2 Case Study 2: Sensitivity of Timber Production and Carbon Sequestration to the Forest Age Structure, Forest Management and Climate Change in Central Finland

Outlines for the FINNFOR Model In this case study, we used a physiological model, where the dynamics of the forest ecosystem are linked to climate and soil through photosynthesis, respiration and transpiration (Kellomäki and Väisänen 1997). The gross photosynthesis over a year provides the total amount of photosynthates available for the maintenance and growth of trees and tree organs (foliage, branches, stem, coarse and, fine roots). Photosynthesis is controlled by climatic factors (radiation, temperature, humidity and atmospheric CO_2) and edaphic factors (moisture, temperature, nitrogen supply) directly and indirectly through stomatal functions. Dynamics of the model ecosystem is further linked to the climatic change through the thermal and hydraulic conditions in soil, and the decomposition of litter and humus with the mineralization of nitrogen.

The allocation of photosynthates among organs is based on the allometry between the mass of each organ and the total mass of tree (Matala et al. 2003). The stem mass and its growth are used to annually calculate DBH and height of object tree representing each tree cohort. Cohorts are defined by tree species, number of trees per hectare, diameter (DBH, cm), height (m) and age (year). Management affects growth through climatic (microclimate in canopy) and edaphic factors controlling photosynthesis and respiration. Management includes regeneration (planting), thinning, and selection of the rotation length. The timing and intensity of thinning can be controlled via basal area reduction at certain dominant height (or stand age), as converted into the number of trees to be removed from each tree cohort. Trees removed in thinning and in final cut are converted to saw logs, pulp wood and logging residues.

Model parameterization based on data from long-term forest ecosystem and climate change experiments (see Kellomäki et al. 2000) and model validation against (i) growth and yield tables (Kellomäki and Väisänen (1997) and parallel simulations with the empirical growth and yield model MOTTI (Hynynen et al. 2002; see Matala et al. 2003; Briceño-Elizondo et al. 2006a), (ii) 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 (Kramer et al. 2002) and (iii) measurements of the growth history of trees in thinning experiments (Matala et al. 2003) have been discussed in detail in many previous studies. Sensitivity analyses of outputs (e.g. growth, timber yield and C stocks) to different climate parameters and to the initial structure of the forest have also been used in analysing the model performance (Briceño-Elizondo et al. 2006a; Garcia-Gonzalo et al. 2007a, b, and c). Based on these studies, the FINNFOR model performs in an acceptable way and it has similar capacity to simulate the growth and development of Scots pine, Norway spruce and birch stands as the empirical MOTTI model.

Layout for the Simulations The simulations were carried out using as a baseline forest inventory data available for the forest management unit located in central Finland (63°01′ N). The study area included 1018 separate stands, covering 1451 ha (based on 2001 inventory). The breakdown of stands was: Norway spruce dominated stands, 64 % of the total area (933 ha); Scots pine dominated stands, 28 % (412 ha); and silver birch dominated stands, 7 % (106 ha). The site fertility varied from fertile to poor sites: 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. For each stand data was available about dominant tree species, average stand age, mean height and mean DBH, stand density and site fertility type.

Since simulation time for 1018 stands under different management regimes and forest structures would have been very time consuming, representative stands were used instead for simulations. For this purpose, all the stands were first classified into groups representing the same dominant tree species, age class (10 years intervals; 0–100 years) and site fertility type. Then, a typical stand was selected from each group. In total, 42 representative stands were selected for the simulations over 100 years simulation time, representing different phases of stand development from seedling stands to mature stands. In each representative stand, the number of trees was divided uniformly 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}.

In this case study, we used a species specific business-as usual management (BT (0,0)), and two other thinning regimes in which the thinning threshold and the remaining basal area after thinning was increased by either 15 % (BT(15,15)) or 30 % (BT30,30)) compared to BT(0,0). Also an unthinned (UT(0,0)) regime was used as a reference. Regardless of tree species and site fertility type, the stand was clear-cut when the average DBH of the trees exceeded 30 cm (or at the latest at age of 100 years). After clear cut, the site was planted with the same species (2500 saplings ha^{-1}) that occupied the site prior to clear-cut.

In addition to the current climate (period of 1961–1990), we used the HadCM2 climate change scenario (2000–2099) compiled by the Potsdam Institute for Climate Impact Research (PIK), Germany, following the model predictions derived from the Hadley Centre Global Circulation Model (GCM) (Erhard et al. 2001; Sabaté et al. 2002). For FINNFOR simulations, the daily weather statistics were also needed to be scaled down to an hourly basis, using the weather simulator developed by Strandman et al. (1993). Under the current climate, the annual mean temperature and precipitation were 3.1 °C and 478 mm $year^{-1}$. Under the HadCM2 scenario, the annual mean temperature and precipitation increased gradually up to 7.2 °C and 560 mm in the period of 2070–2099. Under the current climate, the CO_2 concentration used was 350 ppm while the HadCM2 scenario represents a gradual increase up to 653 ppm over the period 2000–2100.

Data Analyses Based on the simulations with representative stands, we analysed the effects of forest age class structure, management regime and climate change projection on total timber production (m^3 ha^{-1}) and its economic profitability (net present value, NPV) and annual mean C stocks (Mg C ha^{-1}) in the forest management unit over 100 year simulation period. Furthermore, we calculated the opportunity costs for preferring C sequestration versus timber production. The age class distributions used were same as in case study 1 (see Table 14.1).

In the calculation of NPV, all the incomes and costs (e.g. planting and other regeneration costs) over 100 years simulation period were considered 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 %. In these calculations, the prices used for different timber assortments were the average stumpage prices (w f^{-3}) for the period 1990–2000 (Finnish Statistical Yearbook of Forestry 2001). Similarly, as the costs of the regeneration operations (soil preparation and plantation) the average prices for the period 1990–2000 for Finnish conditions we applied. All these results were calculated over 100 year simulation period, considering alternative forest age class structures, management regimes and climate scenarios in the management unit. In calculation of the opportunity costs for timber production for preferring C sequestration, carbon stock of wood based products were not considered. The potential opportunity (marginal) cost of increasing C stock (potMC) refers in this work to the differences in the C stock and in NPV of timber production for those management regimes, which either maximise the carbon stock or NPV, respectively. On the other hand, the current marginal cost (curMC) refers the differences in the C stock and NPV of timber production when one shifts from the current management to that which maximises the C stock in the forest ecosystem.

14.3 Results

14.3.1 Case Study 1: Sensitivity of Timber Production and Carbon Sequestration to the Forest Age Class Distribution and Climate Change from Southern to Northern Finland in a Period of 90 Years

Under the current climate the total timber production over 90 years simulation period is the highest in the south and the smallest in the north, regardless of the age class distribution (Table 14.2). Climate change affects differently depending on the region analysed. For example, the total timber production was from 16 to 30 % lower in southern finland under the SRES B1 and SRES A2 climate change scenarios compared to the current climate, regardless of forest age class structure (Fig. 14.2). The highest difference between current climate and the gradually changing climate scenarios were in the last 30-years period simulated. In central Finland, climate change had a minor effect on total timber production (decrease of 2–5 % observed compared to current climate). In northern Finland, the total timber production increased clearly under the gradually changing climate compared to the current climate, i.e. up to 22–26 % under the SRES B1 and SRES A2 climate change

Table 14.2 Total amount of harvested timber yield ($m^3\ ha^{-1}$) over three following 30 year time periods for different age class distributions in Norway spruce forests with the business as usual management, under the current climate and gradually changing climate (SRES A2 and SRES B1 scenarios) from southern to northern Finland. The four age class distributions used were: A (normal distribution); B (uniform); C (left-skewed) and D (right-skewed)

	Current climate (CRU)				Climate change (A2)				Climate change (B1)			
	A	B	C	D	A	B	C	D	A	B	C	D
Southern Finland												
2040	157	163	104	213	153	159	103	207	151	158	101	208
2070	204	200	241	164	161	150	187	105	159	148	187	103
2100	173	185	156	224	60	71	63	96	130	148	121	196
Total	534	547	501	601	374	381	353	408	440	455	409	507
Central Finland												
2040	160	164	107	208	158	163	107	208	158	163	107	206
2070	196	192	233	156	198	193	231	158	195	191	230	155
2100	179	189	163	224	151	163	133	204	166	178	153	217
Total	535	545	503	587	506	519	471	570	520	532	490	578
Northern Finland												
2040	127	132	86	170	129	134	87	171	135	140	100	174
2070	140	138	168	115	171	165	197	133	161	156	182	129
2100	134	138	111	164	191	200	174	233	186	194	165	226
Total	401	408	365	448	492	499	458	538	481	489	446	530

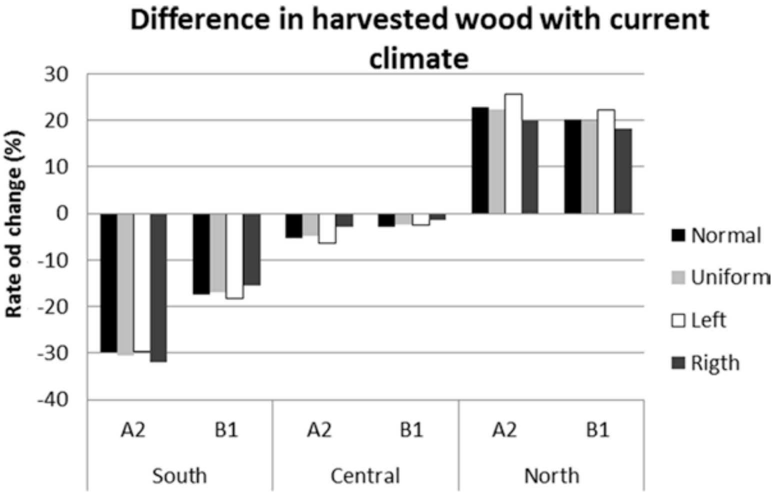

Fig. 14.2 Relative (%) differences in amount of harvested timber in a period of 90 years for different climate change scenarios (SRES A2 and SRES B1) compared to the current climate from southern to northern Finland for different age class distributions (normal distribution, uniform, skewed to the *left*, skewed to the *right*)

scenarios. Regardless of the climate and region considered, the initially right skewed age class distribution (more mature stands) provided the highest total amount of harvested timber over the 90 years period, followed by the uniform forest age class distribution.

Under the current climate, the annual average ecosystem carbon stock per hectare over 90 years simulation period was not largely affected by the forest age class distribution or study region. Initially normal age class distribution resulted in slightly higher C stock, while the distribution skewed to the right showed the lowest values (Table 14.3) In southern Finland, the average C stock in soil was lower under the gradually changing climate scenarios than under the current climate, regardless of forest age class distribution (Fig. 14.3). Average C stock in trees was also in general lower under the SRES A2 climate scenario. Under the SRES B1 climate change scenario only left skewed age class distribution resulted in lower average C stock in trees compared to the current climate. In central and northern Finland, the average carbon stock in trees increased under changing climate compared to the current climate. The increase was considerably high in northern Finland. Also average carbon stock in soil increased in northern Finland, for all age class distributions, unlike in central Finland (Fig. 14.2).

Table 14.3 Annual average total carbon stock (C) in the ecosystem (Ce, Mg ha^{-1}), in trees (Ct, Mg ha^{-1}) and in soil (Cs, Mg ha^{-1}) from southern to northern Finland over 90 years, for different age class distributions: A (normal distribution); B (uniform); C (skewed to the left) and D (skewed to the right)

	Initial age class distribution											
	A			B			C			D		
Location	Current climate											
	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct
South Finland	75	38	37	74	39	35	74	38	36	73	39	34
Central Finland	79	41	38	78	41	37	77	39	38	76	41	35
North Finland	75	43	32	73	42	31	73	41	32	72	43	29
	Climate change (A2)											
	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct
South Finland	72	38	34	70	38	32	66	35	31	70	39	31
Central Finland	84	41	43	81	40	41	77	38	39	80	40	39
North Finland	93	48	45	90	48	42	85	44	41	88	47	41
	Climate change (B1)											
	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct
South Finland	77	38	39	75	38	37	71	36	35	74	39	35
Central Finland	86	42	44	83	41	42	79	39	40	81	41	40
North Finland	92	49	43	89	48	41	83	44	39	88	48	40

14.3.2 Case Study 2: Sensitivity of Timber Production and Carbon Sequestration to the Forest Age Structure, Forest Management and Climate Change in Central Finland in a Period of 100 Years

Total timber yield per hectare as well as the NPV over the 100-year simulation period were affected by the initial age class distribution, the thinning regime and climate used. However, regardless of the thinning regime, under the current climate, in general, the largest timber yield and NPV were obtained when the initial forest was dominated by old stands mature for regeneration (right-skewed distribution), followed by the forest with the uniform and forest with a normal distribution, respectively. The distribution skewed to the left (forest dominated by young stands) gave the smallest timber yield and NPV, respectively (Table 14.4).

In general, timber yield increased if thinning was delayed and the remaining basal area after thinning was kept higher compared to business as usual BT(0,0), which was the case in BT(15,15) and BT(30,30). Under the current climate, the largest timber yield was obtained with BT(30,30). The highest NPV with 3–% discount rate (7206 € ha^{-1}) was obtained with BT (0,0) regime for right-skewed age class distribution (Table 14.4). The NPV was clearly lower for a uniform age class distribution (5507 € ha^{-1}), normal age class distribution (5068 € ha^{-1}) and left-skewed age class distribution (3887 € ha^{-1}). Regardless of thinning regime applied, initial age class distribution affected in a similar way.

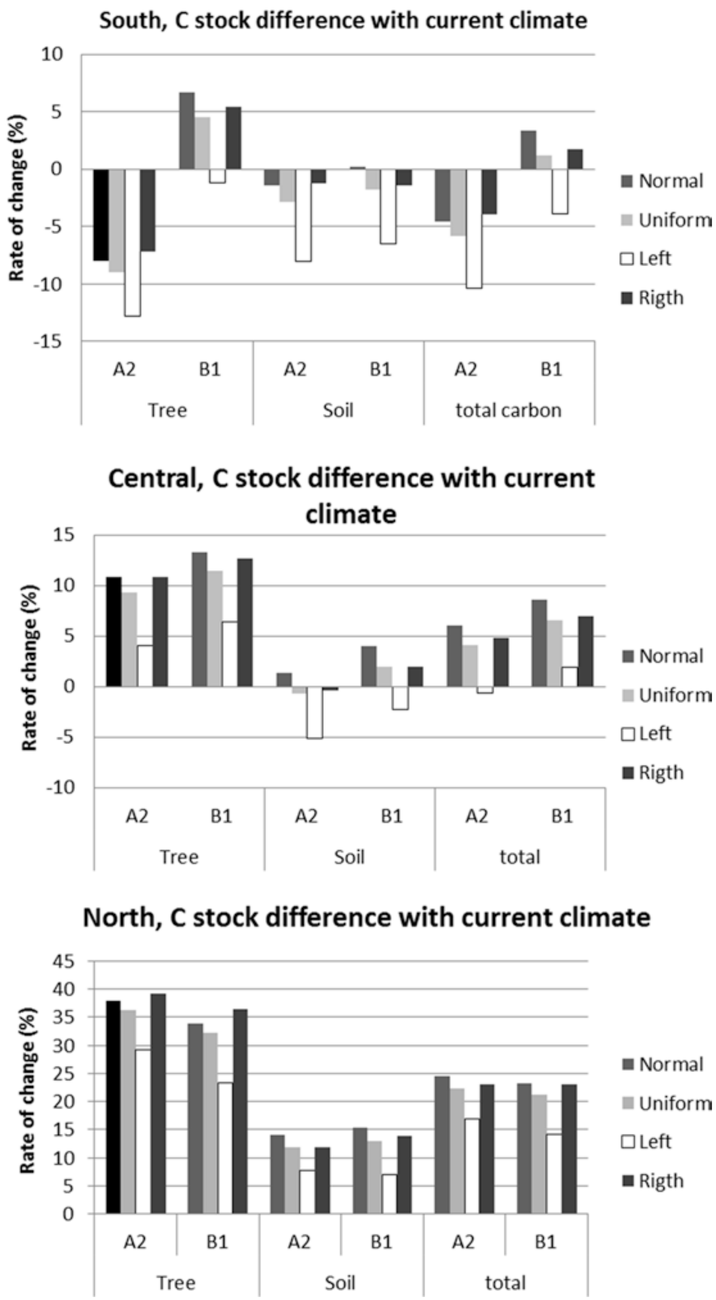

Fig. 14.3 Relative (%) differences in average annual mean C stock over 90 years for different climate change scenarios (SRES A2 and SRES B1) compared to the current climate from southern to northern Finland for different age class distributions (normal distribution, uniform, skewed to the *left*, skewed to the *right*)

Table 14.4 Total amount of harvested timber ($m^3\ ha^{-1}$) and its net present value (NPV, € ha^{-1}) with discount rate of 3 % over a period of 100 years, for different age class distributions (A, B, C or D) and different thinning regimes under the current and changing climate. The four age class distributions used were: A (normal distribution); B (uniform); C (left-skewed) and D (right-skewed)

Management scenarios	Initial age class distribution							
	A	B	C	D	A	B	C	D
	Current climate							
	Harvested timber ($m^3\ ha^{-1}$)				NPV (€ ha^{-1})			
UT(0,0)	484	469	491	432	3884	4401	2807	6248
BT (0,0)	598	613	562	669	5068	5507	3887	7206
BT(15,15)	619	638	592	696	5191	5619	4093	7273
BT(30,30)	650	669	628	725	5214	5646	4136	7292
	Climate change (HadCM2)							
	Harvested timber ($m^3\ ha^{-1}$)				NPV (€ ha^{-1})			
UT(0,0)	546	535	567	499	4331	4841	3186	6685
BT (0,0)	684	692	632	742	5752	6168	4548	7819
BT(15,15)	724	731	674	778	5840	6265	4707	7964
BT(30,30)	762	769	716	814	5944	6352	4798	8021

Under the changing climate, the effects of age class distribution and thinning regime on the timber yield and NPV were similar to those under the current climate. However, the timber yield increased up to 11–17 % depending on the thinning regime and the forest age structure compared to the current climate (Fig. 14.4). The highest relative increase in timber yield was found for the normal age class distribution and the lowest for the right-skewed age class distribution in thinned stands. Any increase in timber yield increased also the NPV, respectively (Fig. 14.4). The highest relative increase of NPV was found for the left-skewed age class distribution and the lowest for the right-skewed age class distribution, regardless of thinning regime used. However, the discount rate used had a strong influence on the NPV. In the case of discount rate of 5 % (not shown values), the NPV for BT(0,0) was even higher than those for other management regimes.

The flows of timber yield and NPV were largely affected by the gradual change of forest age class structure of forests over time, regardless of initial forest structure and climate conditions considered. The normal and uniform age class distributions gave the most even timber harvest over time, whereas the initial age class distribution skewed to the left resulted in highest timber flows towards the end of the simulation period, opposite to the age class distribution to the right. At the end of the 100-year simulation period, all the age class distributions became skewed to the left under the current climate. In addition, also thinning regimes and climatic conditions applied affected the growth rate of trees and thus, the timing of final cut (based on average DBH), and the age structure of forests (Table 14.5). Thinnings and final cuts were done slightly earlier under the changing climate. At the end of the 100 years simulation period, all age class distributions under the climate change resembled the normal distribution.

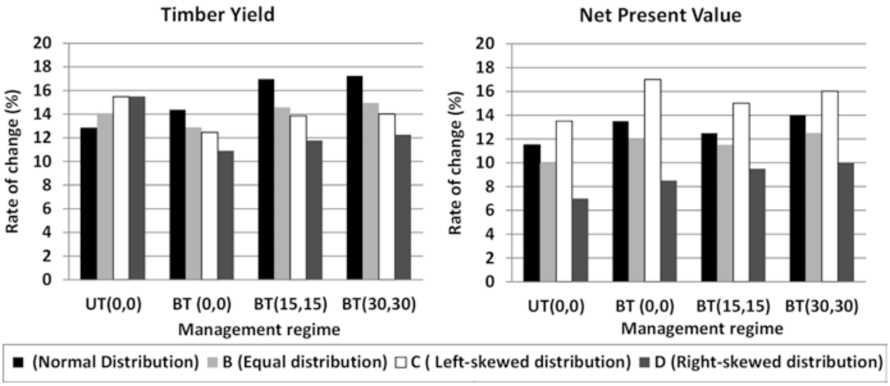

Fig. 14.4 Effect of climate change (HadCM2) on timber yield and NPV of harvested timber for different management regimes and age class distributions

Table 14.5 Percentage of area occupied by each of age class over time for different initial age class distribution used (A, B, C or D) when the business-as-usual management regime is used (BT(0,0))

	Year 2001				Year 2060				Year 2100			
	Age				Age				Age			
Initial age class distribution	1–20	21–40	41–70	>70	1–20	21–40	4–170	>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

Over the whole management unit and 100-year simulation period, the annual average ecosystem C stock per hectare was similar, regardless of initial 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 14.6). Without thinning (UT(0,0) it was obtained the highest average ecosystem carbon stock per hectare, being about 45 % higher than with the BT(0,0) regime. However, by applying thinning regimes which maintained on average higher stocking over rotation compared to the BT (0,0), also the C stock increased (i.e., BT(15,15) and BT(30,30) compared to the BT(0,0) (Table 14.6). Under the changing climate, the effect of age class distribution and the thinning regime were similar to the current climate. However, the climate change slightly increased the annual average ecosystem carbon stock of forests regardless of forest age structure applied (Table 14.6).

Table 14.6 Annual average total carbon stock (C) in the whole ecosystem (Ce, Mg ha^{-1}), separately in trees (Ct, Mg ha^{-1}) and in soil (Cs, Mg ha^{-1}) for different age class distributions over a period of 100 years: A (normal distribution); B (uniform); C (skewed to the left) and D (skewed to the right) (the difference between them corresponding to C in soil), depending on the age class distributions: A (normal distribution); B (uniform); C (skewed to the left) and D (skewed to the right)

Management scenarios	Initial age class distribution											
	A			B			C			D		
	Current climate											
	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct
UT(0,0)	145	77	68	146	77	69	145	77	68	147	76	71
BT (0,0)	102	63	39	102	63	39	100	62	38	104	64	40
BT(15,15)	108	65	43	108	65	43	106	64	42	110	66	44
BT(30,30)	114	67	47	114	67	47	111	65	46	115	67	48
	Climate change											
	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct	Ce	Cs	Ct
UT(0,0)	153	76	77	154	76	78	154	77	77	156	75	81
BT (0,0)	105	62	43	104	61	43	103	61	42	105	63	42
BT(15,15)	110	63	47	110	64	46	108	63	45	110	64	46
BT(30,30)	116	66	50	115	65	50	113	64	49	116	66	50

Management regimes and climate change affected also both C stocks in soil and in trees and in a similar manner, but the effects were higher on C in trees (Table 14.6). Climate change affected in different ways these C pools. Under the changing climate, the C stock in trees increased, opposite to that in soil (Fig. 14.5). Also, climate change affected C stocks differently depending on the initial age class distribution and the thinning regimes applied (Fig. 14.5).

In general, regardless of the climate applied and initial forest age class structure used, the highest timber yield over 100 years was observed under the BT(30,30) regime (Table 14.4), whereas the annual average ecosystem C stocks was the highest without thinning (UT(0,0)) (Table 14.6). Therefore, it was not possible to simultaneously achieve using same thinning regime both the highest total timber yield and ecosystem C stock in the forest area over 100 years simulation period. However, our results showed that it is possible to increase both annual average C stocks in the forest area and total timber yield using thinning regimes maintaining higher stocking of the rotation compared to the BT(0,0).

Under the current climate, the difference in annual average ecosystem C stocks per ha^{-1} between BT(30,30) and UT(0,0) ranged between 31 Mg C ha^{-1} and 34 Mg C ha^{-1} depending the initial age class distribution. The preference of the UT(0,0) regime instead of the BT(30,30) would result in the potential marginal cost (**potMC**), with a range from 33.1 € Mg ha^{-1} for the right-skewed age class distribution to 42.8 € Mg ha^{-1} for the normal age class distribution with NPV at discount rate of 3 % (Table 14.7). If higher discount rate is used, the marginal cost for carbon sequestration decrease.

Under the changing climate, the annual average ecosystem C stored in forest area is higher as are the total timber yield and its NPV, compared to the current climate. Thus, the cost of enhancing C stock is similar than under current climatic conditions, ranging between 32.2 and 42.8 € Mg C^{-1} (p = 3 %), depending of the initial age class distribution (see Table 14.7). However, the relative effects of age class distribution are similar to the current climate.

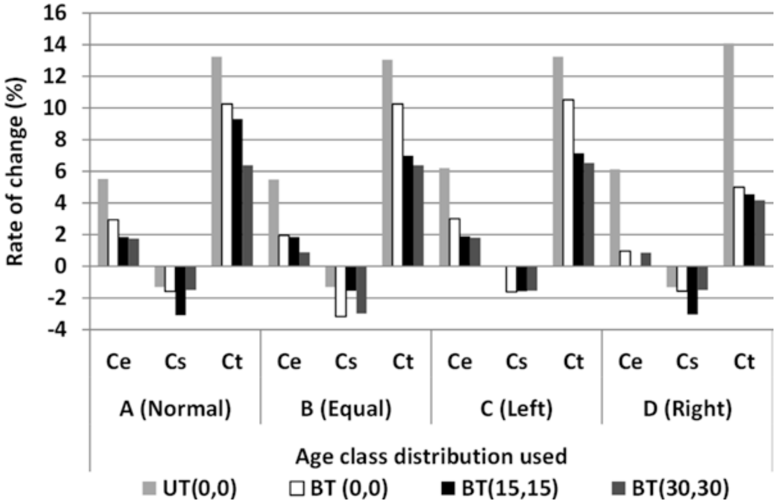

Fig. 14.5 Relative effect of climate change on the C stocks (Cs: C in soil, Ct: C in trees and Ce: C in total ecosystem) for different thinning regimes, under different age classes distributions (A, B, C, D)

Table 14.7 Potential[a] (potMC) and current[b] (curMC) marginal cost of annual average ecosystem carbon sequestration by sink enhancement in the forest area (€ Mg C^{-1}) depending on the initial 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 %

Initial age class distribution	PotMC			CurMC		
	1 %	3 %	5 %	1 %	3 %	5 %
	Current climate					
A (Normal)	54.1	42.8	28.7	10.5	27.7	26.1
B (uniform)	50.9	39.1	25.5	9.8	25.5	23.5
C (left)	54.4	38.9	23.5	14.3	23.7	20.3
D (right)	43	33.1	21.6	−0.1	22.3	21.5
	Climate change					
A (Normal)	49.6	42.8	28.2	7.1	29	25.3
B (uniform)	45.8	39	24.8	6.4	26.8	22.7
C (left)	51.3	40.2	23.8	12.6	27.7	20.7
D (right)	36.2	32.2	20.2	−6.7	21.7	19.8

[a]Potential cost derives 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 derives from de differences in sequestered carbon and generated incomes from timber production of the best alternative for carbon sequestration and the current management used

Our results showed also that shifting from current management BT(0,0) to the one that maximize 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 cost of 27.7 € Mg C^{-1} (p = 3 %, Table 14.7). Possible increases in the ecosystem carbon sinks were higher under the climate change than under the current climate. However, by shifting from use of BT(0,0) towards BT(30,30) allows to maintain higher ecosystem C stock without any loss of NPV. In absolute terms, their difference was up to 10–12 Mg C ha^{-1} over 100 years, depending on the initial age class and species distribution used.

14.4 Discussion and Conclusions

14.4.1 Evaluation of Main Findings

The environmental conditions (climate, site), management and structure of forest (age structure, species composition) affect together the productivity (growth, timber yield) and carbon sequestration of forests. In this work, we employed two different ecosystem models in two separate case studies to assess the effects of forest age structure, management and gradual climate change on timber production, its profitability, and carbon sequestration (C stocks) of Finnish boreal forests with Norway spruce as main (dominant) tree species.

Based on our findings, timber harvests were fairly even over time when the initially normal and uniform age class distributions were used. The initially left-skewed distribution (dominated by young stands) had higher later harvests, opposite to the initially right-skewed one. In the latter case, the NPV (with discount rates of 1–5 %) of the timber harvest was the highest due to early incomes, respectively. Thus, regardless management regime and region used, the initial age class distribution affected largely the total timber yield, NPV and average C stocks of forest area over time.

Based on previous studies, in southern Finland high increase in temperature will enhance the growth and success of Scots pine and silver birch, opposite to Norway spruce, which may suffer drought especially on sites with low water holding capacity (Briceño-Elizondo et al. 2006a; Kellomäki et al. 2008). Also in our work, we observed a reduced timber yield and carbon stocks in Norway spruce forests under gradually changing climate (especially under SRES A2) in southern Finland, opposite to northern Finland. The differences in relative sensitivity to climate change in central Finland in different case studies could be explained at least partly by differences in climate change scenarios, i.e. climate change was more drastic in the first case study.

Furthermore, regardless of climatic conditions and initial age class distribution used, the highest annual average C stock and the lowest NPV were observed without thinning, as has been found previously also in several studies (Deward and Cannell 1992; Karjalainen 1996; Thornley and Cannell 2000; Fürstenau et al. 2006).

However, the C stocks in the forest ecosystem may be maintained in higher level compared to the business as usual management if higher stocking is preferred over rotation. However, C stocks are also affected by the stand structure (age, species composition), region and properties of the climate and site (Mäkipää et al. 1998, 1999, Vucetich et al. 2000; Pussinen et al. 2002).

In central Finland (case study 2), we observed an increase in C stocks regardless of management regime used under the changing climate due to an increase of growth rate, as was suggested previously also by Karjalainen et al. (1999, 2003) and Mäkipää et al. (1999). On the other hand, we found that the C in the soil decreased under the changing climate compared to the current climate. This was due partly to higher temperatures, which decreased the growth and litter production and enhanced the decomposition of SOM as also observed in other studies (e.g. Grace 2001, 2005; Karjalainen et al. 1999, 2003). This was not in northern Finland (case study 1). In northern Finland, carbon in soil and in trees increased considerably under changing climate, regardless of climate change scenario and age class distribution due to higher water availability and length of growing season. This was opposite to the southern Finland under the SRES A2 climate change scenario, since tree growth in the south was reduced due to increase of drought effects. Based on our findings, in managed boreal forests both timber production and C sequestration may be increased by applying management in which higher stocking is maintained over rotation compared to that of business-as-usual management.

14.4.2 Conclusions

Our forest ecosystem model simulations showed that both timber yield and C stocks of forests are affected largely by forest structure, management and climatic conditions (climate, region). Furthermore, the preference of high C stocks of forests in management may induce opportunity costs for timber production. Our results also showed that initial age class structure of the forest affects largely the flows of timber yield and C stocks over time. Depending on the projected climate change used, the responses observed may also be even opposite as we showed in this work, i.e. revealing the uncertainties related to climate change and its effects. Also the frequency and magnitude of natural disturbances (fire, storm, drought, insects) may change under the changing climate, which were excluded from our analyses. We also assumed that after the final harvest, stands would be planted with the species that were present prior to final cutting. In practice, the choice of species in regeneration and subsequent management may change from those applied in the past. Regarding C sequestration, this study focused solely on C stocks in the forest ecosystem, therefore excluding carbon stock in forest products. In addition to timber harvest, there is also a need to explicitly address other goals for forest management 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 optimisation routine would be needed to simulate a mixture of management preferences over a forest area to determine how forest management could be optimally adapted to meet goals in the context of a changing climate.

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Part V
Case Studies

Chapter 15
Mediterranean Pine Forests: Management Effects on Carbon Stocks

Miren del Río, Ignacio Barbeito, Andrés Bravo-Oviedo, Rafael Calama, Isabel Cañellas, Celia Herrero, Gregorio Montero, Dianel Moreno-Fernández, Ricardo Ruiz-Peinado, and Felipe Bravo

15.1 Introduction

Carbon stored in forest systems is of great interest from a management point of view since, on the one hand, it is easily modified through silvilcultural practices (e.g., rotation length, thinning, etc.), while, on the other hand, it affects the mean lifespan of wood products. In the Mediterranean area, the role of forest as carbon sinks is particularly significant since usually ecosystem services provided by forests are frequently of greater value than their direct productions. Therefore, quantifying the

M. del Río (✉) • A. Bravo-Oviedo • R. Calama • G. Montero
D. Moreno-Fernández • R. Ruiz-Peinado
Department of Sylviculture and Management of Forest Systems,
INIA-Forest Research Centre and Sustainable Forest Management Research Institute,
Universidad de Valladolid & INIA, Madrid, Spain
e-mail: delrio@inia.es; bravo@inia.es; rcalama@inia.es; montero@inia.es;
daniel.moreno@inia.es; ruizpein@inia.es

I. Barbeito
Laboratoire d'Etude des Ressources Forêt Bois (LERFoB), INRA centre of Nancy,
Champenoux, France
e-mail: ignacio.barbeito-sanchez@inra.fr

I. Cañellas
Joint Research Unit INIA-UVa, Department of Forest Systems and Resources,
CIFOR-INIA, Madrid, Spain
e-mail: canellas@inia.es

C. Herrero
Joint Research Unit INIA-UVa, Department of Forest Resources, Universidad de Valladolid,
Palencia, Spain
e-mail: chdeaza@pvs.uva.es

F. Bravo
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest
Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain
e-mail: fbravo@pvs.uva.es

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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).

Aboveground biomass is usually estimated from forest inventories through biomass equations and biomass expansion factors at different spatial scales (e.g. Schroeder et al. 1997; Somogyi et al. 2007; Teobaldelli et al. 2009; Castedo-Dorado et al. 2012; Fang et al. 2014), whereas belowground biomass is often indirectly estimated from the aboveground biomass (Cairns et al. 1997; Mokany et al. 2006). Information is for other biomass components such as litter, dead organic matter or for soil carbon less abundant, 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 is notably enhanced over the last decade (e.g., Jandl et al. 2007; Woodall et al. 2008; Nave et al. 2010; Vesterdal et al. 2013).

A good understanding of the effect of forest management on the C cycle is crucial to integrate C storage into management strategies to mitigate climate change. The number of studies focussing on the effects of different management options on the C cycle and on C stocks has considerably increased (e.g., Campbell et al. 2009; Jurgensen et al. 2012; Powers et al. 2012; Ruíz-Peinado et al. 2013, 2014). However, most studies focussed on particular sites, species and management regimes, lacking therefore a generalization of the impact of forest management on C sequestration. In this respect, available historical records are useful for analysing the effects of past management activities on C stocks while forest growth models can be helpful to estimate future C stocks under different management alternatives (Kolari et al. 2004; Balboa-Murias et al. 2006).

Furthermore, it is important to consider the impact global change on carbon storage, since there are ample evidences that changes in temperature and rainfall patterns observed during last decades are modifying forest growth rates (e.g. Pretzsch et al. 2014), including some in Mediterranean pine species (Andreu et al. 2007; Martín-Benito et al. 2010a; Río et al. 2014). 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 (Prestzsch 2002). On the other hand, process based models are more versatile to account for a broader climatic conditions providing more reliable forecast of forest growth in a changing environment (Fontes et al. 2010). However, in some cases they are less suitable than empirical models for simulating management regimes.

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 strategies should be based on historical analysis as well as on ecological knowledge in order to determine the extent to which the traditional forest management practices are adequate in the face of climate change. For example, the regulation of stand density through thinning was found to mitigate the drought effect on tree growth in Mediterranean pinewoods (Martín-Benito et al. 2010b; Linares et al. 2009; Fernández de Uña et al. 2015).

Table 15.1 Area of the Mediterranean pine forests in Spain by species according to Montero and Serrada (2013) (in thousands of hectares)

Species	Area	%Total Forest area
Pinus halepensis	1926	10.5 %
Pinus pinaster	1373	7.5 %
Pinus sylvestris	1184	6.5 %
Pinus nigra	625	3.4 %
Pinus pinea	390	2.1 %
Total	5498	30.0 %

Table 15.2 Carbon stocks in million of Mg C for the Mediterranean pine forests in Spain in 2010 based on Spanish National Forest Inventory

Species	C stocks	Net C sequestration (net growth/year)
Pinus halepensis	46.14	1.36
Pinus nigra	42.46	1.01
Pinus pinea	17.25	0.50
Pinus pinaster	62.48	1.06
Pinus sylvestris	81.05	1.86
Total	249.38	5.79

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 halepensis* Mill. (Aleppo pine), *P. nigra* Arn. (European black pine), *P. pinea* L. (stone pine), *P. pinaster* Ait. (Maritime pine) and *P. sylvestris* L. (Scots pine), and these cover more than 5 million ha as dominant species (Table 15.1), which means around 30 % of the national forest area (Montero and Serrada 2013).

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 those most frequently used in afforestation programmes and therefore, are of particular interest in terms of carbon sequestration.

Carbon accumulation in Spanish pine forests was over 249 million of Mg C in 2010, based on the second and third Spanish National Forest Inventories. The net annual amount of carbon sequestration through forest growth is around 5.8 million of Mg C (Table 15.2). More than half of the total C 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 information on forest carbon come from a number of sources including: (i) carbon stock estimates under different management plans using a chronosequence trial in *Pinus sylvestris* forests; (ii) simulations based on the process model 3-PG of the effect of different thinning regimes on *Pinus pinaster* biomass under a climate change scenario; (iii) a comparison of the effect of different age structures in *Pinus pinea* forest using the PINEA growth model which includes the biomass allocated in cones and considers the different wood uses; and finally, iv) a model for estimating coarse woody debris.

15.2 Effect of Silviculture in Carbon Stocks in Scots Pine Forests

The issue of carbon role in forests is one of the values that must be considered in order to decide how they should be managed. This value is known to vary greatly during stand development, and depends on the site and the type of management executed. Most carbon budget estimates are based on measurement from middle-aged or mature stand and little data are available for different stages of forest succession (Kolari et al. 2004). 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). However, few studies focussed on forest management effects on those pinewoods (Ruiz-Peinado et al. 2013, 2014). In this section, we evaluated the relationship between carbon storage and different types of management over the length of the harvest rotation in Scots pine stands.

15.2.1 Dataset

A chronosequence trial was established in Pinar de Valsaín and Pinar de Navafría, managed Scots pine (*Pinus sylvestris* L.) forests located in the Central Mountain Range of Spain. In Valsaín, a group shelterwood system is applied, opening the stand gradually and allowing regeneration to take place naturally in a 40 years period. A moderate thinning regime is applied from the stem-exclusion stage onwards. The regeneration in Navafría forest is achieved using the uniform shelterwood system. Soil preparation is needed most of the times to achieve natural regeneration, although when this does not succeed, seedlings were planted. An intensive thinning regime is applied from the early stages.

Data were collected in six and five rectangular 0.5 ha permanent research plots that were installed in Valsaín and Navafría forests, respectively in 2001 (Table 15.3). These plots cover all the current age classes in both forests: from 1 to 120 years in Valsaín, where rotation length is 120 years, and from 1 to 100 years in Navafría, where rotation length is 100 years. In addition, in 2006 two additional plots were installed in the uppermost and lowermost limits of Scots pine growth spectrum, in uneven stands where few silvicultural interventions are carried out. In the lower limit (1300 m) Scots pine appears mixed with Pyrenean oak (*Quercus pyrenaica* Willd.) (plot V7, Table 15.3) while in the upper limit (1800 m) the pinewood is mixed with high mountain shrubs (mainly with *Juniperus communis* L. ssp *alpina*) (plot V8, Table 15.3). Since installation time, plots have been remeasured in 2006 and in 2011.

Table 15.3 Stand level variables in the experimental plots in Navafría forest (N1,N2,N3,N4 and N5) and in Valsaín (V1,V2,V3,V4,V5,V6,V7 and V8) at installation time: Plot size (m × m); Age class (years); *N* number of trees per ha with dbh ≥ 10 cm and dbh < 10 cm; dbh ≥ 10 cm and dbh < 10 cm, mean diameter at breast height (cm) of trees with dbh ≥ 10 cm and with dbh < 10 cm respectively; BA, basal area (m^2/ha); H ≥ 10 cm and H < 10 cm, mean height (m) of trees with dbh ≥ 10 cm and with dbh < 10 cm respectively

Plot	Plot size	Age class	N ≥ 10 cm	N < 10 cm	dbh ≥ 10 cm	dbh < 10 cm	BA	H ≥ 10 cm	H < 10 cm
Valsaín									
V1	100 × 50	1–20	458	8876	23.2	5.5	37.0	15.1	3.2
V2	85 × 58.8	21– 40	1484	1118	16.9	9.2	40.8	14.7	7.0
V3	100 × 50	41–60	1296	26	20.8	9.2	48.3	16.4	11.9
V4	100 × 50	61–80	686	–	30.5	–	53.3	23.5	–
V5	70.7 × 70.7	81–100	552	–	34.7	–	54.2	22.1	–
V6	100 × 50	101–120	332	–	38.9	–	41.3	24.0	–
V7	70.7 × 70.7	Uneven	526	1020	20.4	2.4	25.3	12.2	2.8
V8	100 × 50	Uneven	488	330	22.0	4.8	22.7	10.7	4.0
Navafría									
N1	70.7 × 70.7	1–20	810	4906	11.9	6.6	27.0	7.6	6.0
N2	110 × 45.5	21–40	2184	298	14.8	9.2	41.9	12.1	11.2
N3	70.7 × 70.7	41–60	680	–	32.9	–	60.6	22.0	–
N4	70.7 × 70.7	61–80	364	–	40.6	–	48.2	20.4	–
N5	70.7 × 70.7	81–100	304	–	42.5	–	44.2	22.1	–

15.2.2 Carbon Estimations

Dry biomass of the whole tree was estimated as a sum of their different fractions (stem, roots, branches and needles) using biomass models both for Scots pine (Ruíz-Peinado et al. 2011) and for Pyrenean oak (Ruiz-Peinado et al. 2012). Using the values provided by Ibañez et al. (2002) the percentage of carbon in the biomass of *P. sylvestris* and *Q. pyrenaica* trees is 50.9 % and 47.5 % respectively.

We modelled the carbon stored over the stand age in each forest (excluding the un-even-aged plots of Valsaín) using a semiparametric approach through smooth penalized splines (Eilers and Marx 1996) employing B-spline basis due to their flexibility and ease of computation. The smooth splines are curves formed by joining together several low-order polynomials at specified locations known as knots (Jordan et al. 2008), with a penalty approach. The use of a low-rank smoother solves the computational problems of other approaches while the penalty approach relaxes the importance of the number and location of the knots. This model can be formulated within a linear mixed effects modelling framework and, therefore, it allows considering the temporal correlation among measurements taken in the same plot (Durbán et al. 2005). The model included the study site as linear predictor and fitted a separate mean curve for each site by using a factor-by-curve interaction model to analyze the differences in the carbon stock among sites. In addition, we estimated the total carbon fixation rates (Mg C ha^{-1}) in each plot and period by comparisons of inventories.

15.2.3 Carbon Stocks and Fixation Rates Under the Two Management Systems

Concerning the even-aged stands, both forests followed a similar trend, reaching their maximum carbon dioxide fixation when stand density was more than 500 trees per ha and the mean diameter was larger than 30 cm, which occurs at the class of 60–80 years in Navafría and 80–100 years in Valsaín. After this stage, when the stand is opened gradually to facilitate the establishment of the seedlings of the next serial stage, and the densities go to approximately 300 trees ha^{-1} with mean dbh around 40 cm, the carbon stocks decreases in both forests. Hence, the two last measurements in Navafría forest (95 and 100 years) correspond to the end of the regeneration fellings, the establishment of the regeneration and, therefore, the end of the rotation period. At 100-year-old, the total C stock (area below the curve) was slightly larger in Valsaín than in Navafría. On the other hand the storage rate was more gradual in valsaín than in navafría (Fig. 15.1).

The growth rates of C evolved different along the rotation period in Navafría than in Valsaín (Fig. 15.2). In Navafría, we found a negative relationship between the growth rates and the plot age: the largest growth rates appeared in the youngest plots (average C = 3.91 Mg C ha^{-1} $year^{-1}$) and decreased to the oldest plot (average C = 0.54 Mg C ha^{-1} $year^{-1}$). In Valsaín, the growth rates increased from the youngest plot (average C = 1.84 Mg C ha^{-1} $year^{-1}$) to V2 (average C = 4.21 Mg C ha^{-1} $year^{-1}$), followed by a progressive decrease to the end of the rotation age (average C of V6 = 1.78 Mg C ha^{-1} $year^{-1}$). The difference of silvicultural systems and the thinning intensity might explain the different trends in growth rates in both forests.

The carbon fixed for the uneven-aged plots located in Valsaín (V7: 82.23 Mg C ha^{-1} in 2006 and 90.52 Mg C ha^{-1} in 2011; V8: 63.26 Mg C ha^{-1} in 2006 and 80.91 Mg C ha^{-1} in 2011) showed lower values than the rest of the even-aged plots studied in this forest.

Low C accumulation values in the uneven-aged plot V8 can be explained by the small tree growth and the low density in high-altitude environments. In the mixed oak and pine stand, located at the lower limit of Scots pine distribution area, C accumulation values were found to be much lower than in any of the pinewood stands (except for N1 and the two last measurements in N5), also reflected in the smaller basal area in plot V7. Analyses of the carbon dioxide amounts previously conducted in the same geographical area (Bogino et al. 2006) also highlighted the difference between pure (482.63 Mg CO_2 ha^{-1}) and mixed Scots pine stands (327.73 Mg CO_2 ha^{-1}). These differences might be due to the lower density of this plot and the much lower productivity rates of *Quercus pyrenaica* than *Pinus sylvestris.*

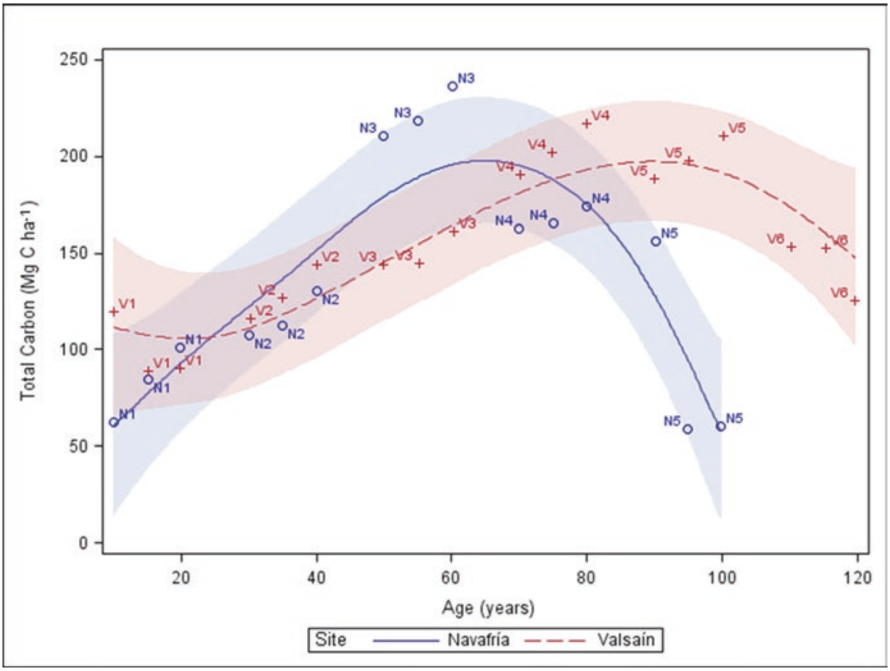

Fig. 15.1 Estimated smooth trends of carbon stock in even-aged stands in solid lines and 95 % intervals of confidence in dashed lines. *Blue* colour = navafría, *red* colour = valsaín. *Blue circle* = observed values in navafría and *red crosses* = observed values in valsaín

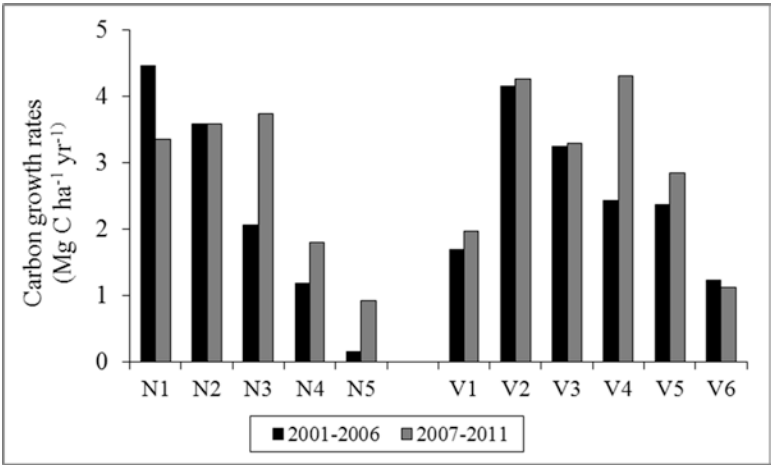

Fig. 15.2 Growth rates of carbon (Mg C ha^{-1} year^{-1}) in even-aged stands in Navafría and Valsaín plots

The figures obtained in this study are higher than the carbon stocks in Scots pine forests in Southern Finland, where values of 216 Mg CO_2 ha^{-1} for a 40 years old plot and 281 Mg CO_2 ha^{-1} for a 75 years old plot were found for similar density stands (Kolari et al. 2004). Concerning management alternatives, in a long-term Mediterranean thinning experiment Ruiz-Peinado et al. (2014) reported mean values that ranged from 128 to 193 Mg C ha^{-1} at the age of 52 years depending on the thinning treatment, with the greatest value in unthinned plots. This higher value in non-managed plots agrees with our finding that less intensive silviculture stores more carbon. Previous studies suggest that elongation of the rotation length increases the carbon stocks in Scots pine forests (Kaipainen et al. 2004; Liski et al. 2004). However, simulation studies suggested that very long rotations do not necessary maximize the carbon balance because the stand productivity declined due to the maximum annual increment has been exceeded (Jandl et al. 2007). Moreover, greater differences between the silvicultural systems applied in Valsaín and Navafría forests may arise if other components are taken into account (Montes and Cañellas 2006; Moreno-Fernández et al. 2015).

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 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 to 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 (Vayreda et al. 2012).

15.3 Mitigation Through Forest Thinning in Mediterranean Maritime Pine Afforestations

Forest adaptation and mitigation have been addressed in the scientific agenda as two important measures to cope with climate change effects. Forest managers are now facing the need to assess the role of silvicultural practices in this context. The most important tending method available to silviculturists is thinning, a partial harvest that regulates the distribution of growing space so that standing trees may benefit in terms of competition, growth and health status (Smith et al. 1997). Favourable carbon sequestration occurs generally when low intensity thinning is applied (Balboa-Murias et al. 2006; Pohjola and Valsta 2006). Aboveground onsite carbon stock is correlated with stocking density (D'Amato et al. 2011) although when off-site carbon stock is considered the total carbon sequestration potential is usually found to increase with thinning (Ruíz-Peinado et al. 2013). However, the positive thinning effect in C sequestration may be modulated by species and the grade of the thinning that it should not exceed the 'marginal thinning degree' that cause a loss in volume production (Hamilton 1981; Río et al. 2008).

An important question to be addressed it is if the currently applied thinning regimes will produce same results under changing climate conditions. This question may be answered using forest growth models that allow for variability in climate conditions such as hybrid models (Kimmins et al. 2010; Fontes et al. 2010). In this section we analyse thinning effects on Mediterranean maritime pine (*Pinus pinaster* Ait. ssp. *mesogeensis*) carbon stocks trough simulations from a forest growth model. The hybrid modeling approach allows us to simulate climate change effects on forest stands and, at the same time to assess the mitigation potential of thinning.

15.3.1 Forest Growth Hybrid Model Approach

We applied the 3-PG model (Landsberg and Waring 1997) to assess the effect of climate change and thinning on an afforested stand of Mediterranean maritime pine in south Spain. The 3-PG model is a canopy carbon-balance model of forest growth designed to simulate biomass production of pure, even-aged stands. It is hybrid in the sense that combines physiological processes and empirical relationships. 3-PG comprises four submodels (Landsberg and Sands 2011): growth modifiers and NPP, stocking and mortality, biomass allocation and soil water balance. The inputs of the model includes weather data (radiation, temperature, vapour pressure deficit and rainfall), site factors like a fertility rating and initial conditions such as initial biomass, stocking or available soil water. The main outputs are biomass pools and available soil water, forest manager variables like volume or basal area, and physiological responses like NPP, LAI or evapotranspiration. More details about the model structure and outputs can be found in Landsberg and Waring (1997) and Landsberg and Sands (2011).

15.3.2 Model Calibration and Climate Scenarios

We first calibrated the model to a stand located in Fuencaliente (Ciudad Real, 38°28'N 4°21'W), by the edge of Sierra Morena in south Spain. The soil type is Xerochrept according to FAO classification and parent material is acidic and mineralogy is mainly quartzite. Soil analysis showed that the fertility is medium-high, the texture classification is sandy-loam. Climate is genuine Mediterranean (Csa

Table 15.4 Summary of forest variables in Fuencaliente experimental site during the time span of the experiment (from 1984 to 2010)

Treatment	Age	N (stems ha^{-1})	Dg (cm)	BA (m^2 ha^{-1})	V (m^3 ha^{-1})	Ho (m)	Wa (Mg dm ha^{-1})
CT	33	1193	22.9	49.0	311.4	14.9	142.1
	59	870	31.3	67.0	589.9	20.6	249.2
MT	33	1420	21.2	49.6	320.8	15.3	144.7
	59	427	37.4	46.6	433.2	21	185.9
HT	33	1570	20.4	51.4	322.8	14.9	145.5
	59	340	39.3	41.3	385.2	20.6	165.2

CT is unthinned plots, *MT* are moderate thinned plots, *HT* is heavy thinned plots, *N* is stocking, *Dg* is quadratic mean diameter, *BA* is basal area, *V* is mean volume, *Ho* is dominant height and *Wa* is above ground woody biomass, including branches

according to Köppen's system). In this stand a thinning experiment was set in 1984. Three replicates of three treatments were established within a random complete block experimental design. The treatments were a moderate low thinning (MT), a heavy low thinning (HT) and a control with no thinning (CT). For the calibration the time span of the experiment ranges from 1984 to 2010. Table 15.4 shows a brief description of state variables at the beginning and at the end of the experiment.

Model parameters were calibrated using data from different sources: a litterfall experiment in the same plots (Roig et al. 2005); biomass data from the site and from young afforestations in Central Spain to obtain biomass allocation parameters; sample needles to fit age-dependent SLA functions; and monthly climatic data for the studied period (reference climate) extracted from a close weather station of the Spanish Meteorological Agency (AEMET). Parameters with lacking information were set to default value according to Sands (2004). Details of the calibration can be requested to authors (Bravo-Oviedo et al. 2011). Figure 15.3 shows the performance of the calibration with observed mean annual climate in monthly time step.

In order to evaluate the role of this thinning regime in mitigating climate change effects we developed a climate change scenario based on RCP8.2 scenario (Van Vuuren et al. 2011). According to the fifth Assessment Report projections in the Iberian Peninsula (MAGRAMA 2013) by 2035 winter temperature would increase 0.9 °C, summer temperature would increase 1.4 °C with a mean annual temperature increase of 1.1 °C. Precipitation would drop 1 % in winter, 3 % in summer and 2 % annually. In addition we arbitrarily reduce the available soil water in 5 % for this scenario.

15.3.3 Simulation Results

The simulated woody biomass under reference and changing climate is presented in Table 15.5. Standing C stock decreases with thinning intensity in both climate scenarios, although the impact of thinning is 5.2 % and 3.3 % lower in the simulated scenario for HT and MT regimes respectively. The climate effect on unthinned stands is therefore higher than in thinned stands. When C losses due to mortality and harvest are incorporated in the C computation the total woody C production is

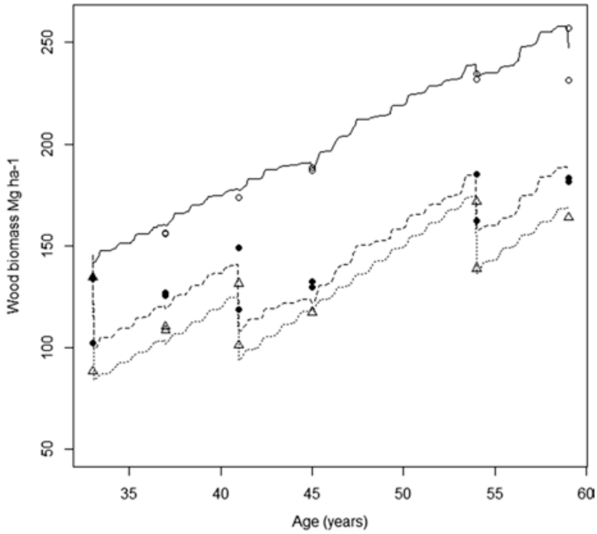

Fig. 15.3 Woody biomass (Mg dry matter ha^{-1}) trajectory of observed data and 3-PG output after calibration of an afforested *P. pinaster* Ait. ssp mesogeensis stand. *Open circles* are observed data in control plots; *solid circles* are observed data in MT plots; *triangles* are observed data in HT plots. Lines are 3-PG monthly estimates. *Solid line* CT plots; *dashed line* MT plots; *dotted line* HT plots

Table 15.5 3-PG simulation results of standing and total woody C (Mg C ha^{-1}) in Fuencaliente in two climate scenarios and three treatments

Scenario	Treatment	Standing woody C	Within scenario thinning impact	Between scenario climate impact	Total woody C production	Within scenario thinning impact	Between scenario climate impact
Reference climate	CT	126.3	–	–	136.4	–	–
	MT	95.3	−24.5	–	152.1	11.6	–
	HT	86.2	−31.7	–	153.7	12.7	–
RCP8.5-based	CT	115.1	–	−8.9	127.4	–	−6.6
	MT	90.7	−21.2	−4.8	142.6	12.0	−6.2
	HT	84.6	−26.5	−1.9	151.4	18.9	−1.5

Reference climate is mean actual climate (1984–2010). Within scenario thinning impact compares MT and HT in relation to CT output. Between scenario climate impact compares the same thinning regime output in RCP8.5-based scenario vs. reference climate

higher in thinned stands. Under changing climate the impact of thinning is also higher in heavier thinned stands (HT), but with little climate impact on total woody C between scenarios.

A trade-off between standing and total woody C production is consistent with previous reports about the role of thinning in mitigating (D'Amato et al. 2011). In the climate scenario tested heavy thinning can play an important role in mitigating negative climate effects. Although standing C stock is reduced in both scenarios, under harsher conditions the C loss is lower in HT plots. Interestingly total C production (including off-site carbon) is virtual the same in heavy thinned stands in both scenarios. Our results suggest that strongest reduction in competition might benefit remaining trees in drier conditions.

Thinning is a key treatment to reduce the impact of extreme droughts on tree growth in Mediterranean pines (Martín-Benito et al. 2010b; Fernández de Uña et al. 2015). As our results highlight, under a drier climate scenario the loss of on-site carbon due to heavy thinning may be lower while the total C higher than in unthinned stands. Therefore, the application of thinning in Mediterranean pinewoods is a crucial strategy for forest adaptation and climate change mitigation.

15.4 Carbon Sequestration in Even and Uneven Aged Stone Pine Stands

Stone pine (*Pinus pinea*) stands have traditionally been managed as even-aged with low stocking densities, facilitating crown growth and increased 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 and Montero 2007; Calama et al. 2008a). 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 by comparing carbon stocks following even-aged and uneven-aged management. For this purpose, an individual-tree growth model and biomass equations were used.

15.4.1 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. Further details regarding the PINEA2 model for even-aged stands can be found in Calama et al. (2007, 2008b). 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, resulting in PINEA_IRR model (see Calama et al. 2008a). This calibrated version also presents a modular structure, including specific functions for site productivity, state attributes (height-diameter, crown dimensions, taper function and cone production) and diameter increment.

15.4.2 Biomass Equations and Carbon Estimations

Tree biomass (by component) was estimated from diameter at breast height (dbh, measured at 1.3 m above ground) and total tree height using the biomass equations proposed by Ruiz-Peinado et al. (2011) for stone pine. Carbon in 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.

15.4.3 Even and Uneven-Aged Alternatives

The growth and development of a 1 hectare 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 15 m at a total age 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/ha at age 20, and that three thinnings are applied during the cycle: one systematic at age 30, reducing stand density to 350 stems/ha; and two selective low thinnings at ages 45 and 60, reducing the density to 250 and 150 stems/ha respectively. Rotation length is assumed to extend up to 100 years, and although shelterwood regeneration fellings are commonly applied during a 20 years period, we simulate single clearcutting at 100 years old.
2. In the uneven-aged structure, it was assumed that the stand displays the uneven equilibrium state (Table 15.6) 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, while overmature large cone producer trees are favoured. Sustainability of the management is warranted if basal area ranges

Table 15.6 Proposed structure for uneven-aged stands of stone pine, rotation period 25 years

Age-class	Before selective felling (N° trees/ha)	Removed in selective felling (N° trees/ha)	After selective felling (N° 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

between 8 and 20 m^2/ha, standing stocking volume between 45 and 100 m^3/ha and a constant recruitment of 150 saplings (dbh > 5 cm) per hectare and 25 years period is assumed.

15.4.4 Carbon Sequestration in Even and Uneven-Aged Stands

The carbon fixed in stands of both age structures over the course of one rotation period (100 years) is presented in Fig. 15.4. The carbon fixed in the even-aged stand reached a maximum of 84 Mg C ha^{-1} by the end of the rotation period while the simulated uneven-aged stand resulted in a maximum of 63 Mg C ha^{-1} and a minimum of 30 Mg C ha^{-1}. Taking into account the total cumulative amount of carbon fixed during the 100 year cycle, while even-aged stands accumulate 145 Mg C ha^{-1}, uneven-aged stands fixed 130 Mg C ha^{-1}, thus being more effective in this role. It also should be pointed out that the uneven-aged stand maintained a constant stock of fixed carbon over 27 Mg C ha^{-1} 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 uneven-aged stand fixed 0.14 Mg C ha^{-1} $year^{-1}$ more carbon than the even-aged stand (Table 15.7), which means a difference of 13.6 Mg C ha^{-1} over the 100 year period.

Biomass extractions (only aboveground biomass) totalled 87.3 Mg C $ha^{-1}$of fixed carbon 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. 15.5). Around 60 Mg C $ha^{-1}$of this carbon was fixed in stems and large branches, which are used as pulpwood and saw timber, while the rest belongs to tree components which remain in forest and decompose. In uneven-aged stands, four thinning treatments were applied over the 100 years (every 25 years), removing 103 Mg C ha^{-1} of fixed carbon, of which 75 Mg C ha^{-1}, destined for the timber industry (Fig. 15.5).

In terms of productivity, the uneven-aged structure favours cone production, which is one of the main objectives in the management of stone pine forests. In terms of weight of green cones (dry weights in Table 15.7), which is the commonly used trade unit, throughout the 100 year cycle uneven-aged stands produce 171 kg

Fig. 15.4 Total above and belowground carbon fixed by even and uneven-aged stands of stone pine over a period of 100 years

Table 15.7 Annual increments of above and belowground biomass, annual cone production (dry weight) and their equivalents in C fixed in even and uneven-aged stands of stone pine

		Aboveground	Belowground	Cones	Total
Biomass(kg ha^{-1}yer^{-1})	Even-aged	1721.1	525.3	76.1	2322.5
	Uneven-aged	2034.4	520.9	108.2	2663.5
C fixation (Mg C ha^{-1} year^{-1})	Even-aged	0.873	0.267	0.055	1.195
	Uneven-aged	1.031	0.265	0.038	1.334

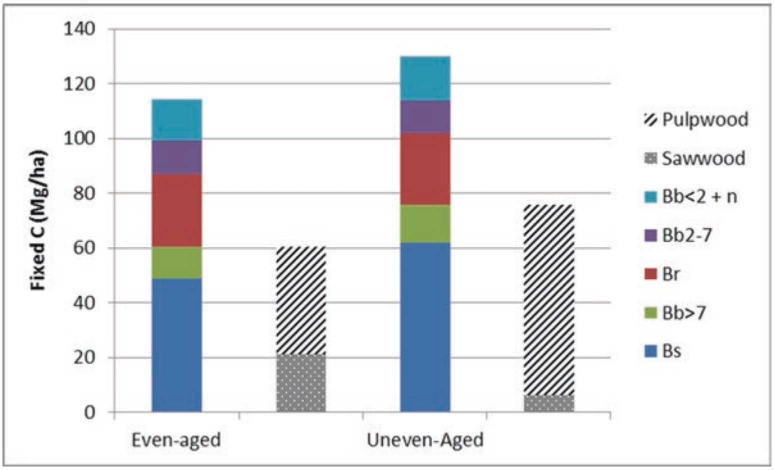

Fig. 15.5 Carbon fixed in extracted biomass by component over the 100-year period in even and uneven-aged stands of stone pine. Bs, Bb$<_{2+n}$, Bb$_{2-7}$, Bb$>_{7}$, and Br correspond to: stem, branches under 2 cm diameter plus needles, branches between 2 and 7 cm diameter, branches larger than 7 cm diameter 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

ha^{-1} year^{-1}, while even aged stands only produce 120.8 kg ha^{-1} year^{-1}. However, contribution of cone production to CO_2 fixation can be considered negligible, since most of the cones are collected each year to obtain edible pine nuts, while the residuals of the industrial process of pine nut extraction are usually burned. With respect to timber production, the uneven-aged structure is also more favourable, although in that case, as bigger trees are harvested at rotation ages over 125 years, main part of this timber is affected by stem rot due to *Phellinus pini*, thus much of the timber production ends up as pulpwood or firewood, with less than 10 % of total stem volume transferred to sawmill industry. Both pulpwood and firewood uses result in short term return of carbon to the atmosphere. On the contrary, while even-aged structure lead to less amount of total biomass fixed, approximately 50 % of the harvested timber volume can be used for sawmills, resulting in extended periods of carbon return 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 and non-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 carbon estimation 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.

15.5 Modelling Coarse Woody Debris in Pine Plantations

Dead wood is a key component 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 and 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. However, in recent decades, forestry has been addressed towards a more close-to-nature approach with the aim of developing forest stands that are comparable to natural ones in so far as structure, composition and regeneration processes (Bauhus et al. 2009). In this perspective, silvicultural restrictions on CWD removal have been proposed or introduced in many regions (Keeton and Franklin 2005). These prescriptions lead to create an adequate stock of CWD and promote CWD structures for maintaining biodiversity (Siitonen et al. 2000). In contrast, nowadays, the growing interest in harvesting logging residues for energy production might reduce both the size and the diversity of woody debris (Wright 2006).

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 5 to 10 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 types, disturbance regimes, topography and stand characteristics (Spies et al. 1988; Harmon and Hua 1991).

In practice, snag and log dynamics are important to define the appropriate quantity, density, size (both diameter and height), 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 and Moghaddas 2005). Studies which focus on modelling the abundance of snags and logs in Mediterranean type forest ecosystems are scarce (Montes and 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*, *P. pinaster* and *P. nigra* in northern Spain.

15.5.1 Database

The study area, situated in the north of Spain, constitutes a homogeneous transitional zone with altitudes ranging from 800 to 1000 m.a.s.l., 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*, *Q. ilex* L. and *Q. faginea* Lam. As a result of an extensive pine plantation program carried out during the 60's, *Pinus* spp. 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 of 50 m. 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 in 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 (diameter at breast height, dbh $\geq$ 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 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 interception point, length, state of decomposition and wildlife characteristics. Decomposition classes were considered following the criterion 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 scaled up to give a basal area per hectare. The log volume was estimated through the equation $V_i = (\pi^2 d_i^2)/8\ L$ (Warren and Olsen 1964; Van Wagner 1968), where V: log volume (m^3/ha), d: diameter of each log (cm), L: total length of the transects, which in this case was 100 m.

An intensive inventory of logs with a diameter greater than 1 cm and lower than 7.5 cm was carried out in 32 out of the 66 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 15.8. The following characteristics were also recorded for each plot: number of non-inventoried 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).

15.5.2 Modelling Approach

15.5.2.1 Two Step Regression Approach

A two step regression approach (Woollons 1998; Álvarez González et al. 2004; Bravo et al. 2008) 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. 15.1), 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.

Table 15.8 Database characteristics used to develop the snag and log models for a pine plantation in Northern Spain

Variable	Mean	Minimum	Maximum	Standard deviation
N (trees ha^{-1})	803	26	1585	341
BA (m^2 ha^{-1})	23.2	5.6	39.3	8.2
Dg (cm)	22.2	13.2	58.3	6.3
BA_{snags} (m^2 ha^{-1})	0.2	0.0	1.2	0.3
V_{logs} (m^3 ha^{-1})	1.5	0.0	5.9	1.9

N trees per ha, *BA* basal area, *Dg* quadratic mean diameter, BA_{snags} basal area of snags; V_{logs} volume of logs

$$P = (1 + e^{-(\alpha + \sum b_i X_i)})^{-1} \tag{15.1}$$

Several predictor variables were tested among stand variables, site variables, climate variables and forest management history. Final logistic regression equations included only significant predictors ($p \leq 0.05$). The goodness of fit was evaluated using the Hosmer and Lemeshow (1989) and the Akaike Information Criterion (Zhang et al. 1997). PROC LOGISTIC SAS 8.1 statistical program was used in the process (SAS 2001). Receiver Operating Characteristic (ROC) curves for each model were used to compare the accuracy of different logistic regression models.

Lineal models were used (Eq. 15.2), 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:

$$\hat{y} = a_0 + \sum a_j X_j \tag{15.2}$$

where $\hat{y}$ is estimated BA_{snag} or Vol_{logs}, X_i are predictor variables as in 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. 15.3) and by fitting a straight line between actual and predicted values (Huang et al. 2003) where slope and independent term should be equal to 1 and 0 respectively.

$$R^2 = 100 * (1 - \frac{S_e^2}{S_y^2}) \tag{15.3}$$

where S_e^2 and S_y^2 are, respectively, the residual sample's variance and the dependent variable sample's variance.

15.5.2.2 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 C_{logs} (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, physiographic variables, and climatic variables. Interactions between species and stand variables were also checked as predictive variables.

The goodness of fit for the carbon model was assessed using the coefficient of 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 $P< 0.05$.

15.5.3 CWD Models

15.5.3.1 Snags and Logs Models

The final logistic model to predict the presence of CWD (Eq. 15.4) includes as independent variables altitude, minimum temperature, clay and silt content and a dummy variable indicating if harvest operation has been carried out in the last 5 years. The Akaike information criterion was 79.709, and the Hosmer & Lemeshow test ($P >$ 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.

$$P = (1 + e^{-(-43.2715 + 2.5501 Alt + 0.3939 MinT + 4.6453 ClayText + 2.4439 SiltText + 2.5136 Harvesting})^{-1} \tag{15.4}$$

where Alt is altitude, m; MinT is minimum temperature, °C; ClayText and SiltText are dummy variables for clay and silt textured soils, respectively, both equal to 0 for sandy soils, and Harvesting is a dummy variable with 1 for harvest and 0 for no harvest.

The final linear models for snag basal area (Eq. 15.5) and log volume (Eq. 15.6) resulted in adjusted coefficients of determination of 17.47 % for snags and 46.05 % for logs. The figure for snags increases with a decrease in the precipitation in June, whereas the volume of logs increases with an increase in the basal area of the stand and when the dominant height decreases.

$$\hat{BA}_{snags} = 2.1846 - 0.0038 \cdot P\_june \tag{15.5}$$

$$V\hat{o}l_{\log s} = 0.8821 + 0.2046 \cdot BA - 0.4028 \cdot Ho \tag{15.6}$$

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. BA_msp is the basal area of the main species in m^2/ha, BA is the basal area of the stand in m^2/ha, and Ho is the dominant height in m.

The proposed two-step model achieves joint model accuracy equal to 39.25 % and 62.75 % for snags and logs respectively. Graph analysis of real and 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.

15.5.3.2 Carbon Content Model

The final carbon content model obtained for logs (Eq. 15.7) resulted in a coefficient of determination of 53.33 % (adjusted coefficient of determination was 47.50 %). The carbon content of logs increases 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. The latter situation suggests early stages of development in a stand where no management intervention has been carried out as yet. Therefore, in this area, forest management plays an important role in the total coarse woody debris carbon sequestration.

$$\hat{C}_{\log s} = -54.5741 + 0.6594 m \log diam + 0.0346 BA - 0.0003 Sp * n \tag{15.7}$$

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 species dummy variables and the number of small stems (trees with diameter < 7.5 cm) (trees/ha). Forest management decisions are based on information regarding current and future forest conditions, so it is often necessary to project the changes in the system over time. The equations obtained allow us to predict the presence of CWD and quantify the amount of snags and logs in pine plantations in northern Spain.

The significant role of dead wood in carbon pooling linked to biodiversity is becoming clearer. The studied *Pinus* spp. planted stands show values of 0.78 Mg C ha^{-1}, which represents the 1 % of the total Carbon fixed in the ecosystems. This percentage is small because of the huge importance of tree live biomass and responds to stand histories and forest legacy, where forest management has been oriented to maximize the commercial value of the harvest and avoid different risks. However, CWD proportion can be a criterion for assessing and determining baseline silvicultural practices at biodiversity level. In this sense, these results can be useful for guidelines for prescribing adequate levels of CWD in terms of density, size and amount in the different components of dead wood.

The two-step model emphasised the importance of forest management in CWD presence in this region and allowed us to combine knowledge (understanding and data) and prediction of system dynamics. Unmanaged forests in Spain are scarce because of centuries of forestry practices. Intensive management activities have led to timber removal and woody material loss for centuries. However, Mediterranean forests currently present a low silviculture activity because of the harvest economic yield. In this sense, the two-step model can provide objective forecasts and information for exploring management options and silvicultural alternatives for sustainable forest management.

15.6 Conclusions

Our results showed how carbon sequestration changes over time, and with different management regimes in Mediterranean pine forests. The rotation length, thinning intensity, stand composition, as well as age structure influenced carbon stocks and carbon sequestration rates, with different results amongst species. A less intense management regime with the extension of rotation length 20 years increased carbon stocks in Scots pine forests. However, for Mediterranean maritime pine heavy thinning increased carbon sequestration when carbon fixed in removed wood was also considered. This highlights the importance of forest management, because despite unmanaged forests can show a higher amount of carbon on-site, managed stands can fixed more off-site carbon while being in a better condition in relation to climate change effects (droughts, pets or diseases, fires...).

Forest growth models are valuable tools to predict carbon fixation in forest systems under different management alternatives. The use of hybrid models such as 3-PG offers the possibility of estimating the growth response to climate, giving reliable information about the impact of climate change on forest carbon stocks. On the other hand, more detailed empirical model as PINEA2 and PINEA_IRR allowed us to study the effect of age structure on different carbon components, including cone production and wood products with different life cycles which may be considered in carbon accounts.

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 the CWD dynamic. The models presented for coarse woody debris allow quantifying the biomass accumulated in this component, and therefore to furthering our understanding of the carbon cycle in pine forests.

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Chapter 16
Carbon Sequestration of Ponderosa Pine Plantations in Northwestern Patagonia

P. Laclau, E. Andenmatten, F.J. Letourneau, and G. Loguercio

16.1 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 and Deccechis 2006). Although the usual management is industry-directed with dense plantations, forest

P. Laclau (✉) • E. Andenmatten • F.J. Letourneau
Instituto Nacional de Tecnología Agropecuaria (INTA), Estación Experimental Agropecuaria Bariloche CC 277, 8400 San Carlos de Bariloche, Río Negro, Argentina
e-mail: laclau.pablo@inta.gob.ar

G. Loguercio
Centro de Investigación y Experimentación Andino Patagónico CIEFAP, Ruta 259 km4, 9200 Esquel, Chubut, Argentina

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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 and Schlichter 2005; Rusch et al. 2005a, b; Loguercio et al. 2005; Sarasola et al. submitted). Also, the arising of market rules to accomplish carbon 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.

16.2 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. 16.1). The regional climate is cold temperate, with a 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 (3000–1000 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).

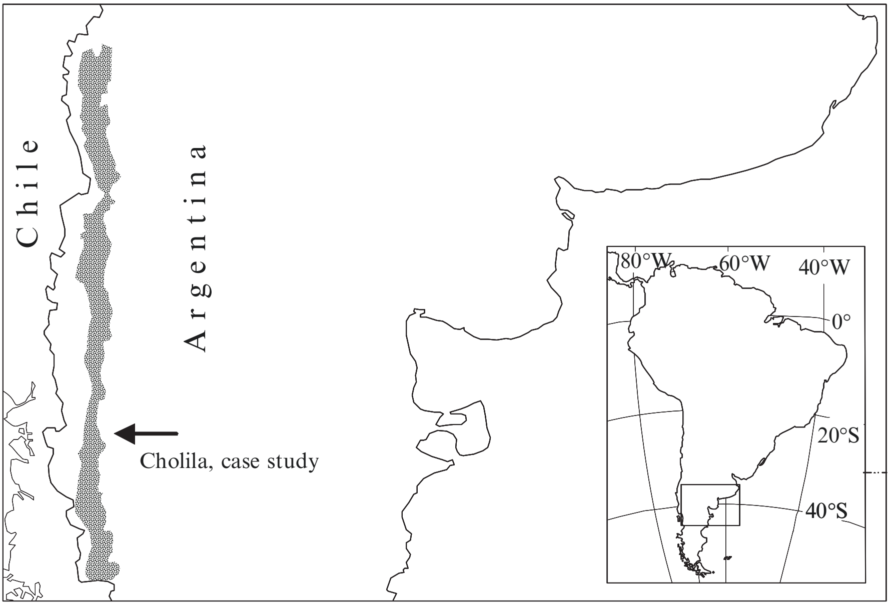

Fig. 16.1 Map location of the subandean region of Northwest Patagonia, suitable for ponderosa pine afforestation (*dotted area*) and location of the case study (Section 5.2)

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. 16.1). This vaguely defined ecotone zone – limited by an annual rainfall range from about 1200 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 and Lorenz 1988). Early in the twentieth 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.

16.3 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^{-1} $year^{-1}$.

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 and 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.

[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).

16.4 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 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. 16.2)
- 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

Fig. 16.2 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)

Grass-steppe → grass-shrub-steppe → shrub-grass-steppe → shrub-steppe

Fig. 16.3 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

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 and Bisigato 1998), trending to more irreversible states and ecosystem aridity. For the sub-Andean Patagonia, the states and transitions from rich to poorer range conditions are (Fig. 16.3).

Recently, some biomass and carbon assessments of various types of steppes were made. Where values found oscillated 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).

16.5 Stand Growth, Biomass, and Carbon Estimates

16.5.1 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 twentieth 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. 16.4).

Fig. 16.4 Unpruned, dense ponderosa pine plantation of 40 years age, in Puerto Patriada, Chubut (Photo: F. Letourneau)

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 1800 m^3 ha^{-1} volume and 131 m^2 ha^{-1} basal area (Moretti and Fritz 1989). The number of standing trees *per* hectare was 865, and the dominant tree height 37 m. The same stand was measured 8 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 16.1. 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 and 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 and Letourneau 1997). The driving variable of the model is the *dominant tree heigh*, Its dynamics can be predicted through site index curves and the

Table 16.1 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	1862	62
41° 59′ S, 71° 31′ W	21	80	23.9	14.0	1786	610	33
42° 00′ S, 71° 08′ W	19	107	33.7	12.8	1200	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	1957	876	30
43° 07′ S, 71° 34′ W	18	79	23.9	9.8	1766	625	34
43° 59′ S, 71° 31′ W	35	96	39.0	21.0	819	1285	40

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 and Cameron 1985) has the form:

$$V = a \times H^{b} \times RD^{c} \tag{16.1}$$

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. 16.1 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. 16.1, 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-soilwater 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. 16.1, the stand yield is expected to increase.

16.5.2 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:

$$\log B_i = a + b.\log x \tag{16.2}$$

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. 16.5. Assuming a carbon content of 50% of the total dry matter (IPCC 2003), these plantations sequestered about 73–330 Mg CO_2eq ha^{-1}, corresponding to a mean capture of about 4–18 Mg CO_2eq ha^{-1} $year^{-1}$ depending on the site class.

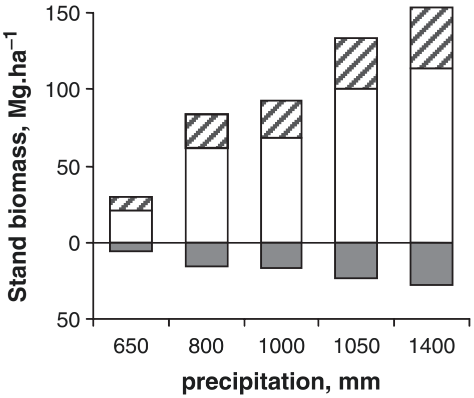

Fig. 16.5 Tree biomass of ponderosa pine stands, estimated from single tree models (Laclau 2003) applied to forest inventories, under different rainfall condition. The hatched sectors represent the crown biomass, the empty sectors, the stem biomass, and the grey ones, the root biomass (Laclau 2006 unpublished)

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 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. 16.5, 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 2 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).

16.6 Regional Scenarios of Carbon Sequestration

16.6.1 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 biomass or carbon stocks by means of allometric equations. Figure 16.6 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. 16.6), 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

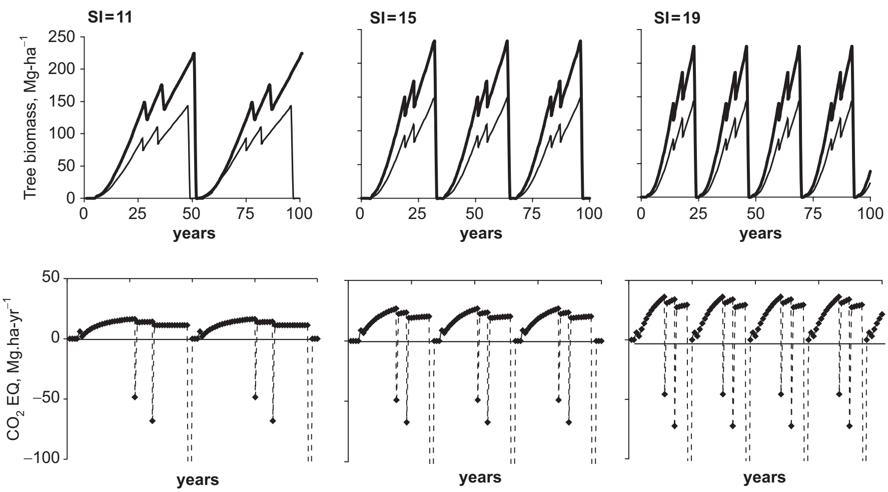

Fig. 16.6 Managed ponderosa pine rotations in three site classes (*above*), and their correspondent CO_2eq accumulation rates (*below*). At a low site quality ($SI_{20} = 11$) the rotation lasts 48 years, in the intermediate ($SI_{20} = 15$) 32 years, and in the best site ($SI_{20} = 19$), 23 years. 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

period was reached earlier (after: 19 years in $SI_{20} = 19$, 25 years in $SI_{20} = 15$, and 33 years in $SI_{20} = 11$).

16.6.2 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. 16.1). 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. 16.7). The dominant soils are allophanic or other volcanic types, with materials deposited over, or mixed with, glacial till (Veblen and 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

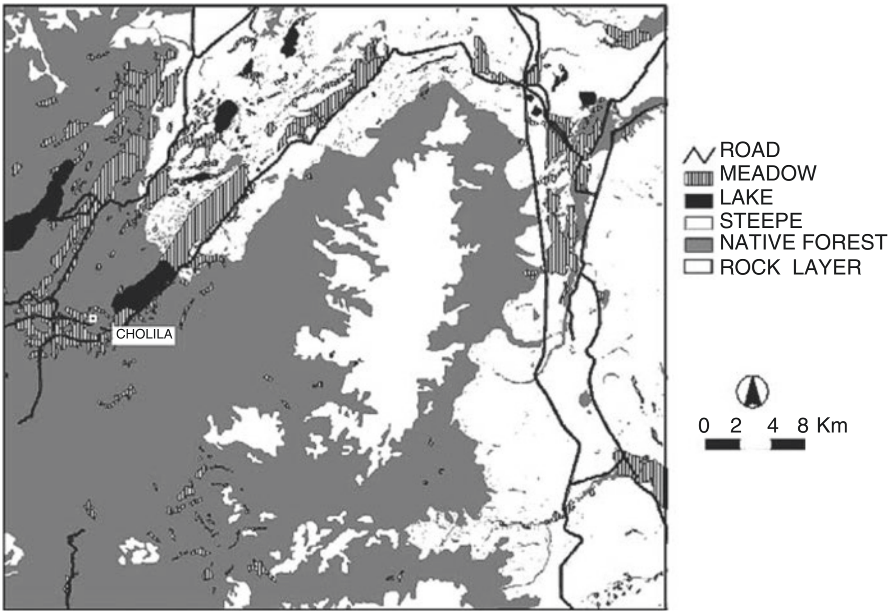

Fig. 16.7 Map of the environmental units of Cholila area. Steppe lands (*light gray units*) are considered afforestable for ponderosa pine

distance of no more than 50 km, while the rainfall consistently decreases from the Andean cordillera to the arid Patagonian steppe (from near 2000 mm year $^{-1}$ to 400 mm year^{-1}, Farias 2003).

The town of Cholila has almost 2000 inhabitants (INDEC 2001), and in the sorroundings there are about 62 rural settlements. A few farms *(estancias)* to the east own between 5000 and 50,000 ha each. To the west of town the farms have less than 3000 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.

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 1500 heads, except for the big farms. 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.

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 16.2.

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).

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 con-

Table 16.2 Site class, internode index (II, m), SI_{20} (m), rotation period (year) and land extension (ha, %) for ponderosa pine at the study region

Site class	II	SI_{20}	Rotation period	Land extension ha	%
I	4.3	17.1	38	2025	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

Table 16.3 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

Vegetation type	Accumulated carbon Mean	SD	CO_2eq incr	Land area (Ha)	(%)
Shrubs (several spp.)	56.1	12.9	25.9	16,557	24.2
Rosa eglanteria thickets	34.8	13.6	27.1	2056	3.0
Shrub steppe	25.3	10.1	20.1	16,964	24.8
Dwarf-shrub steppe	25.3	10.1	20.1	1759	2.6
Grass steppe	24.6	9.9	19.8	24,546	35.9
Degraded steppes	13.0	5.6	11.2	6446	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

sisted 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 aboveand below-ground 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 16.3).

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 16.2).
- The afforestation gradually extended along 10 years, at a rate of 2050 ha *per* year, with continuing reforestation after each rotation.

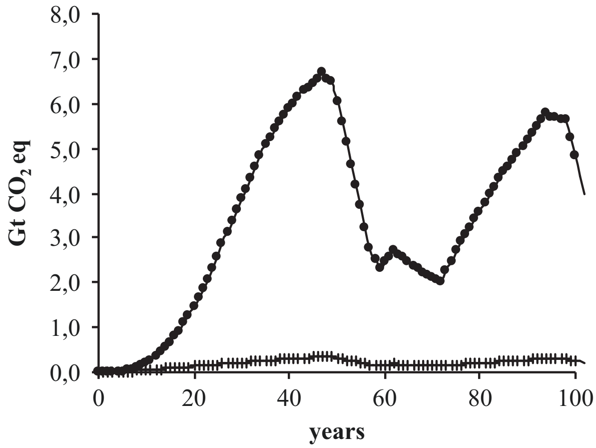

Fig. 16.8 Carbon sequestering scenario for 20,500 ha afforested lands with ponderosa pine during succesive rotations (*black-circled line*) and baseline (*crossed line*)

- 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. 16.8). In spite of the assumed increase, the baseline will remain low (Fig. 16.8). The net carbon capture (afforestation gain minus baseline), after a peak of 6.3 Gt MgCO_2eq at 50 years, will vary between 2 and 6 Gt CO_2eq.

16.6.3 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 2500 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. 16.9a. The *accumulated mean* biomass for each site should be as indicated in Fig. 16.9b.

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 and Deccechis 2006) of about 5000 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.

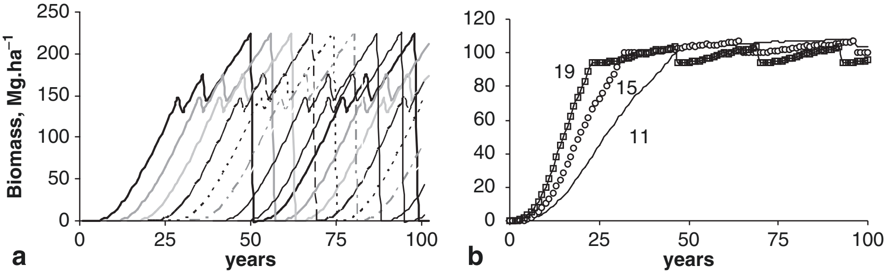

Fig. 16.9 (**a**) Stand biomass curves of succesive rotations of ponderosa pine, and (**b**) accumulated mean biomass curves for three site qualities: $SI_{20} = 19$ (*squared line*), $SI_{20} = 15$ (*empty circles*) and $SI_{20} = 11$ (*line*)

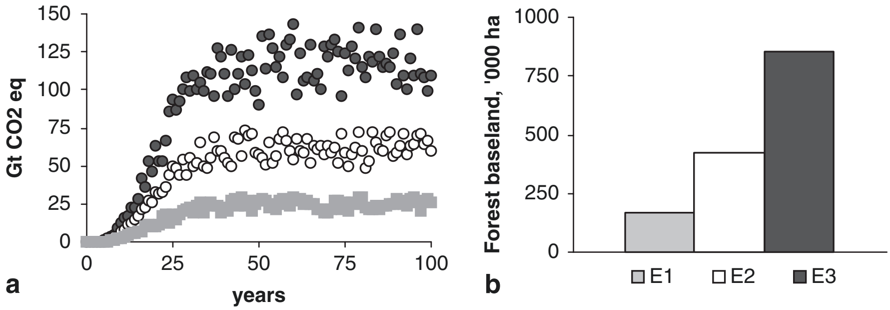

Fig. 16.10 Simulation output for regional scenarios E1 (*gray symbols*), E2 (*empty black-outlined symbols*) and E3 (*black filled symbols*); (**a**) carbon sequestration, (**b**) forest baseland

– 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. 16.10a). 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. 16.10b).

16.7 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 and 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.

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Chapter 17
Assessing Pine Wilt Disease Risk Under a Climate Change Scenario in Northwestern Spain

G. Pérez, J.J. Díez, F. Ibeas, and J.A. Pajares

17.1 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).

G. Pérez
Centro de Sanidad Forestal de Calabazanos, Junta de Castilla y León, Palencia, Spain

J.J. Díez • F. Ibeas • J.A. Pajares (✉)
iuFOR – Sustainable Forest Management Research Institute,
Universidad de Valladolid – INIA, Palencia, Spain

Departamento de Producción Vegetal y Recursos Forestales, ETS de Ingenierías Agrarias,
Universidad de Valladolid, Palencia, Spain
e-mail: jpajares@pvs.uva.es

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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. 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 and 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

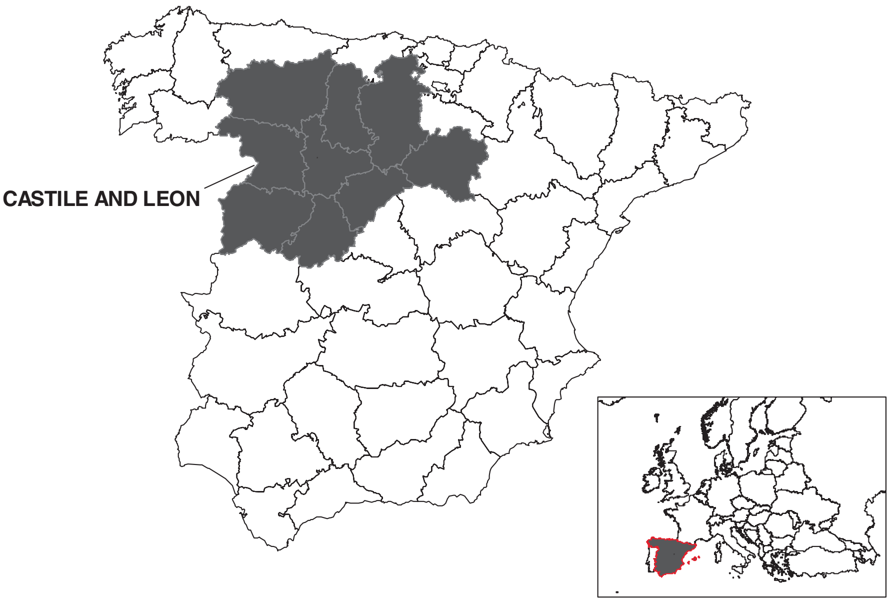

Fig. 17.1 Location of Castile and Leon in north-western Spain

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 were present (Maleck and 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 and 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. 17.1) 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.

17.2 Material and Methods

17.2.1 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 and 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 17.1).

Table 17.1 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 17.2 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

17.2.2 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) Arcview 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 17.2). 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 17.2 outlines the process for generating the final theme maps showing the risk levels for all pine stands and for each pine species.

17.3 Results and Discussion

17.3.1 Risk Levels in Castile and Leon Pinewoods

Result of risk levels for Castile and Leon considering all pine stands together are presented in Table 17.3. 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. 17.3). Only near 5% of the pinewoods exhibited a very high risk of PWD development (Table 17.3). 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 17.3).

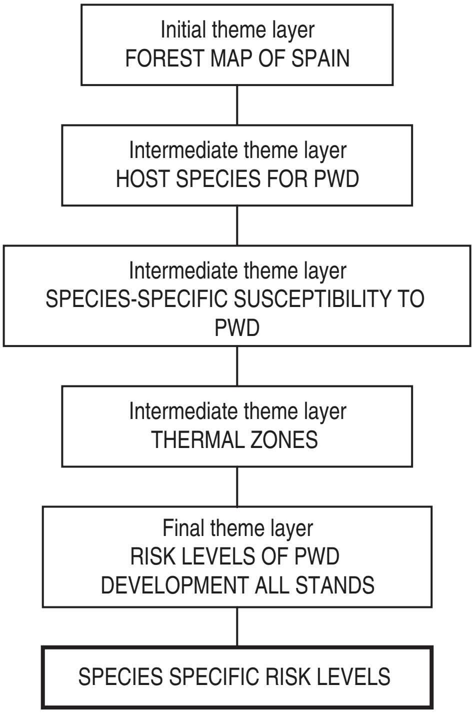

Fig. 17.2 Sequence for generating pine wilt disease risk maps in Castile and Leon

Table 17.3 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

17.3.2 Species-Specific Risk Levels of Disease Development

17.3.2.1 *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 17.4). Stands showing higher susceptibility are located in southernmost areas of the Central Range, in the provinces of Avila and Salamanca (Alberche and Tiétar rivers,

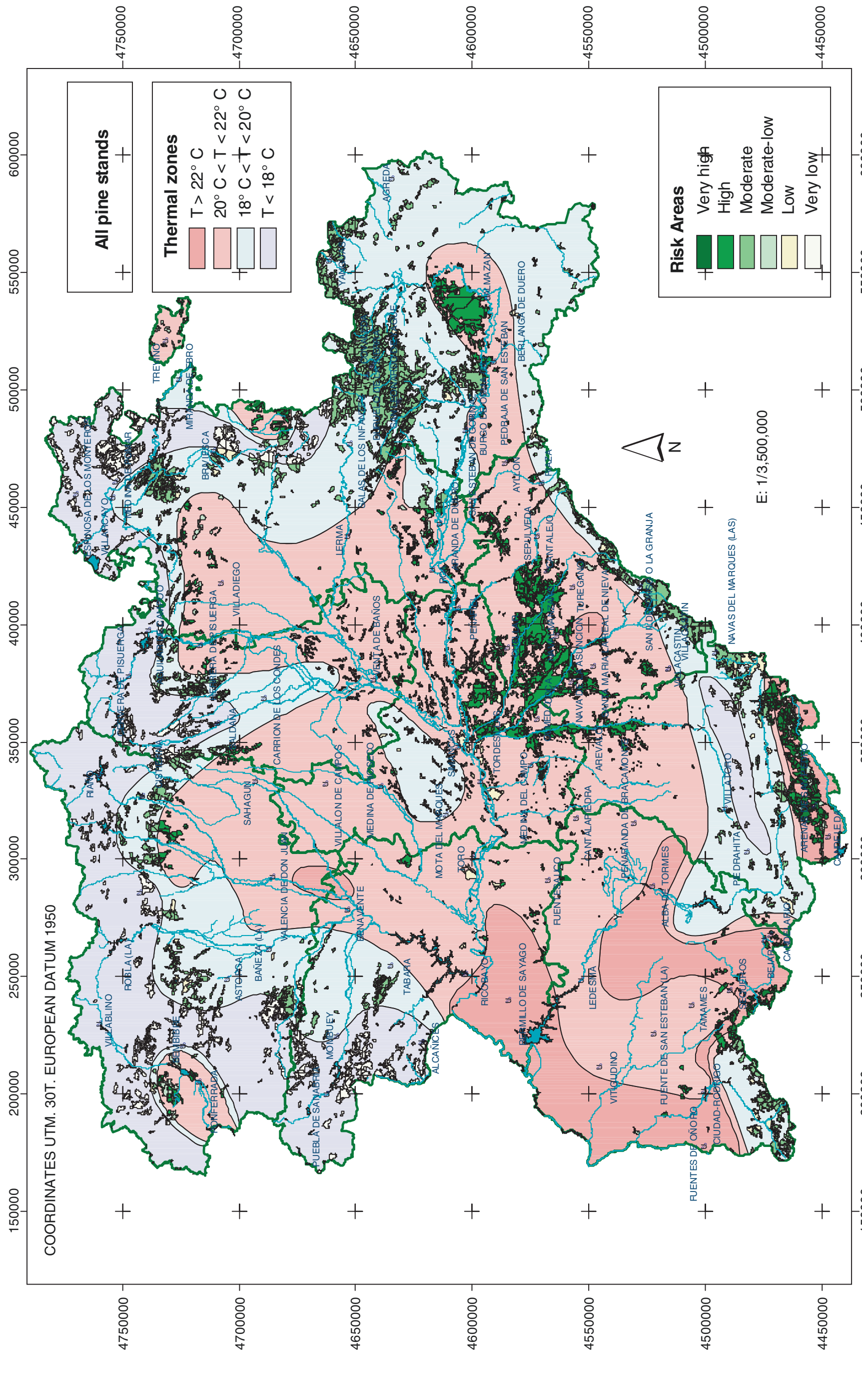

Fig. 17.3 Current risk levels of pine wilt disease in all pine stands in Castile and Leon

Table 17.4 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

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. 17.4, Table 17.4).

17.3.2.2 *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 17.5). 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. 17.5, Table 17.5).

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.

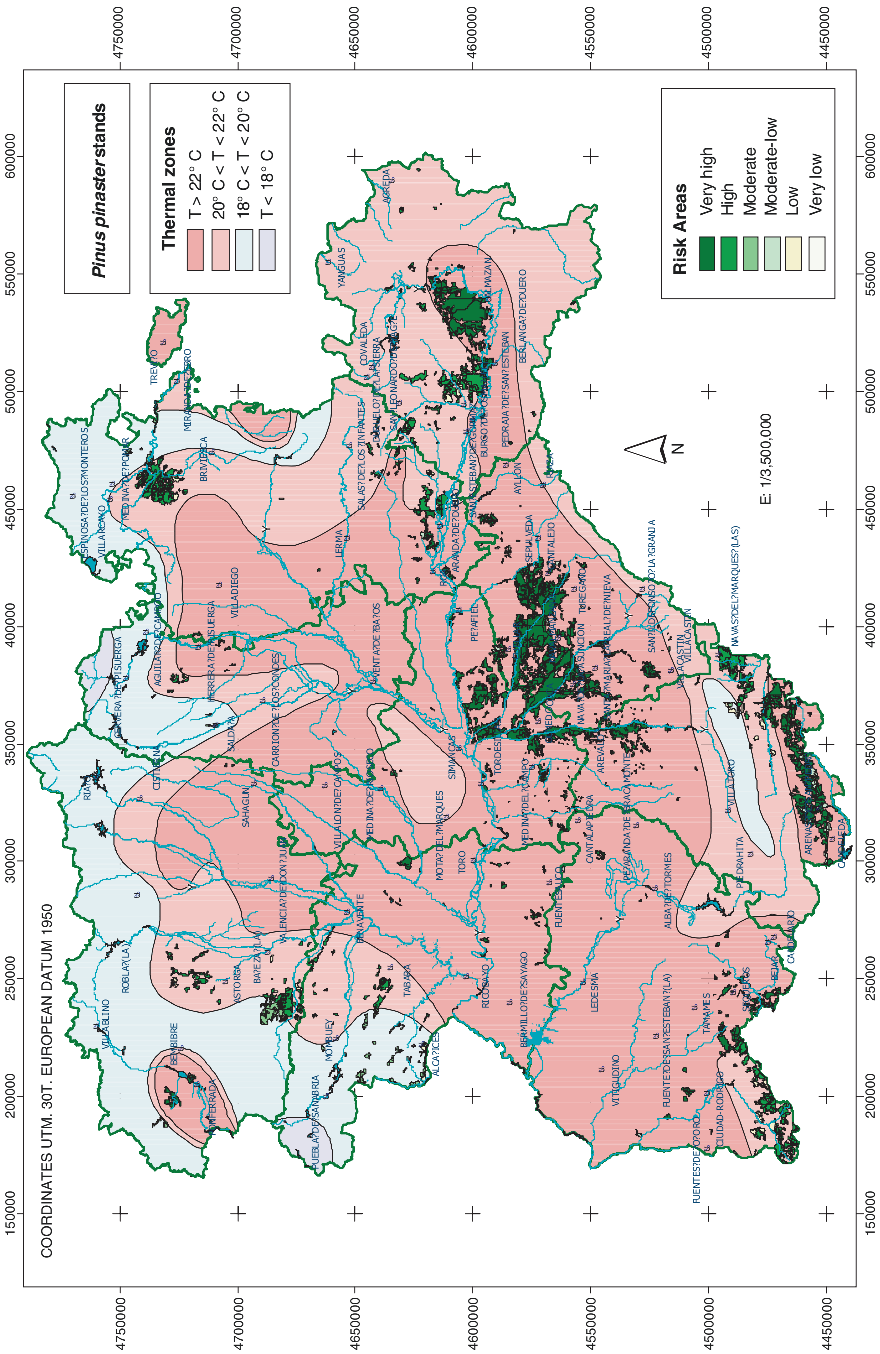

Fig. 17.4 Risk levels of pine wilt disease in *Pinus pinaster* stands in Castile and Leon. Predicted after 2 °C increase of mean July temperature

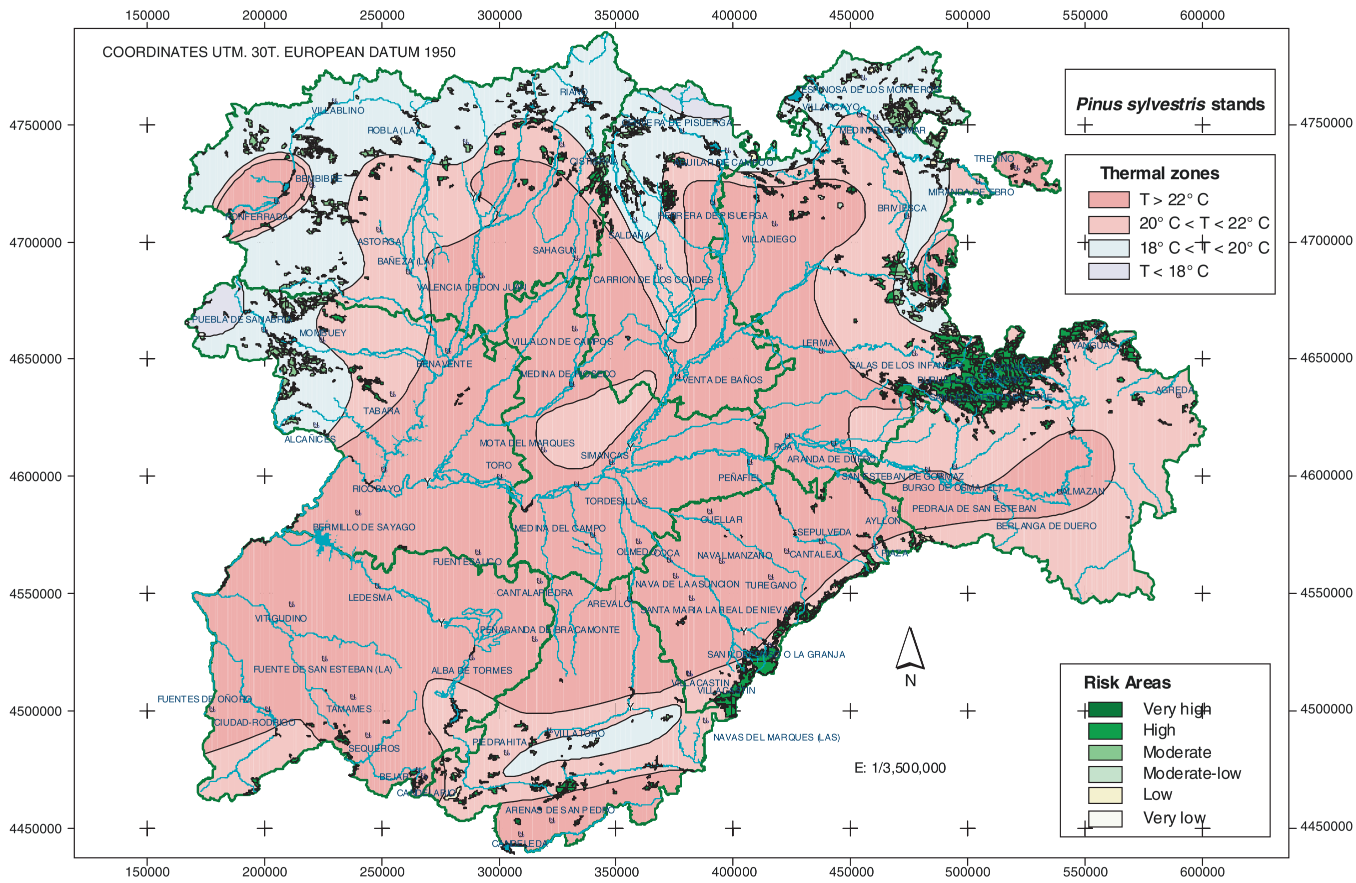

Fig. 17.5 Current risk levels of pine wilt disease in Pinus sylvestris stands in Castile and Leon

Table 17.5 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	5271 (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	3029 (0.8)	3029 (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

Table 17.6 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)	6163 (8.6)	−42.9
Moderate-low	4222 (5.9)	4222 (5.9)	0
Low	4797 (6.7)	9832 (13.9)	7.2
Very low	11,209 (15.7)	0	−15.7

17.3.2.3 *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 17.6). 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. 17.6). 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.

17.3.2.4 *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 9000 ha. All these pines appear in temperate valleys and plains, but their low specific susceptibility leads them to lower risk

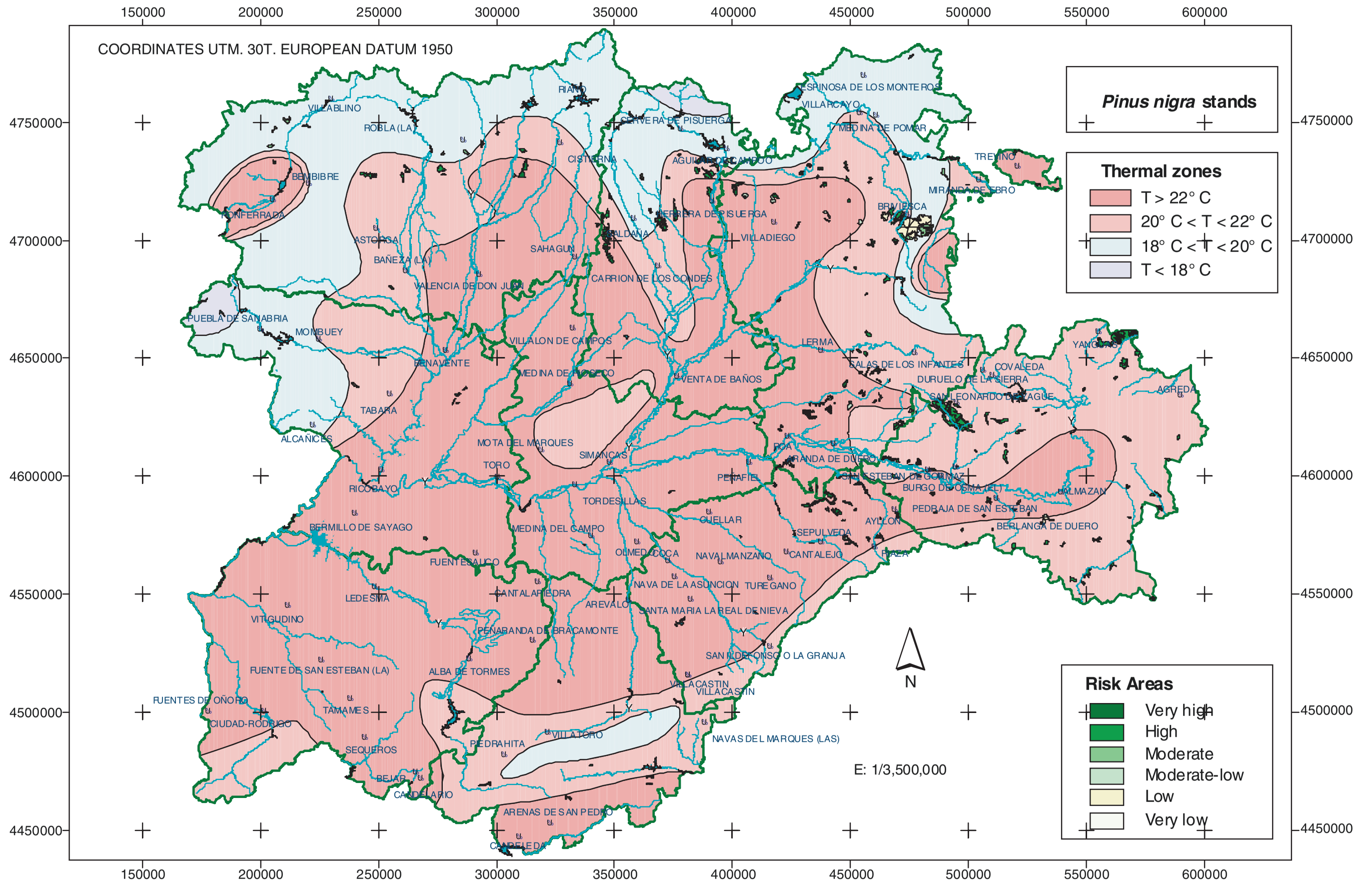

Fig. 17.6 Risk levels of pine wilt disease in Pinus sylvestris stands in Castile and Leon predicted after 2 °C increase of mean July temperature

Table 17.7 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	1524 (4%)	26,923 (78%)	74
Low	32,857 (96%)	7459 (22%)	−74
Very low	0	0	0
Pinus halepensis			
Moderate-low	18,761 (82.5)	18,761 (82.5)	0
Low	3845 (16.9)	3977 (17.5)	0.6
Very low	132 (0.6)	0	−0.6
Pinus radiata			
Moderate-low	4453 (51.2)	4458 (51.2)	0
Low	568 (6.5)	4251 (48.8)	42.3
Very low	3683 (42.3)	0	−42.3

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 17.7). 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 one 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.

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.

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Chapter 18
Soil Organic Carbon Sequestration Under Different Tropical Cover Types in Colombia

Flavio Moreno, Steven F. Oberbauer, and Wilson Lara

Abstract Soils are important to the global carbon cycle because they store the largest pool of carbon of terrestrial ecosystems, part of which could be released to the atmosphere as a result of land use changes. This study assesses how soil organic carbon stocks differ as a function of soil depth and the tropical forest successional state. Soil shafts 4 m-deep were excavated in six primary forests (PF), six young secondary forests (SF, 6–12 year), and four pastures (PAS) plots located in a watershed of the Porce River in the Colombian Andes mountains. Soils were sampled from the four walls of each shaft using a progression of 14 depths between 0 and 4 m depth. Clay content, soil bulk density, and C content were determined at each depth. In order to understand the soil organic carbon (SOC) change in these cover types, we modeled the data using non- linear equations and found significant differences in carbon content (Mg C 100 Mg^{-1} soil; 1 Mg = 10^6 g) and carbon stocks (Mg ha^{-1}) among cover types. The highest stocks of SOC to 4 m depth were found in soils of PF (227.9 Mg ha^{-1}), followed by SF (192.5 Mg ha^{-1}), and PAS (171.2 Mg ha^{-1}). These results suggest that converting primary forests to degraded pastures over several decades leads to substantial losses of soil organic carbon. Development of secondary forests in the sites of abandoned pastures starts to recover soil carbon stocks at the early stages of forest development. Differences of soil organic carbon inventories between PF and PAS were only 11.5 Mg ha^{-1} in the first 30 cm but 56.8 Mg ha^{-1}down to 4 m. These results strongly support the need to assess the effects of land use change on SOC by sampling along the whole soil profile. Monitoring the superficial layers alone could yield misleading conclusions.

F. Moreno (✉) • W. Lara
Departamento de Ciencias Forestales, Universidad Nacional de Colombia, Sede Medellín, Medellín, Colombia
e-mail: fhmoreno@unal.edu.co

S.F. Oberbauer
Department of Biology, Florida International University, University Park, Miami, FL 33199, USA

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18.1 Introduction

Soils play a central role in the dynamics of carbon in the biosphere because they contain the largest carbon stock in terrestrial ecosystems. The first one meter layer of soil has three times as much as the carbon in aboveground vegetation and twice as much as in the atmosphere (Batjes and Sombroek 1997; Schlesinger 1997). About 30 % of current carbon emissions by land use change comes from loss of soil organic carbon (SOC) (Houghton 1991, 1999).

The conversion of forests to create pastures results in the compaction of surface soil layers caused by organic matter decomposition, cattle trampling, and erosion processes, particularly if soils are unvegetated, overgrazed, or steeply sloped (Raich 1983; Cassel and Lal 1992; Townsend et al. 1995; Fearnside and Barbosa 1998; de Camargo et al. 1999). Compaction, erosion, and rapid organic matter mineralization from soils increase the bulk density of superficial layers (Fearnside and Barbosa 1998; Hillel 1998). The SOC inventory per square meter in any specific layer is the product of carbon content (Mg C Mg^{-1} soil), soil bulk density (Mg m^{-3}), and depth of the soil layer (m) (Rosenzweig and Hillel 2000), so increased bulk density associated with habitat conversion results in overestimations of C stocks in these layers. For this reason, differences in SOC between cover types are rarely detected and analyses are usually confounded when sampling is restricted to the superficial horizons, even if the C content between cover types is different (Fearnside and Barbosa 1998).

Studies on the effects of forest conversion to pastures on soil carbon have produced mixed results, ranging from increases, to no change, to decreases (Fearnside and Barbosa 1998). In a meta-analysis of the effect of land use changes on SOC, Guo and Gifford (2002) found that soil carbon stocks increased by 8 % when forests were replaced by pastures. In another review, Murty et al. (2002) found no significant change of soil carbon associated with conversion of forests to pastures. Other studies found a net decrease of SOC under pastures (VanDam et al. 1997; Fearnside and Barbosa 1998; Rhoades et al. 2000; Veldkamp et al. 2003; Neumann-Cosel et al. 2011). These contradictory results seem to stem from differences in sampling scheme (for example shallow vs. deep sampling), site differences (e.g., in clay content or slope), or differences produced by management and land use (changes in soil bulk density after land use change, the effect of intensive management on pastures, or pasture age (de Koning et al. 2003). However, under poor management, pastures are normally a net carbon source (Fearnside and Barbosa 1998).

The effects of land abandonment and the subsequent regrowth of secondary forests on soil carbon dynamics also seem confounded for the same reasons described above. Guo and Gifford (2002) did not find significant differences in carbon stocks between secondary forests and pastures, although values tended to be lower in forests than in pastures. De Camargo et al. (1999) found no differences in soil carbon stocks among a range of land cover types that included secondary forests, pastures, and primary forests, largely because of high variability in the data and uncertainties in C analyses at low contents deep in the soil. However, De Koning et al. (2003)

found considerably lower soil carbon stocks under old age pastures than under secondary forests (15.5 Mg ha^{-1} less C up to 50 cm depth). Rhoades et al. (2000) found an increase of 1.9 Mg ha^{-1} $year^{-1}$ of soil C under secondary forests regenerating from traditional pastures or sugar cane, with a return of total soil C stock to preclearing levels within 20 years. Neumann-Cosel et al. (2011) found similar stocks under pastures and young secondary forests (<15 year-old) and higher under pastures > 100 year-old, which suggests that increases of soil carbon through reforestation is difficult in the short term.

One important source of confusion about the effects of land use changes on SOC is due to differences in sampling depth. Generally, in studies where sampling was restricted to the first 100 cm of soil depth or less, there was an increase or no significant difference in SOC associated with the conversion of forests to pastures (Neill et al. 1997; Post and Kwon 2000; Guo and Gifford 2002; Murty et al. 2002; Desjardins et al. 2004). In contrast, studies conducted in tropical areas that addressed stocks and losses of SOC in deep soil profiles found that deep layers are dynamic pools of carbon, that are affected by land use changes (Trumbore et al. 1995; de Camargo et al. 1999; Jobbagy and Jackson 2000; Sommer et al. 2000; Veldkamp et al. 2003). Soil layers deeper than 1 m constitute a significant proportion of total soil volume in highly weathered tropical soils. Although contents below 1 m deep are low, they consistently contribute about 40–60 % of total SOC inventory and are responsible for a substantial proportion (perhaps even more than 50 %) of total C released by forest conversion (Trumbore et al. 1995). There is little scientific justification for considering only the first meter of the soil profile in calculation of soil carbon stocks, especially in histosols and deeply weathered tropical soils that can store significant amounts of carbon in deep layers (Batjes and Sombroek (1997).

The SOC pool results from the balance between inputs from litter, woody detritus, and dead roots and outputs from decomposition, leaching, and erosion (Sampaio et al. 1993; Delaney et al. 1997). Inputs of SOC are generally larger in forests than in other land covers (Richter et al. 1999; Post and Kwon 2000). When forests are removed, soil perturbations often increase carbon outputs from the soil (Batjes and Sombroek 1997; Fearnside and Barbosa 1998). The objective of our study was to assess the differences in total SOC stocks among three tropical forest successional cover types: lightly impacted primary forests (PF), young secondary forests (SF), and degraded pastures (PAS), in the Andes of Colombia. The second objective was to evaluate the variation of SOC with soil depth in each vegetation coverage class. We hypothesized that the three vegetation coverage classes studied contain significantly different quantities of carbon, with less carbon in soils of PAS as a result of less carbon inputs in relation to outputs. Soils of PF were hypothesized to store the highest carbon stocks. Soils of SF were hypothesized to be intermediate between PAS and PF because this cover type is supposed to restore carbon stocks to levels prevailing before disturbance. We also hypothesized that carbon distribution in the soil profiles was different among these vegetation coverage classes, reflecting the differential effect of each forest coverage type with depth.

18.2 Methods

18.2.1 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 central cordillera of the Andes of Colombia in the northeast of the state of Antioquia. The study area is within a 5000 ha land tract acquired as a watershed protection for the Hydroelectric Project Porce II, which started electricity generation in 2001 (Fig. 18.1).

The annual average rainfall in the center of the study area between 1990 and 2003 was 1927 mm (SD = 272 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 in June–August. The annual average temperature from 2001 to 2003 was 23.7 °C in the bottom of the valley at 1000 m.a.s.l. (meters above sea level). The study plots were located in the Tropical Premontane Moist Forest life zone (*sensu* Holdridge 1978) at an elevation from 1000 to 1300 m.a.s.l.

The study was carried out in 12 permanent plots where growth and carbon dynamics was being measured, six of them in young secondary forest (SF) and six in primary forests (PF). We also selected four areas in degraded pastures close to the forest plots. There were fewer PAS plots because lands were abandoned upon purchase, so it was hard to find active pastures in the area. Management in these

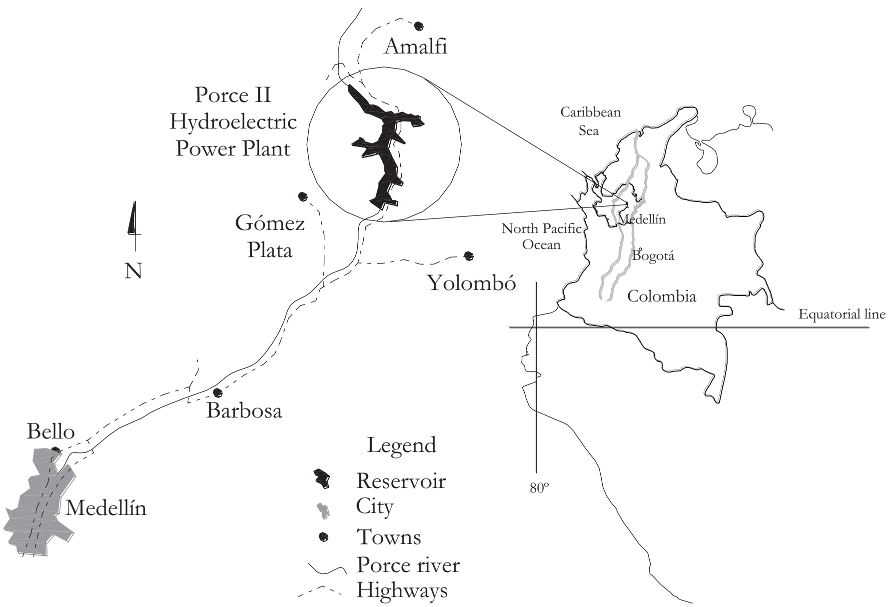

Fig. 18.1 Location of the study area

Table 18.1 Forest structure in plots of SOC inventory[a]

Variable	Cover type			
	PF		SF	
	Average	SD	Average	SD
Basal area/ha (m^2)	36.31	1.53	18.18	11.08
No. Individuals /ha (>5 cm Diameter)	8558	3228	9737	3808
C in above ground biomass (Mg/ha)	71.7	9.63	29.69	21.01

[a]Data based on 6 0.1 ha plots of PF and 6 0.05 ha plots in SF. C inventories were estimated from allometric equations developed for the study area, with determination of C content for each tree component (leaves, branches, wood, reproductive structures)

pastures has been minimal with no fertilizer application or burning, and only occasional hand weeding. Average basal area of PF plots was 36.3 m^2/ha (Table 18.1). Estimated ages of SF plots varied from 8 to 14 year. old and had an average basal area of 18.1 m^2/ha (Table 18.1). These plots were previously occupied by degraded pastures that were abandoned to natural succession upon land purchase. According to local inhabitants, pastures in this region were more than 50 year. old. Soils in the study area developed from igneous rocks composed mainly by quartz-diorite. They are well drained, very acidic (pH<5), with low natural fertility, and are classified as Ustoxic Dystropept (Jaramillo 1989).

18.2.2 Soil Sampling

We dug a rectangular shaft (80 x 120 cm) to 430 cm depth in each plot. As soil carbon content decreases exponentially with depth (Raich 1983; Trumbore et al. 1995; Batjes and Sombroek 1997; VanDam et al. 1997; Jobbagy and Jackson 2000), we used a progression of sampling intervals (Raich 1983) that resulted in more intensive sampling of superficial soil layers than deeper layers. We sampled soils at 0, 5, 10, 20, 30, 50, 75, 100, 150, 200, 250, 300, 350, and 400 cm. The four shaft walls per sampling depth were sampled, with two soil samples from each wall, one for bulk density and the other for chemical analyses. Samples of bulk density were taken by gently pressing a 5 cm diameter stainless steel cylinder horizontally into the soil. The sampling depth was controlled with a metric tape (± 1 mm precision) attached to the wall. All samples were packed in polyethylene bags and tagged.

18.2.3 Soil Analysis

Cores for bulk density were weighed after drying at 101 ± 1 °C for 72 h. (Elliott et al. 1999). The four samples for chemical analyses per depth level were composited into a single sample using the same amount of soil from each sample. Samples were air-dried, ground with mortars, and sieved through a 200 µm mesh, to eliminate fine roots from the samples, which were being accounted for independently, as well as to remove coarse fragments and litter. Air drying is a convenient and widely used procedure of soil processing as much as air-dried soil has relatively constant weight and minimal biological activity (Boone et al. 1999). Soil texture was determined by the hydrometer procedure (Singer and Munns 1999). Carbon content was determined in a NC 1500 Carlo Erba CHN analyzer (Carlo Erba Instruments, Milan, Italy). Duplicate determinations of C content were performed in three shafts of PF and three of SF to check the accuracy of determinations.

18.2.4 Data Analysis

We averaged the four values of bulk density per depth in each shaft. We used the carbon content and bulk density in each sampling depth to estimate the carbon content per centimeter of soil depth at the respective depth according to the following equation (Rosenzweig and Hillel 2000),

$$SOC = a\,p\,C\,d \tag{18.1}$$

Where

SOC: organic carbon in a soil layer 1 cm depth (Mg ha^{-1}).
a: Area (1 ha = 10,000 m^2).
p: Average soil bulk density at any specified depth (Mg m^{-3}).
C: Carbon content in soil (Mg C/100 Mg soil).
d: Depth of soil layer. Because we estimated carbon mass per centimeter of soil depth, *d*=0.1 m.

We conducted analysis of variance (ANOVA) of bulk density and clay content data according to a completely randomized split plot design, using the mixed procedure of the SAS System version 9 (The SAS Institute Inc., Cary, NC). The large plots in the analysis were assigned to vegetation cover types, and soil depths were the subplots. Residuals were tested for the ANOVA assumptions of normal distribution (Kolmogorov-Smirnov D statistic), independence (inspection of residual plots) and heteroscedasticity (differences of extreme variances less than 3 when grouped by depth) (Dean and Voss 1999; Quinn and Keough 2002).

Normally the C stock to any soil depth has been estimated by summing the C inventories in the successive soil layers, each one estimated with Eq. 18.1 according to the layer width. This approach implicitly assumes that C content is the same along each soil layer, which typically is not true, and could introduce errors when layer thicknesses are large. Instead, we treated both C content and SOC as continuous variables. To estimate the total amount of C stored to any depth in the soil profile, we did curve-fitting with models whose first derivative had a similar trend as our data set. In the case of SOC, the fitted line represents the rate of change of the amount of carbon per cm depth. Hence, the function that described this trend can be considered as the derivative of the cumulative function of carbon stored along the soil profile. To model this trend we tested the derivative of different nonlinear accumulative, increasing, asymptotic models, such as Weilbull, Gompertz, Chapman-Richards, Bayley, Bertalanffy, Mitscherlich, and Levakovic I (Zeide 1993; Vanclay 1994; Schabenberger and Pierce 2002). We also tested the hyperbolic model, also called Langmuir model (Schabenberger and Pierce 2002). We looked for asymptotic models because they would allow us to estimate the maximum amount of carbon that could be stored in the entire profile of any specific soil. Models were evaluated using the p-value, R^2, Durbin-Watson statistic of autocorrelation, and residual plots. The best fit was obtained with the derivative of the Levakovic I model, which has the following expression,

$$SOC = b_1\, b_2\, b_3\, b_4\, x^{-b4-1} \left(\frac{x^{b4}}{b_2 + x^{b4}}\right)^{b3+1} \tag{18.2}$$

where,

SOC is the organic carbon in a soil layer 1 cm depth (Mg ha^{-1}) at depth *x*.

x: Soil depth (cm).

b_1, b_2, b_3, b_4: Constants. b_1 is the asymptote or maximum amount of carbon stored in a given soil.

The total amount of C stored from the surface to any depth can be estimated by the integral of Eq. 18.2, which has the following expression,

$$C = b_1 \left(\frac{x^{b4}}{b_2 + x^{b4}}\right)^{b3} \tag{18.3}$$

where,

C: Total amount of carbon (Mg ha^{-1}) stored to depth *x*.

We used Eq. 18.2 to fit the data of C content (Mg C/100 Mg soil) as well as to estimate the SOC stocks in a soil layer 1 cm depth (Mg ha^{-1}) by weighted non-linear regression (nlin procedure, SAS). Because Eq. 18.2 is not defined when x=0, we entered a value of 1 cm for data taken on the soil surface. Comparisons between covers were done through sum of squares reduction tests (Schabenberger and Pierce 2002).

Table 18.2 Averages and standard deviations of bulk density and clay content in lightly impacted primary forests (PF), young secondary forests (SF), and degraded pastures (PAS)

Variable	Cover type		
	PF	SF	PAS
Bulk density (Mg/m^3)	1.393 (0.060)	1.418 (0.079)	1.339 (0.052)
n	84	84	56
Clay content (%)	31.91 (12.28)	19.75 (12.14)	24.96 (11.81)
n	84	84	56

18.3 Results

18.3.1 Soil Bulk Density

Average of bulk densities along the whole soil profile ranged from 1.42 for SF to 1.34 for PAS (Table 18.2). Kolmogorov-Smirnov tests for normality showed that bulk density was normally distributed ($D=0.039$, $p>0.15$). Differences in bulk density among cover types were not significant (Table 18.2). However, variation of bulk density with depth was highly significant ($F_{(13, 163)} = 25.19$, $p<0.0001$), as was the interaction cover type x depth ($F_{(26, 163)} = 2.80$, $p<0.0001$), indicating that the trend of bulk density change with depth was different among cover types. Pair wise comparisons by the Tukey-Kramer test showed that at the surface (depth = 0 cm) PF had lower bulk density than SF ($p<0.003$) and PAS ($p<0.046$). At 10 cm depth PF also had lower bulk density than SF ($p<0.018$). Differences among vegetation covers were not statistically significant at other depths (Fig. 18.2).

18.3.2 Soil Organic Carbon Inventory.

The content of organic carbon, expressed as a percentage of soil mass (C%) and the soil carbon stores in layers 1 cm depth ($Mg\ ha^{-1}$), showed a clear decreasing trend with increasing depth in all cover types. Highest values occurred at the surface and decreased approaching an asymptote of zero as depth increased. To obtain an independent estimation of coefficients b_1 to b_4 for each cover in a unique analysis framework, we fit Eq. 18.2 to the total set of observed data for C (%) and C stores in layers 1 cm depth ($Mg\ ha^{-1}$) using the nlin procedure of SAS (Schabenberger and Pierce 2002). The model was highly significant for both variables (C% and carbon stores in $Mg\ ha^{-1}$ ($p<0.0001$) with R^2 values higher than 91 %. This analytical approach had the advantage of letting us evaluate the difference of curves among cover types via sum of squares reduction tests, which evaluates the increase of the sum of squares residual (SSR) of the full model (i.e., the model with 12 parameters, four for each cover type) when the effect of cover is dropped from the model (reduced

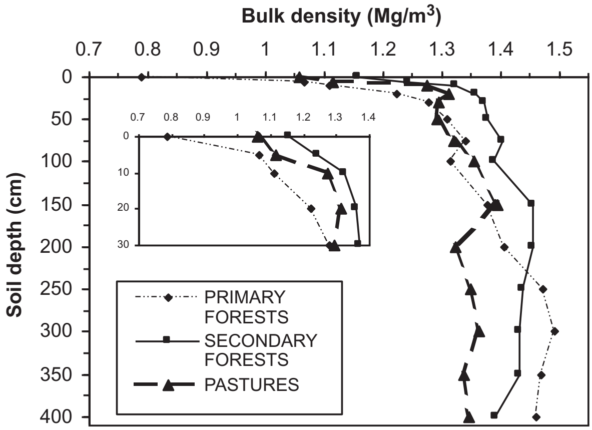

Fig. 18.2 Average soil bulk density per cover type in each sampled depth. The inset shows data close to the soil surface (0–30 cm)

model, no cover included, four parameters). The statistic has an F distribution and was calculated according to Schabenberger and Pierce (2002).

Because residuals were larger at the surface and smaller at greater depths, we used weighted non-linear regression (Schabenberger and Pierce 2002), with $(\text{depth})^{1.2}$ as the weight variable to correct for heteroscedasticity. The power value of 1.2 was determined by iteration to provide the best correction of heteroscedasticity.

The sum of squares reduction test showed that the effect of cover types was highly significant for both variables ($p< 0.0001$). Comparisons among covers via orthogonal contrasts, using the procedure nlmixed of SAS revealed that curves of carbon content (%) along soil depth were different among the three cover types studied ($p<0.0001$, results not shown). Soil carbon content in PF was greater than that of SF and PAS; in SF it was slightly greater than in PAS between the soil surface and 1 m depth, but below 1 m they were practically identical. Because C content changes with depth, to assess the average size of SOC differences we estimated the average value of each function, which is the integral of the function evaluated between the points of interest. We then used the estimated parameters to solve the Eq. 18.3 over the sampling interval (0–400 cm depth) and obtained the average for each vegetation cover type. The average difference of C content (Mg C/100 Mg soil) between PF and SF was 0.116 % and between PF and PAS was 0.137 %, while that between SF and PAS was only 0.021 % (Table 18.1).

Comparisons among vegetation cover types for soil carbon stores in layers 1 cm depth (Mg ha^{-1}) were also performed via orthogonal contrasts. We detected no differences between the curves of carbon mass along soil depth of PF and SF, but differences existed between these two covers and the PAS (PF vs. SF ($F_{(1, 203)} = 2.21$, $p>0.1382$) and PF-SF vs. PAS ($F_{(1, 203)} = 23.15$, $p<0.0001$)). Curves of PF and SF were very close in the first 30 cm of soil, and then the curve of SF clearly drifts toward the curve of PAS with lower C stores than PF at every depth. In the very surface soil (first 4 cm), the curve of SF was above that of PF (Fig. 18.3).

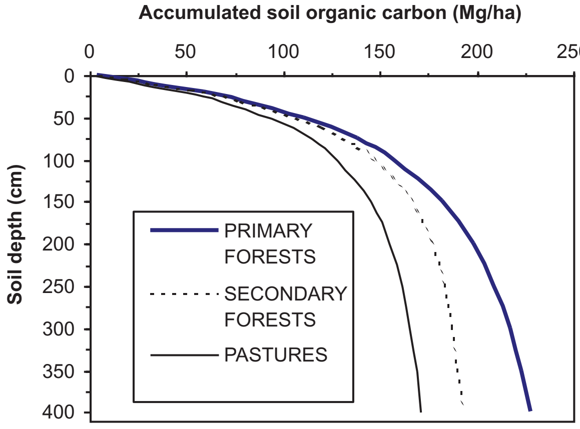

Fig. 18.3 Estimated soil carbon inventory with depth in three vegetation cover types

Table 18.3 Mean value of carbon content (%) between 0 and 400 cm-depth and estimated coefficients of Eq. 18.2 for organic carbon stocks (Mg ha^{-1} cm^{-1}) in soils of lightly impacted primary forests (PF), young secondary forests (SF), and degraded pastures (PAS). b_1 is the asymptote or maximum amount of carbon stored in a given cover type (Mg ha^{-1})

Cover type	Mean C%	b_1	b_2	b_3	b_4
PF	0.472	265.7	88.77	0.9426	1.0381
SF	0.356	204.8	355.60	0.6493	1.3648
PAS	0.335	188.4	80.75	0.9779	1.1123

Table 18.4 Soil carbon inventories in the studied covers estimated down to different depths (Mg ha^{-1}) and percentages with respect to lightly impacted primary forests (in parenthesis)

Variable	Cover type		
	PF	SF	PAS
Total C down to 10 cm	33.04	33.37 (101 %)	27.20 (82.3 %)
Total C down to 30 cm	79.45	77.94 (98.1 %)	67.95 (85.5 %)
Total C down to 100 cm	157.22	147.21 (93.6 %)	128.28 (81.6 %)
Total C down to 400 cm	227.93	192.52 (84.4 %)	171.18 (75.1 %)

By solving Eq. 18.3 with the parameters of Table 18.3, we estimated the amount of carbon accumulated from the surface to any depth (Table 18.3, Fig. 18.3). Differences of carbon stores were only 1.5 Mg/ha between PF and SF and 11.5 Mg/ha between PF and PAS in the upper 30 cm. Differences became progressively larger at greater depths (Table 18.4, Fig. 18.3). Total SOC inventory down to 4 m depth was 227.9 Mg/ha in PF, 192.5 Mg/ha in SF, and 171.2 Mg/ha in PAS (Table 18.4); therefore, differences down to 4 m depth were 35.4 and 56.7 Mg/ha between PF-SF and PF-PAS, respectively.

18.4 Discussion

Primary forests are the natural land cover in the study area in the absence of anthropogenic disturbances. Once natural forests were cleared, pastures were established after transitory crops and lasted for many decades until lands were abandoned. Secondary forests develop after pasture abandonment and eventually return to the predisturbance condition of primary forest.

Assuming that soils sampled in this study represent a chronosequence (i.e. a sequence of related soils that differ from one another in certain properties primarily as a result of time elapsed under an specific land use) where PF is the predisturbance condition as well as the final result of the successional process after pasture abandonment, PF can be used as a reference for comparing the soil carbon changes under the other land covers. Also, as the pastures in this study were old and degraded, we can assume they represent the typical state of soils under unmanaged pastures in the study area. Hence, they can be used to assess the changes associated with secondary forests through natural succession.

Average differences of carbon content between PF and SF (0.116 %), PF and PAS (0.137 %), and SF and PAS (0.021 %) do not seem to be large. However, these values mean that average carbon content of PF soils (0.47 %) was 40.9 % larger than that of PAS (0.34 %) and 32.6 % larger than in SF (0.36 %). The fact that differences in soil carbon content between PF and SF were smaller than difference between PF and PAS suggests that secondary succession replenishes soil carbon stocks toward the predisturbance levels found under PF. These results also suggest that abandonment of pastures has the potential to substantially increase soil carbon content (about 40 %) once the structure of secondary forests becomes similar to that of PF. On the other hand, results shown here suggest that substantial losses of soil carbon occur as a result of forest clearing and establishment of degraded pastures.

Differences of C inventory to 4 m deep between PF-PAS, PF-SF, and SF-PAS were 56.8, 35.4, and 21.3 Mg/ha, respectively (Table 18.4). Differences of carbon stocks in aboveground biomass between PF and SF (42.4 Mg/ha, Table 18.1) are similar to differences of soil carbon inventories shown above and suggest that soil carbon is a highly dynamic component of carbon cycle in forests. Data of C content in aboveground biomass of pastures studied is 3.3 Mg/ha (Toro Jaddy, unpublished), which shows that vegetation is a much dynamic component of C stock than soil in terrestrial ecosystems.

Estimated carbon inventories shown here are soil volume-based estimations, so when bulk densities among cover types are different, more soil mass is being sampled in the soil with the higher bulk density, hence biasing the carbon inventories. For example, our carbon inventories for the upper soil layers of PAS and SF could be overestimated because of the increased bulk densities in these layers. Even though carbon contents were consistently lower in the whole profiles under SF and PAS than under PF, carbon stocks were very similar between PF and SF in the first 50 cm depth and between PF and PAS in the first 20 cm (Fig. 18.3). This is a common situation when the study is restricted to the surface layers of soil, with increased bulk density by compaction in the more degraded soils, such as those under pastures (Fearnside and Barbosa 1998; Veldkamp et al. 2003). Compaction of surface layers might be responsible for the failure to detect differences in soil carbon stocks among cover types in such studies.

These artifacts introduced by compaction of surface layers in more degraded soils could also be present in our data; however, our large sampling depths should have diluted their effect. The difference in average bulk density between PF and SF in the whole profile equaled 1.8 %. Hence, if the increase in bulk density under SF was produced by soil compaction, the soil mass in the sampling depth of 4 m in PF would equal the soil mass to 3.92 m depth under SF (4 m – 4 m*0.062). However, the carbon accumulated to that depth in SF is 192.2 Mg/ha, only 0.32 Mg less than the amount to 4 m (192.5 Mg/ha). The same estimation for PAS shows that its soil mass equals that of PF at a depth of 421.6 cm, in which case the total carbon inventory of PAS soil is 172.07 Mg/ha, only 0.89 Mg more than the estimate to 4 m depth. For this reason we did not correct our estimates by bulk density.

Clay distribution along the soil profile differed among cover types, with an increase from soil surface to 100 cm in PF and SF, and then a decrease in deeper soil. In PAS there was no clear trend. Also, average differences of clay content among cover types were close to significant, with the highest value in PF followed by PAS and SF (Table 18.2). We expected more similar results of clay content because sites for location of shafts were carefully chosen to meet homogeneous characteristics. Every SF shaft had one PF or PAS shaft within a distance of less than 500 m; also the landscape, topography and soil type was very similar between them. Clay content is positively associated with SOC (Feller and Beare 1997; Fearnside and Barbosa 1998; Jobbagy and Jackson 2000; de Koning et al. 2003). Hence, the variation in clay content could bias the conclusions about the effect of land use changes. However, applying a correction for clay content is based on the assumption that soil clay content is not affected by land use change. However, this assumption does not seem reasonable because deforestation could affect clay content in a number of ways, such as soil erosion or transport of clay particles downward in the soil profile (Fearnside and Barbosa 1998). Analysis of covariance of total soil carbon stocks among vegetation cover types using the average clay content in each soil profile as a covariate revealed that this variable was not significant (results not shown).

Stocks of SOC to 4 m depth were highest PF, intermediate in SF, and lowest in PAS. From the chronosequence perspective (i.e., any time in the past lands of all cover types studied here had primary forests), we interpret the differences between PF and PAS as a net loss of 24.9 % of SOC stock of PF (56.8 Mg/ha) after more than five decades following forest removal. About half of this amount (27.81 Mg/ha) was lost below 1 m depth. Losses in the first 30 cm were only 11.5 Mg/ha, which suggests that carbon losses from conversion of forests are larger at greater soil depths. Hence deep layers of soil carbon are highly dynamic.

Secondary forests seem to have already started the recovery of SOC stocks of PF, as much as their SOC stock are higher than that under PAS in an amount equal to 9.3 % of the PF stock (21.3 Mg/ha). Differences of carbon stores between PF and SF are practically negligible in the 0–30 cm layer (0.33 and 1.51 Mg/ha at 10 and 30 cm, respectively). At 1 m depth, differences of 10.01 Mg/ha, as compared to those between PF and PAS of 28.94 Mg/ha for the same depth, suggest that after a short period of secondary succession (around 10 years) soil carbon in the superficial layers has been actively replenished. The inferred effect of secondary vegetation in the recovery of SOC at deeper layers is almost negligible (a difference of 25.4 Mg/ha between PF and SF, almost identical to that between PF and PAS of 27.81 Mg/ha).

To see where in the soil profile this recovery of C stock took place under SF, and to quantify losses of C from PF in relative terms, we estimated the differences in C content along SF and PAS profiles with respect to PF and expressed them as a percentage of PF C content at each sampling point. Our results are consistent with other studies that the greatest C losses occur below the first 30 cm of soil (Trumbore et al. 1995; Fearnside and Barbosa 1998; de Camargo et al. 1999; Jobbagy and Jackson 2000; Sommer et al. 2000; de Koning et al. 2003; Veldkamp et al. 2003). The smallest losses of C content occurred between 10 and 30 cm both in SF and PAS (13.9 %–16.5 % in SF and 17.8 %–195 % in PAS). Carbon losses increased both in top layers (0–10 cm) and deeper ones (>30 cm) to a maximum at 4 m depth of 41.5 % in PAS and 52.3 % in SF. The increased carbon losses in the 0–10 cm layer can be explained by the lower input of plant debris under SF and PAS as compared to PF, which has the largest source for litter and woody detritus. Production of fine roots has been reported to be lower in the top 10 cm of degraded pastures soil than in PF, but no differences occurred at greater depths (Nepstad et al. 1994). Similarly, no differences of fine root biomass (0–25 cm soil) were found among forests of three successional stages from 10 year.-old SF to PF (Cavelier et al. 1996). We also did not find significant differences of root growth between PF and SF in the 0–30 cm soil layer in the same plots (Moreno Flavio, unpublished data). Similar fine root production among cover types down to 30 cm depth could minimize significant carbon losses in the 10–30 cm soil layer of SF and PAS.

The most significant carbon losses occur below 30 cm, with progressively larger losses at greater depths. Carbon losses under PAS are greater than under SF from 0 to 150 cm depth; in deeper soil the trend is reversed. From 0 to 150 cm deep average clay contents of SF and PAS are very similar; however, clay content is clearly lower in PAS than in SF below 150 cm. Though clay content did not differ among the three cover types when analyzed for the whole soil profile, these trends suggest that mineralogy influences the soil carbon storage, with more carbon in soils with higher clay content (Feller and Beare 1997; Fearnside and Barbosa 1998; Jobbagy and Jackson 2000; de Koning et al. 2003). However, the action of SF in recovering soil carbon in the shallower layers of soil above 1.50 m depth seems clear. Furthermore, relative losses estimated at greater depths are more error-prone because of the smaller absolute carbon contents at those depths (de Camargo et al. 1999).

Few studies have used models to estimate the soil carbon storage as a function of depth. Jobbagy and Jackson (2000) advocated the use of mathematical functions to improve the assessment of SOC budgets at depth. They used logarithmic models and the Beta function to predict both the content and inventory of carbon. However, they found that logarithmic models overestimate the SOC. Mendham et al. (2003)

fit an exponential regression to the distribution of C (Mg ha^{-1} cm^{-1}) with depth. Among advantages of these kinds of models are that they help prediction and allow comparisons between studies made at different depths.

One important characteristic of the approach shown here is the estimation of the maximum amount of carbon stored in a given soil, which is the asymptote of Eq. 18.2 (Table 18.3). Even though this is an extrapolation of the actual sampled interval, it has the potential as a useful tool for improving the estimations of the total carbon stored in the whole soil profile and correcting significant underestimations in soil carbon inventories. Asymptotes as compared to the inventories down to 4 m depth (227.9, 192.5, and 171.2 Mg/ha, for PF, SF, and PAS respectively), suggest that despite the very low carbon contents at great soil depths, additional carbon is stored below 4 m. Results from the Brazilian Amazon in soil shafts of 8 m depth show important carbon inventories below 4 m depth (Trumbore et al. 1995; de Camargo et al. 1999).

Maximum carbon stocks estimated from asymptotes in SF and PAS represented 77.1 % and 70.9 % of those under PF. Stocks down to 4 m in SF and PAS were 84.5 % and 75.1 % of those under PF; at 1 m depth they were 93.6 % and 81.6 %; and at 30 cm were 98.1 % and 85.5 %, respectively (Table 18.4, Fig. 18.3). This progression suggests that our capacity to detect differences between vegetation covers increases as the studied depth also increases. Thus, studies performed above 50 cm (Raich 1983; Veldkamp 1994; de Koning et al. 2003; Desjardins et al. 2004) or even 1 m depth are probably inadequate to detect differences in the soil carbon inventory.

Dividing Eq.18.3 by the asymptote (parameter b_1) allows the estimation of the relative size of carbon stock as a function of soil depth (Fig. 18.4). This tool allowed

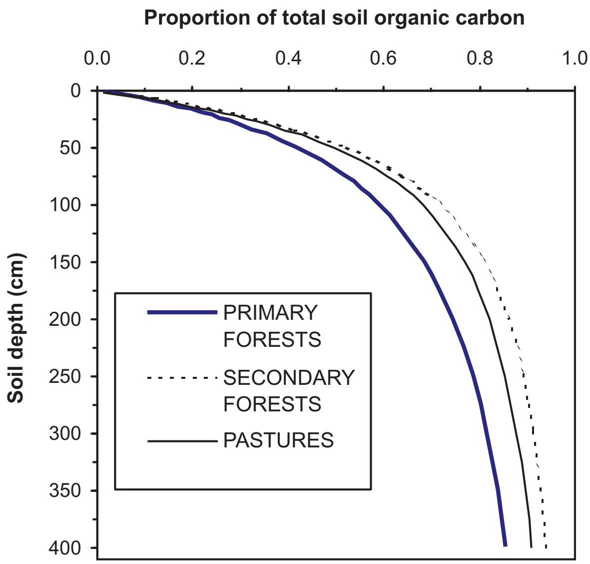

Fig. 18.4 Relative amount of soil carbon along soil profile with respect to the maximum amount in each cover type

the comparison, in relative terms, of the carbon storage between soils with different carbon inventories. For example, C inventory to 30 cm equals 29.9 % of total stock in PF, 38.1 % in SF, and 36.0 % in PAS (Fig. 18.4). Similarly, down to 1 m, there was 59.2 %, 71.9 %, and 67.9 % of total C in PF, SF, and PAS respectively. For this reason, sampling to any fixed depth in different cover types accounts for different proportions of the soil carbon stock. Based on these estimations, monitoring the same proportion of the carbon stock requires deeper samples in PF (Fig. 18.4). For example, 50 % of the total carbon stock was reached at a depth of 70 cm in PF, 46 cm in SF, and 50 cm in PAS (Fig. 18.4).

18.5 Conclusions

Soil organic carbon content (Mg C/100 Mg soil) was consistently higher from 0 to 4 m deep in PF than in SF and PAS. Differences in percent carbon were 0.116 % between PF and SF, 0.137 % between PF and PA, and 0.021 % between SF and PAS. These differences accounted for a difference of soil organic carbon inventory of 35.4 Mg/ha between PF and SF, 56.8 Mg/ha between PF and PAS, and 21.3 Mg/ha between SF and PAS. From a chronosequence perspective, these differences suggest a net loss of 24.9 % of SOC stock of PF after more than five decades under degraded pastures. About half of this amount (27.81 Mg/ha) was lost below 1 m depth; losses in the first 30 cm were only 11.5 Mg/ha. Deep layers of soil carbon seem to be highly dynamic because carbon losses from conversion of forests were larger at greater soil depths. These results emphasize the importance of sampling along the whole soil profile to assess the effects of land use change on SOC.

SOC stocks were higher in SF than under PAS in an amount equal to 9.3 % of the PF stock. Differences of carbon stores between PF and SF were practically negligible from 0 to 30 cm depth (0.33 and 1.51 Mg/ha at 10 and 30 cm, respectively). Differences of 10.01 Mg/ha from 0 to 1 m depth between these two covers, as compared to those between PF and PAS of 28.94 Mg/ha at the same depth, suggest that after a short period of secondary succession (around 10 years) soil carbon in the surface layers had been actively replenished. The effect of secondary vegetation in the recovery of SOC at deeper layers is almost negligible (a difference of 25.4 Mg/ha between PF and SF, almost identical to that between PF and PAS of 27.81 Mg/ha).

Our results show that the conversion from primary forest to old degraded pasture leads to substantial losses of soil organic carbon to the atmosphere, particularly from deeper soil levels. As secondary succession takes place in abandoned pastures, recovery of lost soil carbon stocks begins from the earliest stages of secondary forest development, primarily at shallow soil levels.

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Chapter 19
Modelling of Carbon Sequestration in Rubber (*Hevea brasiliensis*) Plantations

Engku Azlin Rahayu Engku Ariff, Mohd Nazip Suratman, and Shamsiah Abdullah

Abstract The issues of global warming and climate change have been widely debated over the past few decades by researchers, managers and leaders. It was estimated that the annual global air temperature may increase by approximately 2.5 °C by the end of the century (NAST 2000). In 1992, the United Nations Framework on Climate Change (UNFCCC) produced an international environmental treaty at the United Nations Conference on Environment and Development (UNCED), held in Rio de Janeiro, Brazil. The objective of the treaty was to stabilize greenhouse gas (GHG) concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Carbon dioxide (CO_2) is one of GHGs that contributes the highest emission (Dulal and Akhbar 2013). Malaysia remains committed to climate change agenda and recently introduced a new national policy on climate change and green technology. The country has recently passed a renewable energy act and reaffirms its commitment to a pledge during Rio Earth Summit and signed the UNFCCC and currently listed into non-Annex 1 countries.

19.1 Introduction

The issues of global warming and climate change have been widely debated over the past few decades by researchers, managers and leaders. It was estimated that the annual global air temperature may increase by approximately 2.5 °C by the end of the century (NAST 2000). In 1992, the United Nations Framework on Climate Change (UNFCCC) produced an international environmental treaty at the United Nations Conference on Environment and Development (UNCED), held in Rio de Janeiro, Brazil. The objective of the treaty was to stabilize greenhouse gas (GHG) concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. Carbon dioxide (CO_2) is one of GHGs

E.A.R. Engku Ariff • M.N. Suratman (✉) • S. Abdullah
Faculty of Applied Sciences, Universiti Teknologi MARA (UiTM), 40450 Shah Alam, Malaysia
e-mail: nazip@salam.uitm.edu.my

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that contributes the highest emission (Dulal and Akhbar 2013). Malaysia remains committed to climate change agenda and recently introduced a new national policy on climate change and green technology. The country has recently passed a renewable energy act and reaffirms its commitment to a pledge during Rio Earth Summit and signed the UNFCCC and currently listed into non-Annex 1 countries.

Carbon sink refers to any vegetation that has net absorption of CO_2 from the atmosphere. The process by which carbon sinks remove CO_2 from the atmosphere is known as carbon sequestration (Suratman 2008, 2014). Since the last decades, much of the research conducted on carbon sequestration has been in natural vegetation, but little attention has been given to man-made landscapes and highly managed systems such as forest or agricultural tree crop plantations (Whittinghill et al. 2014). As rubber tree is one of major agricultural tree crop plantation species in Malaysia, their roles in mitigating climate change through carbon sequestration should not be overlooked. The Malaysian government formulated its research program through the Malaysian Rubber Board (MRB) with an objective of promoting a modern and productive rubber plantation sector through the implementation of appropriate crop management technologies. Numerous researches have been conducted to improve latex and timber production through tree breeding programs. For instance, a new latex timber clones (LTC) was developed with the purpose of increasing latex and timber yield production. The clones are fast growing and have the potential to improve latex and timber yield production up to 3500 kg/ha of latex and 1.5 m^3/stem of rubberwood log, respectively (MRB 2002).

The photosynthesis process is studied and modelled the most by previous studies which related directly with effect of carbon dioxide. According to Frank (2004), the predictive relationship between leaf photosynthesis and growth does not have much evidence. However, biomass production is closely related to light interception, which is mainly determined by leaf area index (LAI) and productivity (Guo 2007). Biomass and LAI are important variables in many ecological and environmental applications (Heiskanen 2005). LAI refers to one half of the total leaf area per unit ground surface area (Chen and Black 1991), and it controls many biological and physical processes in the water, nutrient and carbon cycle (Waring and Running 1998). There are studies concerning LAI and biomass which demonstrated that photosynthesis, respiration, and dry matter accumulation could be expressed as a function of LAI (Hodges and Kanemasu 1977), higher radiation interception was linked to high LAI resulting in greater biomass production (Singels and Donaldson 2000). In addition, a few studies used LAI as a variable to estimate biomass of the mountain birch forests using remote sensing method. Starr et al. (1998), Bylund and Nordell (2001) and Dahlberg et al. (2004) have studied the biomass and developed allometric relationships to approximate the biomass of a tree component or the total biomass of single trees but they preferred some more easily measured variable, such as DBH or height. However, this study used ground measurement method to identify the suitability of LAI in estimating biomass or carbon sequestration in the development of allometric equations.

Several studies have demonstrated that agricultural and forestry sectors could mitigate significant quantities of GHG emissions through reforestation

(Rojo-Martinez et al. 2005; Wauters et al. 2008; Lewandrowski et al. 2014). While the important role of forest plantations in strengthening the socio-economic has been established, their roles in ecosystem functions are not fully understood and well documented. Rapid responses are needed to meet the increasing challenges of the industry as they unfold. Therefore, a study on the tree biomass would be useful as an effective way to estimate the extent of the amount of carbon sequester from rubber tree stands. Previous studies found that biomass estimates are also useful for quantifying net primary productivity, energy pathways, nutrient and carbon cycles and harvestable biomass yields (i.e., Ketterings et al. 2001; Saglan et al. 2008). For strategic carbon management, an auditing of national carbon stocks and fluxes, and measurement and monitoring of carbon stocks in reforestation programs require collection of convincing biomass data (IPCC 2007). Saglan et al. (2008) stated that a major challenge in measuring and monitoring carbon sequestration potential of plantation systems is measuring plant above ground biomass (AGB), which stores a significant portion of carbon assimilation. The most accurate way to measure and monitor AGB, and to estimate the state and change in carbon stocks for a plantation, is through periodic destructive sampling method (Saglan et al. 2008). However, Kale et al. (2004) and Delitti et al. (2006) claimed that harvesting and weighing a sufficient number of trees to represent the size and species distribution in an ecosystem is complex, time consuming, destructive, tedious, and labour intensive. Therefore, Kaonga and Bayliss-Smith (2010) have developed non-destructive techniques using regression models that relate biomass growth parameters by allometry. Hence, it provides more environmental friendly opportunity, rapid and yet less costly method as applied in this study.

Various allometric equations from global research have been developed for prediction of AGB or carbon stocks of trees in primary forests (e.g., Brown et al. 1995; Arau'jo et al. 1999; Chave et al. 2001; Basuki et al. 2009; Henry et al. 2010), secondary forests (e.g., Nelson et al. 1999; Ketterings et al. 2001) and shrub (Specht and West 2003; Scalan 1991; Burrow et al. 1999). However, there were only a few studies on allometric equation developed for rubber tree (Shorrocks et al. 1965; Chaundhuri et al. 1995; Dey et al. 1996; Rojo-MartÍnez et al. 2005; Yoosuk 2005; Kosei et al. 2014). Using only diameter and height as parameters in previous studies have raised many discussions on the usage of other plant physiology parameters such as chlorophyll content, stomatal conductance, photosynthesis and others to determine biomass and carbon stock of rubber trees. Thus in this research, tree allometric equations are obtained by establishing the relationship between eight tree parameters which include DBH, height, chlorophyll content index, stomatal conductance, photosynthesis rate, transpiration rate, leaf area index and age with carbon sequestration.

The work reported in this chapter aims at making a practical contribution to the development of methods for determining carbon sequestration potential of rubber plantations. The models for carbon sequestration estimation are needed for effective resource management planning not only aimed at maximizing the potential benefits of trees, but also important in their roles in measuring, reporting and verifying carbon stocks in rubber plantations as required for REDD+ mechanism of the country. Therefore, the objectives of this study include (i) to investigate the relationships between carbon sequestration and DBH, tree height, chlorophyll content index, stomatal conductance, photosynthesis rate, transpiration rate, leaf area index and age of rubber trees, and (ii) to report the development, evaluation and validation of predictive models for estimating carbon sequestration potential for rubber plantations in Malaysia.

19.2 Methodology

19.2.1 Study Site

The study site chosen was in the southwest of Selangor, Peninsular Malaysia, located from latitudes of 2° 35′ to 3° 55′ N and longitudes of 100° 45′ to 102° 00′ E (Fig. 19.1). The climate is wet tropical with a mean annual temperature of 26 °C and a mean annual rainfall of 3000 mm. The state of Selangor consists of forested and non-forested areas which are 250,650 ha and 542,160 ha, respectively. The forested area include inland forest (136,860 ha), peat swamp forest (82,890 ha), mangrove forest (18,899 ha) and forest plantation (11,381 ha) (Department of Forestry 2014). Agricultural tree crop plantations cover about 1.43 % of the total land base which comprise agricultural tree crop plantations such oil palm, rubber tree, coconut, coffee and cocoa (Department of Agriculture 2014). The topography in the sampled areas is moderately undulating and comprised mostly of flat lowland areas with altitude less than 100 m above sea level. The soils in Selangor are made up of Rengam, Serdang and Munchong series, which are suitable for agricultural tree crop cultivation (Department of Agriculture 1966).

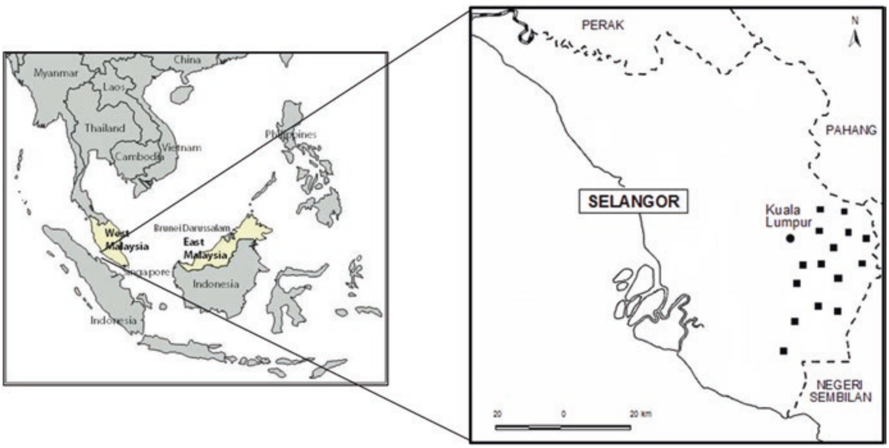

Fig. 19.1 Map of state of Selangor and the locations of study sites (■)

19.2.2 Field Measurements

The data were collected in 2011 and 2012. The rubber plantations chosen for field sampling are owned by estates and small landholders which are mostly managed by the Rubber Industry Smallholders Development Authority (RISDA), Ministry of Rural and Regional Development, Malaysia. The area and age of rubber plantations selected were ranged from 1 to 3 ha and from one to 28 years of age. For each selected rubber tree stand, circular plots were used to sample a total of 150 rubber trees. The size of the circular plot is about 0.03 ha with the distance between the rubber trees was not less than 2 m. The total number of circular plots per stand was ranged between 15 to 17 plots and each plot consisted of 12 trees.

In all plots, the DBH (1.3 m above ground height) of sampled trees was measured using a DBH tape. Tree heights of rubber trees were measured using a digital hypsometer (Model Opti-Logic InSight 400LH Laser Range Finder). Measurements of stomatal conductance, chlorophyll content index, photosynthesis rate and transpiration rate were made on ten randomly selected, matured and fully expanded leaves respectively. Ten leaves were selected from each rubber tree for all growth stages in the four variables. Positions of measurements were taken at the centre of the leaves surface (away from the main vein) to ensure adequate measurement of the surface area. The leaves were also chosen from the middle and upper parts of the rubber trees to synchronize the readings. Leaves with signs of pest damage or other damages were avoided.

Stomatal conductance, photosynthesis rate, and transpiration rate were measured on the abaxial surfaces of the leaves using Portable Photosynthesis System (Model TPS-2). This instrument is a completely self-contained unit for measuring the CO_2 assimilation (photosynthesis) and transpiration (water loss by evaporation). On the other hand, chlorophyll content index was measured on the abaxial surfaces of the leaves using a chlorophyll meter (Model Minolta SPAD 502 Plus). It measured absorption at 650 and 940 nm wavelengths to estimate chlorophyll levels. A Plant Canopy Analyzer (Model LAI-2200 PCA) (LI-COR Inc., Lincoln, Nebraska) was used to estimate the LAI of the rubber trees. Two repeats were made for each measurement, with one under open space (direct sunlight) and four below canopy readings. Measurements for all variables were made between the hours of 8:00 to 12:00 with temperature ranged between 25 and 35 °C each day to minimize diurnal influences and to standardize light levels for the measurements.

19.2.3 Data Analysis

19.2.3.1 Predictor and Response Variables and Descriptive Statistics

The selection of predictor variables (Xs) of plant physiology and rubber tree attributes were made through an extensive review of literature on the subject matter. The variables consisted of DBH, height (HT), chlorophyll content index (CC),

Table 19.1 The variables and abbreviations used in regression models

Symbol	Variables	Abbreviations
Y	Carbon sequestration	CS
X1	Diameter at breast height	DBH
X2	Height	HT
X3	Chlorophyll content index	CC
X4	Stomatal conductance	SC
X5	Photosynthesis rate	PN
X6	Transpiration rate	TRPT
X7	Leaf area index	LAI
X8	Age	AG

stomatal conductance (SC), photosynthesis rate (PN), transpiration rate (TRPT), LAI and tree age (AG). In this study, the response variable (Y) is carbon sequestration (CS). These variables are listed in Table 19.1. Descriptive statistics is used in the initial phase of a statistical analysis. The statistics such as the mean, standard deviation, minimum and maximum values are useful to identify the trends and possible relationships in the data to aid in determining the directions for subsequent analysis (Table 19.2).

19.2.3.2 Pearson's Correlation Coefficients and Scatter Plot Matrix

For an interval or ratio level scales, the most commonly used correlation coefficient is Pearson's r. A correlation matrix is generated by summarizing the relationships found to exist between CS data versus DBH, HT, CC, SC, PN, TRPT, LAI, and AG.

19.2.3.3 Model Building

The total of 2100 sampled trees were split randomly into two independent data sets. First, a set of 1402 trees (two-third) was used for building the model. Second, a set of 698 trees (one-third) were used for validating the models (Table 19.2). An ordinary least square (OLS) estimator was used to develop multiple linear regression models (Neter et al. 1996). Prior to model building, data exploration and preliminary model fitting were carried out to evaluate models that best fit the data. Various functions were initially explored and tested, but only those that meet regression assumptions (i.e., homogeneity of variance, linearity, normality, and non-autocorrelation) with high goodness of fit were retained. Subsequently, a series of data transformation were carried out to find out whether better models could be developed from this process. It was found that logarithmic transformation (ln) of response variables (CS) and the predictor variable (DBH and HT) [i.e., $\ln Y = \beta_0 + \beta_1 \ln X_1 + \beta_2 \ln X_2$] was the best transformation which produced model that fully meets regression assumptions with high 'goodness of fit'.

Table 19.2 Descriptive statistics of rubber tree parameters and carbon sequestration data sets used for model-building (n=1402) and validation data set (n=698), respectively

	Model-building data set					Validation data set				
Variable and unit	No. of rubber trees	Mean	SD[a]	Min.[b]	Max.[c]	No. of rubber trees	Mean	SD	Min.	Max.
Carbon Sequestration (tonnes/ha)	1402	5.82	1.31	1.80	7.50	698	5.26	1.32	1.90	7.50
Diameter at Breast Height (cm)	1402	2.89	0.58	1.20	3.80	698	2.87	0.59	1.30	3.80
Height (m)	1402	2.56	0.58	1.00	3.50	698	2.56	0.58	1.00	3.50
Chlorophyll Content Index (index)	1402	70.49	7.33	49.20	96.50	698	70.47	7.33	50.30	96.10
Stomatal Conductance (μmol m^{-2} s^{-1})	1402	750.29	1162.00	1.00	9135.00	698	709.06	1004.00	2.00	8453.00
Photosynthesis Rate (mmol m^{-2} s^{-1})	1402	10.90	14.77	−24.50	98.70	698	11.26	14.86	−51.00	94.30
Transpiration Rate (mmol m^{-2} s^{-1})	1402	2.18	1.39	0.02	9.26	698	2.14	1.33	0.02	9.50
Leaf Area Index (index)	1402	2.15	0.84	0.62	3.41	698	2.13	0.86	0.62	3.40
Age (year)	1402	12.20	8.17	1.00	28.00	698	12.02	8.15	1.00	28.00

Note: [a]Standard Deviation, [b]Minimum, [c]Maximum

Homogeneity of variance and linearity of data were evaluated from residual plots whereas autocorrelation and normality were checked with Durbin-Watson statistics and probability plot, respectively. While logarithmic transformation is reported to increase the statistical validity of regression analysis by homogenizing variance, it introduces a slight downward bias when data are back-transformed to arithmetic units (Baskerville 1972). Seven 'good' candidate models were selected based on an evaluation of the usefulness of predictor variables in prediction. A stepwise selection method was used in regression model using a PROC REG command in SAS version 9.3 (SAS Institute Inc. 2012).

19.2.3.4 Model Validation

Plots of predicted values against measured values, and the prediction error of response variables against predicted values in the validation data set were examined to find any areas of poorer prediction data. In addition, the following root mean squared error (RMSE) and estimated correlation index square (I^2) were calculated to evaluate the results from validation process:

$$\text{RMSE} = \sqrt{\frac{1}{n}\sum_{n}^{i=1}(yi - yi)^2}$$

$$I^2 = 1 - \frac{\sum_{i-1}^{n}(yi - yi)^2}{\sum_{i-1}^{n}(yi - \bar{y}i)^2}$$

Where n is the number of stands, y the measured response variable, $\bar{y}$ the mean of the measured response variable, and $\hat{y}$ is an estimated value of the response variable.

19.3 Results and Discussion

19.3.1 Relationships Between Variables

From the analysis, the highest positive r value was recorded for CS versus DBH followed by CS versus HT, AG, LAI and CC. However CS versus PN, SC and TRPT showed negative r values. The majority of predictor variables were significantly correlated with CS, ranging in r values from 0.99 to −0.26 ($p \leq 0.001$). However, there was no significant correlation between CS and PN ($r = -0.07$; $p \geq 0.05$). Results showed that CS was positively and significantly correlated with DBH, HT, AG and LAI (r = 0.99, 0.96, 0.86 and 0.85, respectively, $p \leq 0.001$) suggesting that as DBH, HT, LAI and AG increased, CS increased. In contrast, CS was inversely correlated to PN, SC and TRPT, indicating that when PN, SC and TRPT decreased

Table 19.3 Pearson's correlation matrix associated with the p values (n=1402)

Pearson's correlation coefficients (r) and p values									
	ln*CS*	ln*DBH*	ln*HT*	*CC*	*SC*	*PN*	*TRPT*	*LAI*	*AG*
ln*CS*	1.0								
	-								
ln*DBH*	0.99	1.0							
	<.0001								
ln*HT*	0.96	0.90	1.0						
	<.0001	<.0001							
CC	0.30	0.35	0.20	1.0					
	<.0001	<.0001	<.0001						
SC	−0.21	−0.20	−0.21	−0.04	1.0				
	<.0001	<.0001	<.0001	0.1668					
PN	−0.07	−0.06	−0.08	−0.01	0.05	1.0			
	0.0142	0.04	0.0037	0.8219	0.0538				
TRPT	−0.26	−0.25	−0.27	−0.01	0.48	0.12	1.0		
	<.0001	<.0001	<.0001	0.6409	<.0001	<.0001			
LAI	0.86	0.83	0.86	0.17	−0.25	−0.08	−0.31	1.0	
	<.0001	<.0001	<.0001	<.0001	<.0001	0.0024	<.0001		
AG	0.85	0.81	0.87	0.11	−0.25	−0.08	−0.312	0.95	1.0
	<.0001	<.0001	<.0001	<.0001	<.0001	0.0022	<.0001	<.0001	

($r = -0.07$, -0.21 and -0.26, respectively, $p \leq 0.05$), CS increased. The extent of correlations between variables is shown in Table 19.3.

The strong positive correlations between DBH and HT versus CS recorded in this study were in agreement with the work conducted by Mulugeta et al. (2009). In their study, positive relationships were observed between biomass of *Eucalyptus globulus* Labill with diameter and height. Another study by Antonio et al. (2007) found a significant increase in the predictive ability of biomass estimation models for *Eucalyptus* when height was included as an additional predictor variable for diameter. Previous study on allometric relationships of different tree species and stand AGB in Gomera Laurel Forest in Canary Island, Spain found a highly significant relationships ($p \leq 0.001$) between AGB and DBH and tree height (Jesus' et al. 2005).

In this study, there was a weak but significant positive correlation recorded for CS versus CC ($r=0.30$; $p \leq 0.001$)). This is comparable with a study carried out by Suharja and Sutarno (2009) who found positive correlations between chlorophylls a, b and c versus dry weight of two varieties of chilli plants. They stated that if the fresh weight of plants increased, the dry weight would increase as well, and so do the chlorophyll a, chlorophyll b and total chlorophyll leaves of both varieties of chili.

As mentioned previously, there was a negative correlation between SC and CS. This could be due to the whole-plant water flux estimations (Köstner et al. 1996; Hubbard et al. 1999; Ryan et al. 2000; Schäfer et al. 2000) which demonstrated that foliar stomatal conductance is consistently lower in larger and older trees than in smaller and younger trees in common environmental conditions. Lower stomatal conductance in taller trees has been explained by greater water limitations as a consequence of a larger restriction to water flow from soil to leaves because of increased path lengths in stems and branches of taller trees (Ryan and Yoder 1997; Bond and Ryan 2000; Mencuccini and Magnani 2000).

19.3.2 Carbon Sequestration Models

Regression models for carbon sequestration were highly significant (all $p \leq 0.001$). The initial equations developed are presented in Models (1) and (2) as follows:

$$\ln CS = c + a \ln DBH \tag{19.1}$$

$$\ln CS = c + a \ln HT \tag{19.2}$$

where CS is in tonnes/ha, DBH is in cm, HT is in m, c is an intercept, and a is the slope coefficient of the regression. The values of the multiple coefficients of determination (R^2) are presented in Table 19.3. Model 1 recorded R^2 value of 0.97 which used only DBH as a predictor variable, whereas Model 2 recorded R^2 value of 0.92 which used only HT as a predictor variable. However, tree biomass can be affected by a combination of DBH and HT. Hence, predictor variable in Model 2 (HT) was incorporated in Model 1 resulting in Model 3 with DBH and HT as the predictor variables. By including DBH and HT in Model 3, a multiple linear regression model is obtained:

$$\ln CS = c + a \ln DBH + b \ln HT \tag{19.3}$$

The good fit of Model 3 was the result from strong correlations between DBH versus CS and HT versus CS. Therefore the combination of DBH and HT in Model 3 has not only improved the R^2 value to 0.99, but also reduced the SE_E to 0.00850 tonnes/ha (Table 19.4).

From the earlier analysis, AG had also shown to establish a strong and significant relationship with CS ($r=0.85$, $p \leq 0.001$). While incorporating AG does not change R^2 value of the model, it has given a positive effect in terms of reducing the Mallow's

Table 19.4 Comparison of candidate models fitted to carbon sequestration estimates (n=1402)

No.	Model candidates	R^2	Adjusted R^2	Mallow's Cp	SE_E	No. predictor variables
1	$\ln CS = c + a \ln DBH$	0.97	0.97	15016.40	0.0285	1
2	$\ln CS = c + a \ln HT$	0.92	0.92	51205.50	0.0458	1
3	$\ln CS = c + a \ln DBH + b \ln HT$	0.99	0.99	69.01	0.0085	2
4	$\ln CS = c + a \ln DBH + b \ln HT + cAG$	0.99	0.99	0.87	0.0118	3
5	$\ln CS = c + a \ln DBH + b \ln HT + cLAI$	0.99	0.99	29.47	0.0099	3
6	$\ln CS = c + a \ln DBH + b \ln HT + cLAI + dAG$	0.99	0.99	1.97	0.0123	4
7	$\ln CS = c + a \ln DBH + b \ln HT + cTRPT + dAG$	0.99	0.99	2.59	0.0121	4

Cp value from 69.01 to 0.87. Therefore, AG was added in the equation (Model 4) as follows:

$$\ln CS = c + a \ln DBH + b \ln HT + cAG \tag{19.4}$$

From a statistical analysis point of view, AG and LAI has established equally strong and significant correlations with CS (i.e., r values of 0.86 and 0.85, respectively; $p \leq 0.001$). Therefore, it was interesting to explore the effects of eliminating AG and adding LAI in the model. As expected, this step has no effect on R^2 value, however it increased the Mallow's Cp and reduced SE_E values. This step had resulted in a model with three predictive variables as follows:

$$\ln CS = c + a \ln DBH + b \ln HT + cLAI \tag{19.5}$$

Of the eight predictor variables, DBH, HT, AG and LAI were identified as the four most strongly correlated variables with CS. Combining them in the equation (Model 6) did not affect R^2 and adjusted R^2 values. However, this step had resulted a decrease in Mallow's Cp value but a moderate increase in SE_E.

$$\ln CS = c + a \ln DBH + b \ln HT + cLAI + dAG \tag{19.6}$$

The final variable of this study was TRPT. While TRPT established a weak relationship with CS, adding this variable in the model had improved Mallow's Cp into a value that was more balanced with the number of predictor variables in the model. This suggested that despite being recognized as the weakest variable, TRPT had a role to play in predicting CS to some extent. Models 6 and 7 seemed to be predicting CS equally well. Therefore replacing LAI to TRPT had produced an equation with four predictor variables as shown below (Model 7);

$$\ln CS = c + a \ln DBH + b \ln HT + cTRPT + dAG \tag{19.7}$$

To choose the best prediction model for CS, evaluations were made between the above seven candidate models based on the usefulness of predictor variables in the models. Other factors considered were in terms of practicality, simplicity and predictive ability.

19.3.3 Model Validation

The estimated regression models were applied to validation data sets, reserved for testing. Table 19.4 shows results from model validation and Fig. 19.2 shows 1:1 relation between observed and predicted CS from validation datasets. The validation summary suggested that reliable estimates of rubber tree CS can be obtained using the above models. However, Models 1 and 2 appeared to have more scattered data than the rest of the models. Models 3 to 7 predicted equally well across CS. Figure 19.3 shows predicted carbon sequestration versus residual of the estimation (observed carbon sequestration-predicted carbon sequestration) for validation.

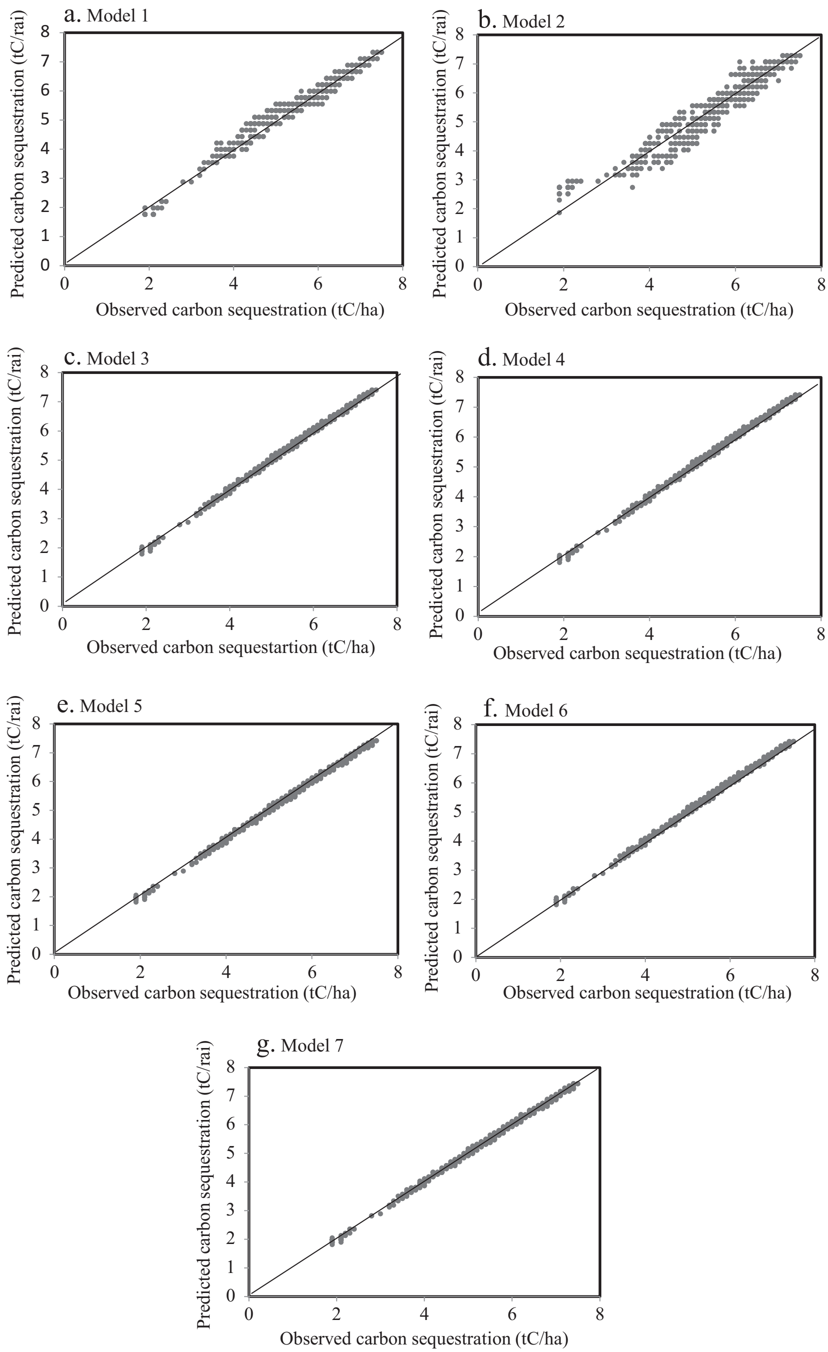

Fig. 19.2 Observed carbon sequestration versus predicted carbon sequestration for validation data sets (n=698)

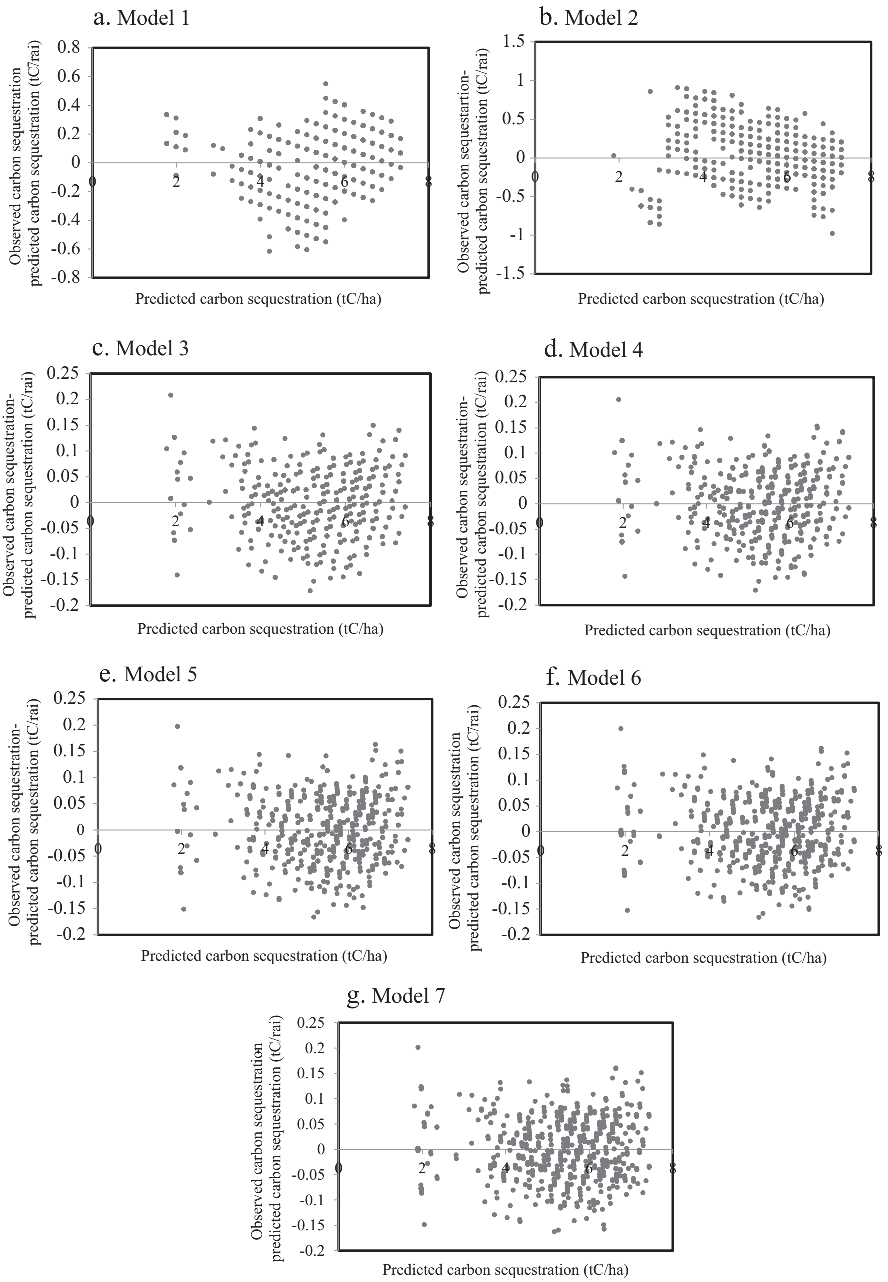

Fig. 19.3 Predicted carbon sequestration versus residual of the estimation (observed carbon sequestration-predicted carbon sequestration) for validation data sets (n=698)

Table 19.5 Summary of model validation results for estimating CS (n=698)

Model no.	p[a]	I^2	RMSE (tonnes/ha)
1	1	0.97	0.21
2	1	0.92	0.37
3	2	0.99	0.06
4	3	0.99	0.06
5	3	0.99	0.06
6	4	0.99	0.06
7	4	0.99	0.06

Note: [a]number of predictor variables in the model

The I^2 values for each model ranged between 0.92 and 0.99 as shown in Table 19.5. However, during the model building stage, Model 3 showed a stronger estimate of CS as compared to other models in terms of the lowest SE_E value recorded (i.e., 0.00850). Upon validation, this model maintained its highest predictive ability by achieving a high $I^2$2 value (0.99) and low RMSE (0.06 tonnes/ha). Therefore, Model 3 might be preferred over other models as it provides a good balance between practicality, predictive ability, and simplicity with a potential time-saving in computation.

A number of studies had found DBH and HT provided best correlations with biomass and carbon stocks. For example, Shorrocks et al. (1965) studied the relationships between a stem diameter of rubber tree at 1.50 m from soil and its dry AGB of biomass. They found a linear function of relationship between diameter (log D) and dry AGB (log B) as $\log AGB = 2.786 \log D - 2.5843$. Another study which determined the relationship between girth and AGB of rubber tree was conducted by Chaundhuri et al. (1995) in India using two rubber tree clones of RRIM 600 and RRII 118. They studied relationships using the biomass as a power function of girth for different age groups from the first year after planting up to the fifth year. A general equation was thus developed as $g(W) = 2.2784X^{2.200}$ (where X is the girth at 15 cm height from bud union).

In another study in Cambodia, Khun et al. (2008) developed a volume equation for standing rubber trees clone PR107 was $V=0.00018381D^{2.23961}$ or $V = 0.0002488D^{2.29535}$ where V is the over bark volume (m^3), D is DBH (cm) and H is total height (m). In Choba, Port Harcourt, Nigeria, Oyebade and Hbitimi (2011) evaluated a set of height-diameter models from twenty plots of rubber plantations. They used nonlinear techniques to develop the functional models with models coefficients derived from 198 samples of standing trees. The predictive models gave a good height-diameter relationship with R^2 values ranging from 0.62 to 0.98. The equation developed was $Y = \beta_0 + \beta_1 \ln D$ where y is the height (m) and D is the DBH (cm) of rubber trees.

Montagua et al. (2005) conducted study on the AGB of *Eucalyptus pilularis* from seven contrasting study sites in Australia. Differences were observed in (1) partitioning of biomass between the stem, branch wood and foliage; (2) stem wood density, and (3) relationship between DBH and height. For all predictor variables, examination of the model residuals of the site-specific and general relationship indicated that using DBH alone as the predictor variable produced the most stable general relationship. The general relationship determined was $\ln AGB = \ln\beta_0 + \beta_1 \ln DBH$, where AGB was in kg and DBH in cm.

A study was conducted in Sumatra, Indonesia by Onrizal et al. (2009) to develop allometric functions for AGB of carbon stock from age one to nine year stands of *Eucalyptus grandis*. The results showed that the allometric equations for both biomass and carbon stock were in good relation with stem diameter (D) as log-linear equation. The best allometric equations for ABG and carbon stock of *Eucalyptus grandis* were $AGB = 0.0678D^{2.5794}$ ($R^2 = 0.99$) and $CS=0.0266D^{2.6470}$ ($R^2 = 0.98$), respectively.

A study using a non-destructive sampling method, such as this study, was conducted in Sumatra, Indonesia. In this study, the methods of choosing values for a and b (constant parameters) did not require destructive measurement as the parameter b can be estimated from the site-specific relationship between height (H) and diameter (D), $H=kD^c$ as $b = 2 + c$ while the parameter a can be estimated from the average wood density (ρ) at the site as $a = r\rho$, where r is expected to be relatively stable across sites. The allometric equation of biomass proposed was therefore $B = r\rho r D^{2+c}$ (Quirine et al. 2001).

19.4 Conclusion

From the analysis, the highest positive correlations were recorded for CS versus DBH followed by CS versus HT, AG, LAI, CC. However, CS versus PN, SC and TRPT were inversely correlated with CS. Accurate predictive models were obtained for predicting carbon sequestration potential in rubber trees. Model 3 (with two predictive variables) was the best performing model, $\ln Y = -1.108 + 1.494 \ln DBH + 0.813 \ln HT$, ($R^2 = 0.99$). The good fit of this model gives an indication of the potential rubber tree attributes especially DBH and HT for estimating carbon sequestration potential in rubber tree plantations.

The predictive models developed provide a convenient and useful tool for resource planners in making forecasts. This may not only assist in the development of policy recommendations to address the climate change agenda, but also to maximise the potential benefits of rubber tree resources in terms of the ecosystem functions. In addition, findings from this work may also provide an important basis for future research to study forest plantations or other types of tree crop components in Malaysia or other countries with similar resource management issues and constraints.

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Chapter 20
Carbon Sequestration in Mediterranean Oak Forests

Isabel Cañellas, Mariola Sánchez-González, Stella M. Bogino, Patricia Adame, Daniel Moreno-Fernández, Celia Herrero, Sonia Roig, Margarida Tomé, Joana A. Paulo, and Felipe Bravo

20.1 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

I. Cañellas (✉) • M. Sánchez-González • P. Adame • D. Moreno-Fernández
Joint Research Unit INIA-UVa, Department of Forest Systems and Resources, CIFOR-INIA, Madrid, Spain
e-mail: canellas@inia.es; msanchez@inia.es; adame.patricia@inia.es; danielmoreno270@hotmail.com

S.M. Bogino
Department of Agricultural Sciences, State University of San Luis, San Luis, Argentina
e-mail: stellabogino@gmail.com

C. Herrero
Joint Research Unit INIA-UVa, Department of Forest Resources, Universidad de Valladolid, Palencia, Spain
e-mail: chdeaza@pvs.uva.es

S. Roig
Department Sistemas y Recursos Naturales, ETS de Ingeniería de Montes, Forestal y Medio Natural, UPM, Madrid, Spain
e-mail: sonia.roig@upm.es

M. Tomé • J.A. Paulo
Instituto Superior de Agronomia, Centro de Estudos Florestais, University of Lisbon, Lisbon, Portugal
e-mail: magatome@isa.ulisboa.pt; joanaap@isa.ulisboa.pt

F. Bravo
ETS de Ingenierías Agrarias - Universidad de Valladolid & iuFOR - Sustainable Forest Management Research Institute, Universidad de Valladolid - INIA, Palencia, Spain
e-mail: fbravo@pvs.uva.es

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"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).

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 Iberian oak forests as well as increase our understanding of their dynamics.

The two oak species on which this study focuses were chosen because of their ecological and economic importance. At national level, cork oak forests occupy more than 500,000 ha in Spain and over 700,000 ha in Portugal 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 in Spain and mixed stands are very frequent. Woodlands dominated by *rebollo* oak were very important for firewood production until about four 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 *Pinus sylvestris* L. and *Q. pyrenaica*, which occupy more than 160,000 ha in Spain (ICONA 1998).

20.2 Biomass Production and Carbon Sequestration in Spanish Cork Oak Forests

The objective of the present section is to estimate the amount of carbon (C) sequestered in Spanish cork oak forests and analyse its evolution over 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/ha).
- 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 rotation is about 150 years, which is considered the upper limit for quality cork production. 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.
- The cork quality of all cork oaks in the stand was medium, which, according to Sánchez et al. (2008), corresponds to a cork thickness of 29.25 mm at the ninth year after debarking.
- 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 and 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.

We are aware of the limitations of an estimation made under the aforementioned hypothesis but nevertheless 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.

20.2.1 Material and Methods

Owing to the fact that the carbon 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:

Table 20.1 Biomass equations for cork oak by Ruiz-Peinado et al. (2012b)

Tree fraction	Equation
Stem and thick branches	$B = 00.00525 \cdot d^2 \cdot h + 0.278 \cdot d \cdot h$
Thick branches	$B = 0.0135 \cdot d^2 \cdot h$
Medium branches	$B = 0.127 \cdot d \cdot h$
Thin branches and leaves	$B = 0.0463 \cdot d \cdot h$
Roots	$B = 0.0829 \cdot d^2$

B is the biomass weight of each tree fraction (kg), d is the diameter at breast height (cm) and h is the tree height (m)

20.2.1.1 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 Ruiz-Peinado et al. (2012b) for cork oak (Table 20.1).

Since the equations developed by Ruiz-Peinado et al. (2012b) consider diameter over cork and the authors do not provide any information with regard to the cork (thickness of cork on trees used to fit the equations or cork rotation year in which they were collected), it is impossible to know how much of the stem and branch biomass is wood and how much is cork. To overcome this shortcoming, we applied the equations from Table 20.1 using diameter under cork. Furthermore, to avoid the overestimation of cork biomass we applied a correction factor that reduced the estimated cork biomass by 5 %.

The diameter at breast height under cork was estimated by applying the diameter increment model for cork oak forests (Sánchez-González et al. 2006):

$$idu = \frac{idu_{po}}{1 + e^{-(0.73 + 94.97\frac{1}{N})}} \tag{20.1}$$

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):

$$du_2 = (83.20 + 5.28SI - 1.53h_1 / du_1)^{1-\frac{\ln(1-e^{-0.0063t_2})}{\ln(1-e^{-0.0063t_1})}} du_1^{\frac{\ln(1-e^{-0.0063t_2})}{\ln(1-e^{-0.0063t_1})}} \tag{20.2}$$

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).

Tree height was estimated using the following height-diameter equation based on the one developed by Sánchez-González et al. (2007):

$$h = 1.3 + (h_{po} - 1.3) \cdot (\frac{du}{du_{po}})^{0.4898} \tag{20.3}$$

where: *du* is the diameter at breast height under cork (cm); h_{po} is the potential height estimated using the model developed by Sánchez-González et al. (2005), du_{po} is the potential diameter under cork estimated using Eq. 20.3.

Having calculated dry biomass through the equations given in Table 20.1, 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.

20.2.1.2 Dry Cork Biomass Estimation

Dry cork biomass or cork weight was estimated using the following expression:

$$w = cb.sh.cu.cork \quad density \tag{20.4}$$

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 250 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):

$$cb_2 = cb_1 (\frac{1 - e^{-0.04 t_2}}{1 - e^{-0.04 t_1}})^{\frac{0.57 + 1.86}{X_0}} \tag{20.5}$$

where: $X_0 = \frac{1}{2}((\ln(cb_1) - 0.57 \ln(1 - e^{-0.04 t_1})) \pm \sqrt{(\ln(cb_1) - 0.57 \ln(1 - e^{-0.04 t_1}))^2 - 4 \cdot 1.86 \ln(1 - e^{-0.04 t_1})})$

cb_i is cork thickness (cm) at cork age t_i (years).

Only data from complete growth years were considered in the cork growth model (3745 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 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.

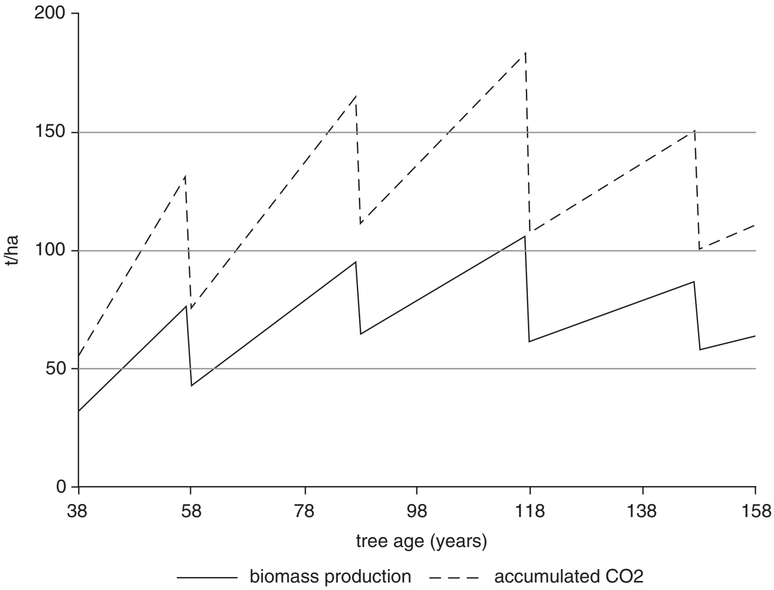

Fig. 20.1 Biomass production and accumulated CO_2 values for wood in cork oak forests

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 CO2 molecule and the carbon atom, 3.67.

20.2.2 Results and Conclusion

The wood biomass production in cork oak forests shows an increasing trend (Fig. 20.1 and Table 20.2). The cumulative wood biomass extracted over the whole period totals 220.77 Mg ha^{-1}. 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. 20.2 and Table 20.2), the maximum value is reached at 88 years, coinciding with the sixth cork harvest.

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. 20.1) and in cork (Fig. 20.2). In the case of cork, the accumulated CO_2 is extracted at each cork harvest although it remains sequestered in cork products. The total amount of accumulated CO_2, taking into account all the cork extractions, is 105.93 Mg·ha^{-1}.

Table 20.2 Wood and cork biomass production in cork oak forests

Age (years)	du	Tres ha^{-1}	*WB*	*CB*	*VCB*
38	14.61	500	31,78	0,00	1,49
48	18.93	500	53,10	2,55	1,28
58	24.39	250	43,13	5,23	1,03
68	28.77	250	59,48	4,88	0,00
78	33.01	250	77,60	5,62	0,00
88	37.11	150	64,62	6,32	0,00
98	43.45	150	78,40	4,45	0,00
108	47.45	150	92,72	4,86	0,00
118	51.26	70	61,72	5,26	0,00
128	61.33	70	70,39	2,94	0,00
138	65.20	70	79,14	3,13	0,00
148	68.88	40	57,82	3,31	0,00
158	77.86	40	63,58	2,14	0,00

du diameter at breast height under cork (cm), *cu* circumference at breast height over cork (cm), *WB* wood biomass (t D.M. ha^{-1}), *CB* cork biomass (t·ha^{-1}), *VCB* virgin cork biomass (t·ha^{-1})

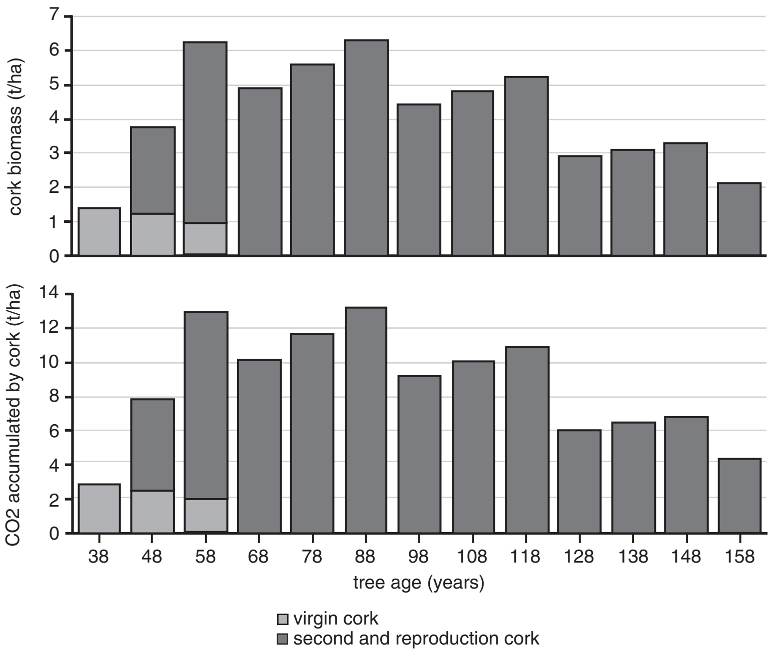

Fig. 20.2 Cork biomass production and accumulated CO_2 values at each debarking

The amount of carbon stored in a cork oak forest with the characteristics described in the introduction section would be 423.94 Mg·ha^{-1} 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. It would also be necessary to develop biomass equations for *Q. suber* that distinguish between the biomass produced by wood and by cork.

20.3 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 (AFN 2010) the species covers an area of 716,103 ha (pure stands and mixed stands). The great majority of the present stands are adult stands characterised by low density; the average stand density in the country is 65 trees/ha and 85 % of the stands have less than 120 trees/ha. The establishment of new plantations began towards the end of the 1980's and today they occupy a large of the country. According to official statistics (Torres 2009; AIFF 2013), rates of planting were 4800 ha/year, 9300 ha/year and 3767 ha/year for the periods 1990–1994, 1995–2003 and 2004–2009, respectively. The objective of the present work is to estimate the carbon sequestered by a stand of average site index and then by 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.

20.3.1 Material and Methods

The estimates of the evolution of carbon stocks in cork oak stands were based on version 5 of the SUBER model (Paulo et al. 2012; Paulo 2011; Tomé 2004), which includes the system of equations for tree biomass and crown width estimation developed for the Portuguese NFI 2005/2006 (Tomé et al. 2007) along with the root biomass equation developed for Spain by Montero et al. (2005). In comparison to the previous version of the SUBER model, which was used for the study presented in Cañellas et al. (2008), the present model includes improved versions of some of the equations, as well as the possibility to simulate non-age-related mortality which, according to data from the Portuguese NFI, is known to occur and to greatly influence the development of cork oak stands (AIFF 2013). The SUBER model is one of the models implemented in the sIMfLOR platform for Portuguese forest simulators (Faias et al. 2012). It allows the development of an existing or new stand to be

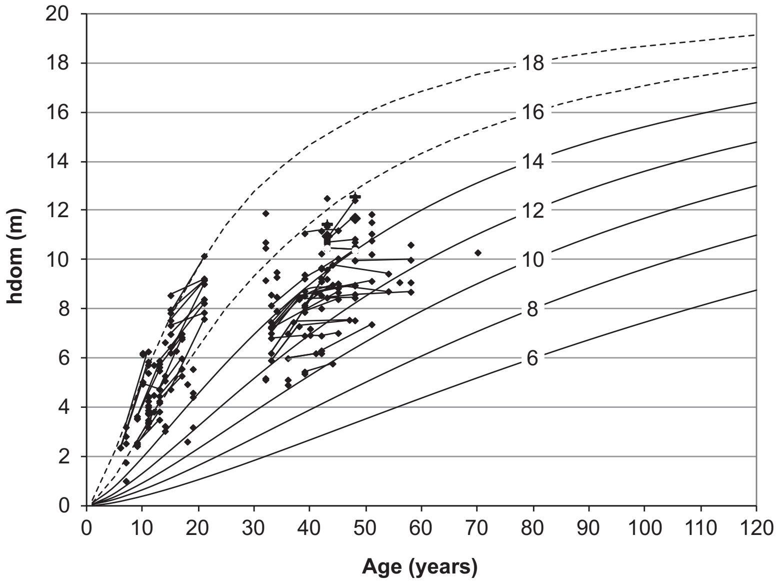

Fig. 20.3 Spanish site index curves for cork oak (Sánchez-González et al. 2005) and the permanent plots established in Portuguese stands for which the age is known. Observations from the same plots are connected with straight lines

simulated under different forest management approaches defined by the user and which are detailed below.

20.3.1.1 Developing the Yield Table

Figure 20.3 represents the site index curves developed by Sánchez-González et al. (2005) for Spain and which have been successfully tested for Portugal (Tomé 2004) together with information from the 42 permanent plots included in the SUBERDATA data base for cork oak growth in Portugal (Coelho and Godinho 2002) and for which the age is known. The figure clearly indicates that the new plantations (younger than 20 years) were established at better sites that the older ones: average site index for young stands is 16.6 m and for older stands 13.6 m. For the purposes of this study, new plantations were assumed to have an average 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 done using measured data (diameter at breast height and total height) from a 12 year old planted stand. The stand was planted with 500 trees/ha and at the time of the measurement there were 400 trees/ha (mortality prior to measurement was 20 %).

- 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 cannot be extracted before the perimeter at breast height reaches 70 cm).
- The debarking coefficient values (ratio between debarking height and tree perimeter at breast height) were defined after a close analysis of the debarking heights recorded in the forest inventory data (AFN 2010): 2 for the first and second cork extractions, and 2.5 for the third and consecutive extractions. These values are below the maximum limit allowed by law (2, 2.5 and 3), but reflect the practice of landowners in order to reduce the stress associated with cork debarking. For the same reason, increasing the debarking height is only applied when stand age is less than 100 years.
- Stand density was defined by thinning each time the crown cover percentage at the time of debarking was higher than 50 %. This crown cover percentage 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 using the equations available for reproduction cork (Paulo and Tomé 2010).
- The rotation considered was 150 years, which is considered the upper limit for quality cork production.

20.3.1.2 Simulation of Future Total Carbon Stocks

The simulation of the future carbon stocks was based on the yearly estimation of the areas of new plantations by age class (age classes of 1 year were used). According to official statistics (Torres 2009; AIFF 2013) rates of planting were 4800 ha/year, 9300 ha/year and 3767 ha/year for the periods 1990–1994, 1995–2003 and 2004–2009, respectively. This last rate (for 2009) was assumed for the years after 2010. This information was used to estimate the area of new plantations per year 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 of less than 6 years to take into account the non-success of some plantations. Total carbon stock at year t was estimated as:

$$Cstock_t = \sum_{150}^{j=1}[0.472(Ww_j + Wr_j) + 0.57Wb_j + 0.5Wc_j]A_{j,t}, \tag{20.6}$$

where *Wwj, Wrj, Wbj and Wcj* 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*.

20.3.1.3 Estimation of Carbon Sequestration during the Period 2008–2012

Carbon sequestered during the 1st commitment period of the Kyoto protocol (2008–2012) was estimated using the stock change approach:

$$\Delta C = Cstock_{2013} - Cstock_{2008}, \tag{20.7}$$

where $Cstock_t$ is the carbon stock at the start of year t.

20.3.2 Results and Discussion

The evolution of a cork oak plantation with the characteristics described in 3.1.1 is presented in Fig. 20.4. It is important to point out that the traditional cork oak stands are not frequently managed with such a high percent of crown cover as they are managed as an agro-silvopastoral system, which requires a low percentage of crown cover (compatible with grazing underneath) and which therefore limits to some extent the stand regeneration. However, most of the new plantations are being managed with cork production as the main objective. This option can only be achieved with a complete use of the space by the trees, therefore a high percentage of crown cover was used in the development of the yield table.

Mature cork production increases up to a stand age of 80 years (maximum value of 3675 kg ha^{-1} for 105 trees ha^{-1}) due to tree growth and increasing of debarking height. Values up to 3000 kg ha^{-1} are maintained up to an age of 145 years, but tend to decrease at older ages. The differences between these values and those presented by Cañellas et al. (2008) are due to the use of lower debarking coefficient values, which better reflect the practice of landowners, as well as lower values for percentage crown cover, closer to reality than those used in the previous study.

Figure 20.5 shows the estimations of total area and the corresponding carbon stock for the first 50 years (until 2040) of young plantations established after 1990. According to the assumptions used for the development of the yield table (see 3.1.1), new plantations of cork oak may sequester as much as 3.0 Mt. over a period of 50 years if the plantation rates of 3767 ha year^{-1} achieved in 2003 are maintained. The estimates of carbon sequestration presented by Cañellas et al. (2008) for the same period (5.9 Mt) assumed annual plantation rates of 9300 ha year^{-1}. The importance of plantation rates has also been demonstrated by Coelho et al. (2012). According to this study, the amount of C sequestered during the 1st commitment period of the Kyoto protocol was 344 10^3 Mg of C sequestered, which means 1.3 10^6 Mg 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.

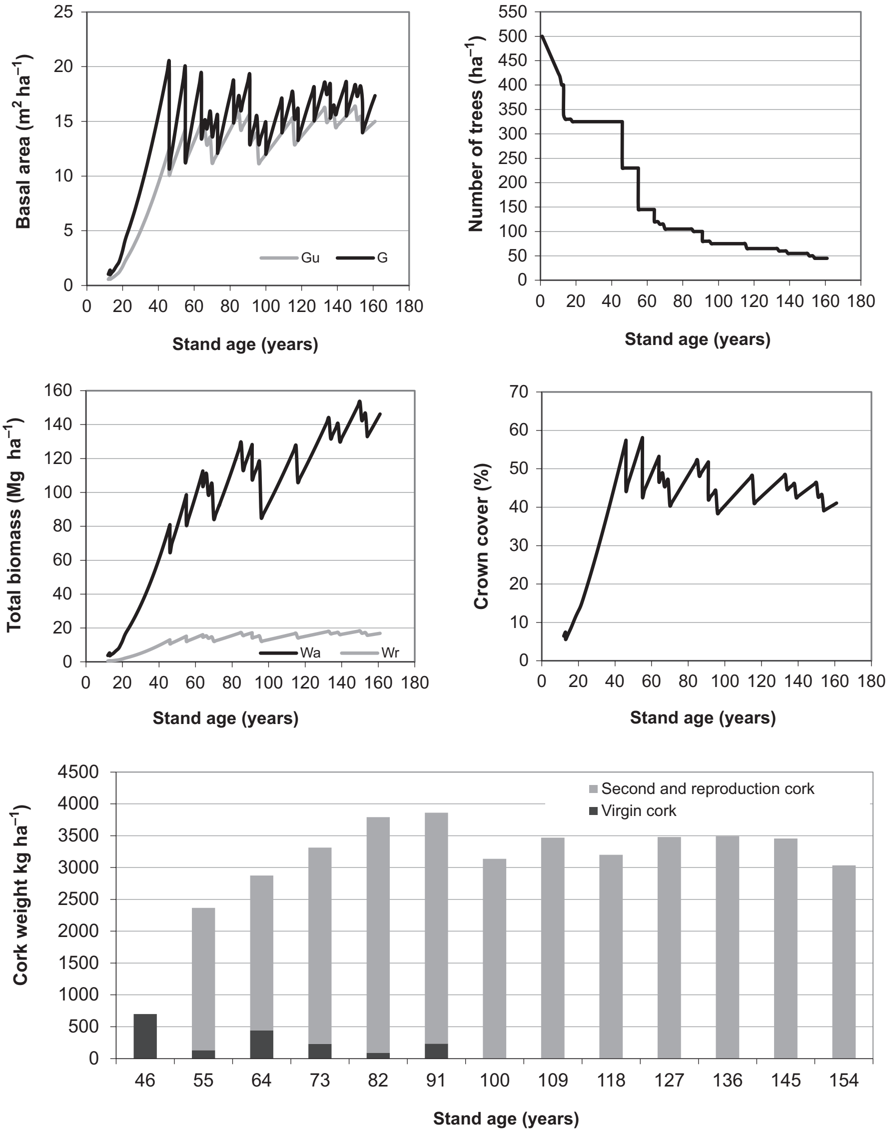

Fig. 20.4 Evolution of a new cork oak plantation with the characteristics presented in 3.1.1. *G* and *Gu* are, respectively, over and under bark basal area and *Wa* and *Wr* are total aboveground and root biomass

This result is very important for Portugal as the European Commission recently (October 2007) reduced Portugal's estimations for reductions induced by afforestation and reforestation due to the fact that the Portuguese estimates had not been adequately justified. The results presented here support Portugal's initial predictions, indicating that far from being an over-estimation, the value presented may be an under-estimation.

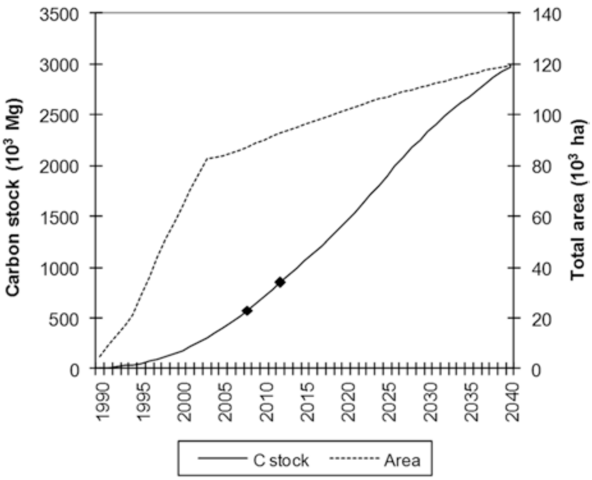

Fig. 20.5 Estimation of the evolution of the area of new cork oak plantations and the corresponding fixation of carbon for the period 1990–2040. The black lozenges indicate the years 2008 and 2013

20.4 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 four 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.

20.4.1 Data and Methods

The data for model construction were obtained from 200 plots belonging to the Spanish National Forest Inventory (SNFI) (DGCN 1996) in the northwest area of Spain, considering the biogeoclimatic conditions defined by Elena Rosselló (1997). 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 is based on equations involving site index (*SI*), stems per hectare (*N*), volume (*V*), and mean height (*Hm*):

$$1st\,\text{equation : site index defined}\,by\,\text{Adame}\,et\,al.\text{(2006)} \tag{20.8}$$

$$2nd\,\text{equation} : N = e^{(8.49459 - 0.114019 \cdot Ho)} \tag{20.9}$$

$$R^2 = 0.37$$

$$3rd\,\text{equation} : Dg = 2.17047 + \frac{2174.69}{N^{0.7415}} + 0.215575 \cdot Ho \tag{20.10}$$

$$R^2 = 0.95$$

$$4th\,\text{equation} : \ln(V) = -0.709618 + 0.936607 \cdot \ln(G \cdot Ho) \tag{20.11}$$

$$R^2 = 0.98$$

$$5th\,\text{equation} : Hm = 1.20895 + 0.643414 \cdot Ho \tag{20.12}$$

$$R^2 = 0.64$$

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.

Table 20.3 Thinning regimes applied in rebollo coppices

Intensive silvicultural treatments	Moderate silvicultural treatments
≈20 years: if initial density higher 2000–2500 trees/ha pre-commercial thinning has to be applied	≈15–20 years: if initial density higher 3500–4000 trees/ha 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	70 years: 3rd low thinning
80 years: 4th low thinning	90 years: 4th low thinning
Rotation: 100 years	Rotation: 120 years

Intensive silvicutural treatments (Site index 16 and 13 Adame et al. 2006) and moderate silvicultural treatments (Site index 10 and 7) are presented in Table 20.3. The yield table according to site quality and silvicultural treatments is shown in Table 20.4.

Dry biomass of the whole tree was estimated as a sum of the different fractions (stem plus branches with a diameter over 7 cm, branches with a diameter between 2 and 7 cm, branches with a diameter of less than 2 cm with leaves, and the below-ground fraction, the roots) using the equations developed by Ruiz-Peinado et al. (2012) for *Q. pyrenaica* (Table 20.5).

The calculation of the amount of carbon stored in the biomass of trees is usually based on biomass expansion factors (Nabuurs et al. 2000). Once dry biomass has been estimated, the percentage of carbon in each fraction and in the whole tree 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.

20.4.2 Results and Discussion

The results 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 *Q. pyrenaica* in northwest Spain are given in Tables 20.6 and 20.7 shows the total carbon fixed (t), taking into account site index and age.

where Cs+b7, Cb7–2, Cb2, Cr and Ct are the carbon stored (t ha^{-1}) in the stem and branches larger than 7 cm, branches with a diameter of between 2 and 7 cm, branches with a diameter of less than 2 cm with leaves, and the roots; respectively. The results for carbon accumulation in *rebollo* oak (Tables 20.6 and 20.7) 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 60–70 years for site index 16 is 1.98 Mg/ha year, 1.66 Mg/ha year for site index 13, 1.22 Mg/ha year for site index 10 and 1.05 Mg/ha-year for site index 7.

Table 20.4 Yield tables according to site quality and silvicultural treatments

Site index 16: intensive silvicultural treatments												
		Main crop							Main crop			
		Before thinning				Crop removed			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	2000	12.2	23.5	87.5	650	10.4	19.4	1350	13.0	18.0	68.1
40	13,0	1350	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 ndex 13: ntensive silvicultural treatments												
		Main crop							Main crop			
		Before thinning				Crop removed			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	2500	10.4	21.4	60.0	700	8.7	10.9	1800	11.1	17.3	49.1
40	10.0	1800	12.7	22.9	79.8	600	10.5	17.3	1200	13.7	17.6	62.5
60	13.0	1200	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	3400	8.6	19.5	38.2	1000	6.4	6.8	2400	9.3	16.3	32.3
50	8.7	2400	10.8	22.1	68.2	750	8.1	11.3	1650	11.9	18.2	56.9
70	10.9	1650	13.5	23.6	89.5	600	10.1	17.3	1050	15.1	18.7	72.3
90	12.4	1050	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	4000	7.49	17.6	21.1	1200	5.62	3.36	2800	8.2	14.6	17.8
50	5.9	2800	9.49	19.8	42.5	900	7.12	7.25	1900	10.4	16.2	35.3
70	7.9	1900	11.9	21.2	59.6	650	8.95	10.8	1250	13.2	17.1	48.8
90	9.2	1250	15.1	22.5	72.8	400	11.4	12.3	850	16.6	18.5	60.4
120	10.4	850	19.0	24.2	87.1							

Table 20.5 Biomass equations for the rebollo oak by Ruiz-Peinado et al. (2012b)

Tree fraction	Equation
Stem and thick branches	$B = 0.0261 \cdot d^2 \cdot h$
Medium branches	$B = -0.0260 \cdot d^2 + 0.536 \cdot h + 0.00538 \cdot d^2 \cdot h$
Thin branches and leaves	$B = 0.898 \cdot d - 0.445 \cdot h$
Roots	$B = 0.143 \cdot d^2$

B is the biomass weight of each tree fraction (kg), *d* is the diameter at breast height (cm) and *h* is the tree height (m)

Total fixed carbon at the end of the rotation on site index 16 represents 13.8 % more than on site index 13, 40.4 % more than on site index 10 and 68.5 % more than on site index 7. The importance of forest management can be seen in the final differences in total carbon sequestration (Fig. 20.6). Furthermore, initial differences at 30 years between intensive and moderate silvicultural treatments ranged between 79.3 % (between site index 16 and site index 7) and 12.1 % (between site index 13 and 10).

20.5 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* Willd. and *Pinus sylvestris* L. located in the Central Mountain Range, Spain. At national level, pure *Pinus sylvestris* stands occupy 678,685.53 hectares, mixed stands of *P. sylvestris* and *Q. pyrenaica* 168,739.48 hectares and pure *Quercus pyrenaica* stands occupy 585,397 hectares (DGCN 1996). 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.

20.5.1 Materials and Methods

Permanent plot data from Valsaín forest (Central Spain) were used to estimate carbon sequestration by oak stands (Table 20.8). Three plots (50x50 m^2) in each of the forest types (mixed, pure scots pine and pure oak woodland) were established. The total, above and below-ground dry biomass by fraction (stem, roots, branches and needles) were estimated using the Ruiz-Peinado et al. (2012a, b) models (Table 20.1). Carbon content was estimated as 0.475 of biomass (Ibáñez et al. 2002; IPCC 2007). The amount of carbon dioxide was estimated by multiplying the carbon amount by 3.67 (ratio between CO2 molecular weight and C atomic weight).

Table 20.6 Carbon concentrations (**t ha^{-1}** ha) in different components of rebollo oak for high quality sites

Site index 16: intensive silvicultural treatments															
Age	Carbon before thinning					Carbon removed					Carbon after thinning				
(years)	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t
30	30.95	6.97	6.86	20.22	65.01	6.43	1.68	1.87	4.78	14.75	24.84	5.32	4.98	15.50	50.64
40	38.90	7.43	6.07	21.75	74.15	9.20	1.97	1.84	5.74	18.76	29.70	5.44	4.18	15.91	55.23
60	48.38	8.12	5.30	23.56	85.37	11.47	2.06	1.54	6.03	21.10	37.21	6.09	3.74	17.59	64.64
80	57.94	9.01	4.71	25.64	97.30	10.83	1.76	1.06	5.07	18.71	47.10	7.25	3.63	20.55	78.52
100	65.40	9.81	4.32	27.48	107.01										
Site index 13: intensive silvicultural treatments															
Age	Carbon before thinning					Carbon removed					Carbon after thinning				
(years)	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t
30	24.73	6.46	7.19	18.37	56.75	4.11	1.31	1.67	3.60	10.69	21.42	5.25	5.56	15.06	47.29
40	31.09	6.78	6.46	19.72	64.06	6.10	1.58	1.74	4.49	13.91	25.42	5.24	4.70	15.30	50.66
60	40.02	7.40	5.78	21.66	74.85	8.15	1.70	1.54	4.95	16.34	31.94	5.70	4.19	16.64	58.48
80	48.32	8.04	5.14	23.28	84.78	11.30	2.04	1.53	5.96	20.83	36.88	5.97	3.56	17.19	63.60
100	57.89	8.93	4.50	25.31	96.63										

Table 20.7 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				
	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t
30	19.30	6.23	8.02	17.08	50.63	2.28	1.11	1.78	2.78	7.95	17.18	5.05	6.13	14.10	42.46
50	26.44	6.65	7.19	19.01	59.29	3.55	1.24	1.67	3.34	9.80	23.84	5.49	5.50	15.87	50.70
70	33.61	7.00	6.36	20.43	67.39	5.46	1.47	1.67	4.16	12.76	28.75	5.56	4.61	16.26	55.18
90	40.73	7.31	5.43	21.35	74.81	6.44	1.38	1.29	4.02	13.13	35.19	6.03	4.11	17.53	62.85
120	48.94	7.98	4.84	23.01	84.78										
Site index 7: moderate silvicultural treatments															
Age (years)	Carbon before thinning					Carbon removed					Carbon after thinning				
	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t	C_{s+b7}	C_{b7-2}	C_{b2}	C_r	C_t
30	8.65	2.16	10.18	15.28	36.27	1.78	1.07	1.90	2.56	7.31	6.87	1.09	8.28	12.73	28.96
50	18.44	4.89	7.87	17.13	48.31	2.85	1.19	1.76	3.08	8.88	18.80	4.91	5.47	13.96	43.13
70	26.35	5.93	6.47	18.28	57.03	4.09	1.27	1.59	3.50	10.44	23.97	5.08	4.69	14.79	48.53
90	32.51	6.11	5.62	19.36	53.60	5.00	1.21	1.26	3.47	10.94	29.70	5.44	4.18	15.91	55.23
120	39.95	6.62	5.02	20.84	72.04										

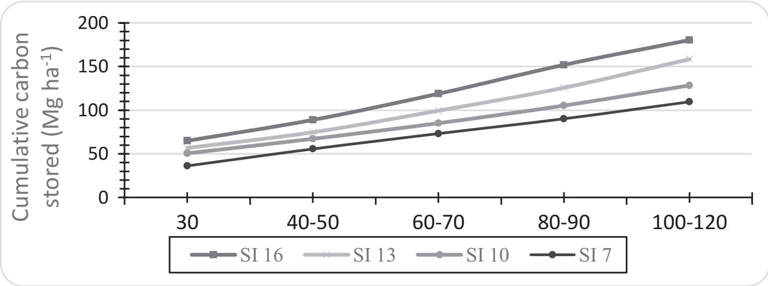

Fig. 20.6 Carbon sequestration (carbon before thinning plus the carbon extracted in previous thinnings) related to age and site index in rebollo oak coppice stands in northwest Spain. *SI* Site Index

Table 20.8 Main characteristics of sampled plots

	Mean	SD	Min	Max
Mixed stands (n=3)				
N	846.67	367.75	460	1192
BA	33.22	12.02	23.94	46.79
QMD	23.06	4.92	18.53	28.29
Mean DBH	19.74	3.67	16.53	23.74
CO_2 Total	376.67	132.38	257.30	519.05
CO_2 above	279.22	98.79	186.21	382.92
CO_2 below	97.45	34.23	71.09	136.13
Pine stands (n=3)				
N	276	114.33	184	404
BA	24.03	10.99	16.44	36.63
QMD	34.1	9.81	24.48	44.08
Mean DBH	42.07	2.36	39.75	44.46
CO_2 Total	472.09	125.61	377.31	614.56
CO_2 above	357.79	92.584	289.36	463.14
CO_2 below	114.30	33.09	87.952	151.42
Oak stands (n=3)				
N	622.67	200.57	480	852
BA	13.86	1.93	12.57	16.08
QMD	17.1	1.52	15.5	18.53
Mean DBH	15.87	0.68	15.09	16.33
CO_2 Total	152.82	23.66	133.75	179.30
CO_2 above	108.77	17.22	93.89	127.63
CO_2 below	44.05	6.61	39.86	51.67

Abbreviations: *N* number of trees (tress ha^{-1}), *BA* Basal area (m^2 ha^{-1}), *QMD* Quadratic mean diameter (cm), *Mean DBH* Mean diameter at breast height (cm), *CO_2 total* total amount of CO_2 (t ha^{-1}), *CO_2 above* total amount of carbon dioxide aboveground (t ha^{-1}), *CO_2 below* total amount of carbon dioxide belowground (t ha^{-1})

Table 20.9 Analysis of variance related to total, aboveground and belowground amount of fixed carbon dioxide

Carbon Dioxide		CO_2 Total		CO_2 Aboveground		CO_2 Belowground	
Source	**DF**	**Sum of Sq.**	**Pr > F**	**Sum of Sq.**	**Pr > F**	**Sum of Sq.**	**Pr > F**
Model	2	161147.46	0.0197	97235.41	0.0213	8071.13	0.0482
Error	6	67725.44		37255.16		4618.71	
Correct total	8	228872.91		134490.57		12689.84	
Lsmeans							
Pure stands of pine		472.09	a	357.79	a	114.30	a
Pure stands of oak		152.82	b	108.77	b	44.05	b
Mixed stands		376.67	ab	279.22	ab	97.45	ab

Note: Different letters denote significant differences in the total, above and below ground CO2 stock between dominant species at the 95 % level (Tukey's test)

A two-way Analysis of Variance (ANOVA) was conducted to test whether 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 (pure stands of Scots pine, pure stands of oak and mixed woodlands) while the response was the total, above and below-ground amount of fixed carbon. The coefficient of variation was previously classified into three classes: coefficients less than 30 % were considered homogeneous stands, coefficients more than 40 % were considered heterogeneous stands and coefficients between 30 and 40 % were considered intermediate stands. As the structure factor was not significant on the first attempt, the ANOVA was repeated using only mixture composition as the explicative factor (results shown below).

20.5.2 Results and Conclusions

Significant differences were found between the total, above and belowground CO2 stock and specific composition (Table 20.9). Forest structure (coefficient of variation of diameter at breast height distribution) was not a significant factor.

The results reveal that the pure woodland of *Pinus sylvestris* fixed the largest amount of total carbon dioxide (472.09 Mg ha-1), significantly higher than the amount obtained in pure woodlands of Quercus pyrenaica (152.82 Mg ha-1). The mixed stand fixed an intermediate quantity (376.67 Mg ha-1) (Table 20.9). As in the total carbon dioxide analysis, significant differences were found between the above and belowground fixed carbon dioxide and the specific composition (Table 20.9). Again, the pure woodland of *Pinus sylvestris* fixed the largest amount of carbon dioxide in above and belowground stocks, statistically higher than the values obtained in the pure woodland of Quercus pyrenaica. The mixed woodlands obtained an intermediate value, which was not statistically different from that of the pure stands.

20.6 Final Remarks

Management objectives and techniques in woodlands dominated by *Quercus* species 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 and cork 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 $ha^{-1}$of carbon dioxide in 100 year old plantations. Even aged beech forest in Atlantic Europe, during the same period, fixed 150 Mg ha^{-1}. Although the species analysed were different from this study, the results were similar to pure *Q. pyrenaica* woodlands. Other studies (Lal 2004) have indicated that woodlands growing in tropical areas can restore 170 Mg ha^{-1} 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 regarding below-ground biomass.

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Chapter 21
Forest Management in the Sahel and Ethiopian Highlands and Impacts on Climate Change

Wubalem Tadesse, Mulugeta Lemenih, and Shiferaw Alem

Abstract Forest resources provide numerous ecosystems services that contribute to building resilient socio-ecological system in the Sahel. They are important natural capital used as safety net during times of drought, which is a common phenomenon in the Sahel region including Ethiopian highlands. The objective of this paper is to indicate the significance of the forests and vegetation of the Sahel region and Ethiopian highland for climate change adaptation and mitigation. There is high amount of carbon stored in the soil and above ground biomass of the forests and vegetation in both the Sahel and the Ethiopian highlands. Both vegetation resources are important for climate change adaptation and mitigation purposes. Despite all these and many other important roles of these forests, they are facing degradation and deforestation problems, in which efforts has to be made for the sustainable management of the forest ecosystems.

21.1 Introduction

The vast Sahel region covers approximately 2.5 million km^2 (ca. 5500 km long and 450 km wide) of land at the southern fringe of the Sahara desert. It ranges geographically from 14° to 18° N latitude in the western part and 12° to 16° N latitude in the eastern section, stretching from the Atlantic Ocean to the Red Sea and spanning parts of 14 African countries, including Ethiopia. The Sahelian belt thus crosses the African continent from west to east almost parallel to the equator. It is described as a unique eco-climatic and biogeographic zone within the African continent (Le Houérou 1980).

W. Tadesse (✉)
Ethiopian Environment and Forests Research Institute, Addis Ababa, Ethiopia
e-mail: wubalem16@gmail.com

M. Lemenih
Farm Africa, Ethiopia office, 5746, Addis Ababa, Ethiopia
e-mail: elerohi@yahoo.com

S. Alem
Mendel University in Brno, Zemedelska 3, 61300 Brno, Czech Republic
e-mail: shife19@yahooo.com

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The combination of geographic location and strong dependence on rain for main economic activities (crop and livestock farming) has made the Sahel highly vulnerable to the impacts of climate change (UNEP 2011). Close to 80 % of the population in the Sahel depends on natural resources for their livelihood, mainly through agriculture and forest products (UNEP 2011). Agriculture is the primary economic sector, generating employment for more than 60 % of the active population and contributing the lion's share to the GDP of many nations in the region. In Ethiopia, nearly 83 % of the active population is employed in agriculture (Mellor and Dorosh 2010). Livestock is another, perhaps more important activity in terms of livelihood, but is also under increasing pressure from climate change (Larwanou 2011). Climatic variability, land and biodiversity degradation, fragile ecosystems and desertification combined with poor institutional and human capacities make sustainable development very difficult (Tacko et al. 2006). Most of these characteristics are common to countries in the Sahelian region and the Ethiopian highlands, where a high population growth rate (over 2.6 % per year) and a high urbanization rate (7 % per year) along with unsustainable extraction of natural resources by local communities exemplifies a severely degrading ecosystem (Teketay et al. 2010; Larwanou 2011).

Forest vegetation is the other major source of livelihood that supplements income from farming, herding or fishing (African Development Bank 2003). The vegetation of the Sahel is unique, with characteristic dwarf trees and shrubs of mainly *Acacia* species. The natural vegetation of the Sahel is well-adapted to the harsh ecological conditions prevailing in the region, where drought and poor soil are typical. Forest vegetation provides mainly biomass energy, firewood and charcoal (Fall 2004). Forests also represent the main food source for livestock during dry seasons and drought periods: the most prominent problem in the Sahel. Non-timber forest products such as shea butter, gum arabic and wildlife resources complement household food and supply medicinal and cash income needs (Konaté 2012). These products constitute a significant safety net that allows households to withstand extended droughts and fill income gaps when crops fail. Forests also represent an essential community asset for adapting to climate change.

The vegetation in the Sahel offers a wide range of environmental services and benefits. It protects soils from erosion and stabilizes climate, which mitigates the impact of harsh environmental conditions and reduces the risk of extreme weather events such as droughts and floods. Though low to the ground and dominated by only a few species, the vegetation there hosts unique biodiversity of global importance. Products from Sahel species also support many national economies in the region (Chikamai and Enrico 2003). Species such as *Acacia senegal* and *Acacia seyal* are among those of great economic importance for commercial gum production. Others such as Commiphora and Boswellia species are very significant to local and global economies. The Sahelian shrublands also play their part in the terrestrial carbon cycle by storing large carbon stocks both in biomass and soil systems.

Ethiopia is located at the eastern fringe of the Sahelian belt. However, only the low-lying arid and semi-arid parts of the country, including the Rift Valley areas below 1500 m.a.s.l., present typical Sahelian climate, vegetation and land use. Above 1500 m.a.s.l. the Ethiopian highlands feature a distinct climate (rainfall, temperature) and vegetation. The highlands are relatively wetter and cooler, with tropical and sub-tropical vegetation. Ethiopia encompasses the majority of the eastern Africa highland massif and is the carbon storehouse of the region.

The objective of this paper is to review and synthesize forest management practices in the Sahel, especially Ethiopia, from the perspective of climate change, mitigation and adaptation.

21.2 Sahelian Forests

The Sahelian region covers a diverse ecosystem spanning from poorly vegetated areas near the Sahara desert to well-vegetated areas bordering lush tropical forests further south. In the Sahelian belt, vegetation typically consists of grassland (including *Cenchrus biflorus, Schoenefel diagracilis* and *Aristida stipoides*) and grass savanna intermingling with woodland and shrubland patches. The climate in the south is predominately semi-humid and humid tropical with equatorial and tropical rainforests, while in the north desert and semi-desert scrub vegetation gradually give way to tall grass savannah, followed by savannah woodland (Strahler and Strahler 2005). Species in the *Acacia* genus (i.e. *Acacia seyal* or *Acacia senegal*) along with *Commiphora africana*, *Balanites aegyptiaca*, *Faidherbia albida* and *Boscia senegalensis* are prevalent in the southern grass savannas and woodlands. The north Sahel is dominated by scrub species such as *Panicum turgidum* and *Aristida sieberana* (Anonymous 2014).

The history of Sahelian forestry is strongly linked to environmental catastrophes such as drought and famine, and political crises. International organizations have been involved in reforming the way forests and natural resources are managed in Sahel. Over the last 40 years, after the first dramatic droughts, forestry paradigms in the Sahel have changed at least three times. The first was characterized by state-led development of exotic species plantations in hopes of high biomass productivity; the second was marked by the co-evolution of the environmental and political contexts, and third featured integrative (farming and forest development) and individualistic approaches such as agroforestry and farmer-managed regeneration (Gautier and Peltier 2000). However, until recently little progress has been registered in terms of forest and tree regeneration, while wood demand continues to increase (Sendzimir et al. 2011).

Traditional agroforestry practices are the most prevalent form of forest and land management in the Sahel (Larwanou 2011). Agroforestry contributes to conservation of biodiversity and facilitates adaptation to climate change by increasing ecosystem resilience against climate extremes (Tacko et al. 2006).

Agroforestry in the Sahel (including more than 10 million ha in Niger) varies with rainfall, soil types and farming practices and management (Larwanou 2011). Agroforestry enhances the resilience of agricultural systems against the adverse impact of rainfall variability (drought, floods, modified weather patterns) as well as soil erosion, pests, diseases and weeds (Mbow et al. 2014). The adequate integration of trees into agricultural landscapes through agroforestry techniques allows for diversified and sustainable crop production. This strengthens the socio-economic resilience of rural populations by generating alternative or additional income sources for farmers while providing environmental benefits (Boffa 1999; Leakey et al. 2005; Tacko et al. 2006). Trees such as *Parkia biglobosa, Vitellaria paradoxa, Tamarindus indica* and several others provide many end-use products, some of which are sold to generate cash income (Boffa 1999). The multi-purpose *Parkia biglobosa* tree, for example, can generate up to US$ 270 annually per household, which is twice as much as crop income (Tacko et al. 2006).

Although natural forests are rapidly declining, agroforestry trees are becoming important suppliers of wood and non-wood forest products. Climatic conditions restrict the interest to engage in large block plantation forest development, which don't occupy a vast area in the Sahel today. Under such conditions and constraints on forest plantation investment, agroforestry development is an important alternative to supply the much needed wood and non-wood forest products. Windbreak and shelterbelt agroforestry systems are widely used to stabilize shifting sand dunes and reduce the effects of dry winds on agricultural crops. Private woodlots also produce wood for commercial purposes (Larwanou 2011). The discovery of the "underground forest" (the deep root system that allows shrubs to re-grow into trees when pruned properly) led to a wide spread agroforestry development and successful reforestation in the Sahel since the early 1980s (Cunningham and Abasse 2005). The main tree species used in reforestation activities of this kind include *Bauhinia reticulata*, *Combretum species*, *Ziziphus spina-christi*, *Z. mauritiaca* and *F. albida*.

Low and unpredictable rainfall, inter-annual variability, extended dry seasons and poor soil conditions are the biggest inherent natural challenges to effective natural regeneration in the Sahel, combined with socio-economic pressures from overgrazing and unsustainable utilization of biomass. Climate change also hinders regeneration in the Sahel as aridity intensifies due to rising average air temperatures and increasingly scarce and unreliable rainfall. These factors impede effective seed germination and successful recruitment of seedlings. Conventional reforestation through planting of seedlings has not proven to be a viable option. However, a new technique has been introduced in the region, known as assisted natural regeneration or 'Farmer Managed Natural Regeneration'. It involves restoring the original tree vegetation on farmlands by nurturing and protecting spontaneous regrowth of tree seedlings and by using pruning techniques that allow young trees to grow faster (Garrity et al. 2010). With the help of this new approach thousands of hectares in the Sahel have been successfully restored, along with their ecosystem and carbon cycle functions.

21.3 The Ethiopian Highland and Its Forest Resources

21.3.1 Forest Resources of Ethiopia

The Ethiopian landscape hosts a wide range of vegetation resources, from characteristically Sahelian scrub in the low-lying arid and semi-arid zones to tropical rainforests at higher altitudes, where rainfall is abundant. The nine recognized vegetation formations in the country are: (i) dry evergreen Afromontane, (ii) *Combretum–Terminalia* (broad-leaved) deciduous woodland, (iii) *Acacia–Commiphora* (small-leaved) deciduous woodland, (iv) *Acacia–Commiphora* (small-leaved) deciduous woodland, (v) lowland dry forest, (vi) lowland semi-desert and desert vegetation, (vii) evergreen scrub, (viii) wetlands (swamps, lakes, rivers and riparian), (viii) moist evergreen montane forest, and (ix) Afroalpine and sub-Afroalpine (Teketay et al. 2010; Lemenih and Woldemariam 2010).

The highlands are defined as land areas above 1500 m a.s.l. They represent the largest mountain system in Africa, comprising over 50 % of the African afromontane land area (Teketay 1996; Yirdaw 2002). Two types of Afromontane vegetation can be found there: dry Afromontane forest and humid (moist) Afromontane forest. Different names have been employed to refer to dry Afromontane forests in Ethiopia, such as tropical high montane conifer forest (Logan 1946), montane dry evergreen forest, highland *Juniperus-Podocarpus* forest, dry montane forest, upland dry evergreen forests, upland dry evergreen forest and mixed upland evergreen forest, coniferous forest and undifferentiated forest (Teketay 2005). The dry afromontane forest mainly features conifer and evergreen species, namely *Juniperus procera, Podocarpusfalcatus*, and *Olea europaea*, while the moist afromontane forest features prevalent broadleaf species, including *Pouteria adolfi-friederici*, *Olea capensis*, *Prunus africana*, *Albizia schimperiana*, *Celtis africana*, *Cordia africana, Polyscias fulva, Schefflera volkensii, Schefflera abyssinica, Coffea arabica* and *Mimusops kummel. C. arabica* originated here; in fact, this type of forest is the only place where wild Arabica coffee exists and grows naturally (Tesfaye 2006). Some examples of humid forests in Ethiopia include the Harenna forest in Bale, Tiro-Boter Bacho and Belete-Gera in the Jimma area, the Bonga forest in Kaffa and the Godere forest in Gambella. Stands of *Yushania alpina*, an indigenous highland bamboo species, can also be found in the humid highlands between 2500 and 3400 m.a.s.l.

21.3.2 Challenges Facing Forest Resources in Ethiopia

In spite of its high forest potential, Ethiopian forest lands have suffered deforestation and forest degradation problems for a long time. In the early twentieth century, only 5 % of the Ethiopian Highlands were forested (Friis 1992). *Podocarpus* and *Juniperus* species cover less than 1 % of potential area (Reusing 2000), which was once covered close to 176,000 km^2 in the central, eastern and northern regions.

Conversion to cropland is the main direct driver of forest loss. Sedentary grain production has been the primary economic activity in the Ethiopian highlands for thousands of years, driving large-scale deforestation in the country (CSA 2008). The population increases by 2.6 % annually, and people depend heavily on natural resources such as pastures for grazing and wood for construction or fuel (Tilahun et al. 1996).

Dry afromontane forests can only be found around churches and in a few inaccessible pockets, due to deforestation of most of the highlands in northern, central and eastern Ethiopia (EFAP 1994; Teketay 1996; Wassie 2007). This has caused widespread soil erosion; about 60 % of the most serious soil erosion in Ethiopia occurs in the highlands. Severe erosion affects 28 % of the highlands and moderate erosion affects another 24 % (Bekele 2002). Other forest resources in Ethiopia are also threatened by high deforestation and degradation. Management efforts have been insufficient for achieving forest sustainability and biodiversity conservation (Lemenih and Woldemariam 2010).

21.3.3 Carbon Stock in Different Types of Forests of Ethiopia and the Sahel

Africa holds one of the largest global biosphere carbon reserves, mainly due to the Conga Basin tropical forest. The immense African woodlands and savannas, which cover roughly 50 % of the continent (Menaut et al. 1985), also contribute considerably to Africa's biosphere carbon reserve. Although their carbon per unit area is small, African woodlands and savannas hold considerable carbon stock due to their vastness (Bartholome and Belward 2005). Unfortunately, 20 % of the African contribution to global land-use CO_2 emissions comes from forest degradation and deforestation (Laporte et al. 2007). The most important emission from the African ecosystem, and the most relevant to the Sahel, comes from forest fires. African ecosystems generate 40 % of global fire emissions, most of which comes from savannah burning (van der Werf et al. 2006).

Ethiopian vegetation is generally intermediate in height, compared to typically low Sahelian vegetation or that of the taller moist tropical forests of the Congo Basin, for example. However, humid montane rain forest can be found in the southwest areas of the country. It stores massive carbon stocks, contributing to terrestrial carbon sink and mitigation of climate change. Data compiled from extensive literature is presented in Table 21.1.

Table 21.1 shows that the forest and tree vegetation of Ethiopia hold an estimated 12.19 billion tons of carbon, excluding soil carbon. This value suggests that the vegetation in the Sahel could play a vital role in climate change mitigation if properly managed, restored and conserved to enhance carbon stock. There are considerable differences in carbon stock among the various vegetation resources, which reflects the effects of climatic and edaphic factors on species composition, growth and subsequent carbon storage capacity. In fact, the number increases dramatically when carbon stored in the soil of forest ecosystems is included. Plants such as Acacia-Commiphora, for example, have low above-ground biomass for carbon stock but store large amounts of soil organic carbon in their forest ecosystems. Table 21.2 shows soil organic carbon (SOC) stock for various vegetation types in Ethiopia.

Table 21.1 Carbon stock in above and below--ground biomass by forest type in Ethiopia

Vegetation type	AGC[a]	BGC	Area (ha)[b]	C stock (t CO_{2e})
Acacia-Commiphora	8.51	2.043	20,100,000	777,756,100
Combretum-Terminalia	23.62	6.377	21,500,000	2,364,763,500
Dry Afromontane	189.911	39.628	1,450,000	1,220,382,350
Moist Afromontane	176.628	45.07	1,450,000	1,178,694,367
Bamboo	95.097	25.443	1,273,070	562,671,478.6
Agroforestry	54.519	15.721	21,298,529	5,485,365,149
Plantation	144.13	34.59	909,500	596,001,413.3

[a]AGC and BGC stock calculations taken from data compiled as tier 2 for national MRV, from a review of over 100 recent publications.
[b]Data obtained from a recent forest sector report that estimated forest area of Ethiopia in 2013 (Anonymous 2015).

Table 21.2 Soil organic carbon stock (SOC) by forest type in Ethiopia (Depth 0.6–1.0 m)

Vegetation type	SOC (t /ha)
Acacia-Commiphora	106.76
Combretum-Terminalia	142.54
Dry Afromontane	187.86
Moist Afromontane	187.99
Bamboo	ND
Agroforestry	129.21
Plantation	ND

ND no data

21.4 Forest Ecosystem Management for Climate Change Adaptation and Mitigation in Ethiopia

Forest management in Ethiopia is based on the multiple use paradigm, with differences according to forest types. While planted forests are mainly managed to supply timber and construction materials, natural forests and woodlands are managed primarily for ecosystem services such as conservation of biodiversity and watershed protection. Fuelwood in the form of firewood and charcoal is the main product harvested from the natural forests and woodlands of Ethiopia and the entire Sahel region. Heavy forest degradation has limited timber production in Ethiopia, hence legal logging is generally banned. However, illegal poaching of valuable timber species continues to increase degradation. To reduce the risk of deforestation and degradation, Participatory Forest Management (PFM) and the introduction of Payment for Ecosystem Services (PES) have recently been introduced in Ethiopia, aimed at incentivizing local communities to engage in sustainable forest management. Carbon credit provisions within the framework of International Climate Change Convention such as the Clean Development Mechanism (CDM) and Reduction of Emissions from Deforestation and Forest Degradation (REDD) seek to encourage and foster participatory forest management. Different initiatives of this type have been developed in Ethiopia including a successful CDM afforestation/reforestation (AR) project called Humbo Farmer's Assisted Natural Regeneration. This is the first African-scale AR CDM project. Ethiopia has a number of AR pipelines in place, based on CDM and REDD+ projects.

In Ethiopia it is well-recognized that deforestation and forest degradation should be reduced by restoring forests that contribute to climate change mitigation and adaptation, and a green economy. In its Climate Resilient Green Economy (CRGE) strategy issued in December 2011, the Ethiopian Government identified the forestry sector as one of the pillars of the green economy envisioned for the country by 2030 (FDRE 2011). The part of the CRGE strategy relevant to the forestry sector aims at 'increased GHG sequestration in forestry through protecting existing forests and re-establishing forests. More specifically, the forestry pillar proposed (FDRE 2011):

- A reduction in the demand for fuelwood via the distribution and use of fuel-efficient stoves and/or alternative-fuel cooking and baking techniques (such as electric, LPG, or biogas stoves) to reduce forest degradation from unsustainable fuelwood harvesting
- Establishment of about two million hectares of new forests through afforestation and reforestation, improving the management of three million ha of natural forests and woodlands to increase carbon sequestration, and reducing GHG emissions in the sector.

21.4.1 Forest Ecosystem Management for Climate Change Mitigation

21.4.1.1 The Bale REDD+ Project

Bale REDD+ project is a pioneer initiative in an eco-region of south-eastern Ethiopia, located between 50°22′–80°08′N and 38°41′–40°44′E. The project spans 500,000 ha of dry and moist highland forests in this Eastern Afromontane biodiversity hotspot (OFWE 2014).

In this area the annual deforestation rate from 2000 to 2011oscillated between 1.1 % for moist evergreen forests and 6.7 % for dry evergreen forests. The objective of the Bale REDD+ project is to reduce deforestation of about 84,000 ha of forests and consequently reduce emissions by a net 37 million CO_2e GHG over the project life period (2012–2031) (OFWE 2014).This project will be co-managed by the government and local communities using the Participatory Forest Management approach, which involves transparent negotiations and benefit sharing through consensus. Local communities will receive 60 % of the revenue from emission reduction payment.

21.4.2 Afforestation and Reforestation CDM Project

The Humbo Farmers' Assisted Natural Regeneration CDM project is restoring 2700 ha of degraded native forest. The overall goal of the project is carbon sequestration in regenerating, bio-diverse native forests while providing incentives in the form of carbon credit to local communities that participate in restoration. This also serves the related objective of reducting poverty in the Humbo area (Ethiopia).

The project relies on a technique called Farmer-Managed Natural Regeneration (FMNR), which has been developed and refined for over 20 years to promote tree farms and forest regeneration in West Africa. This technique involves area exclosure and enrichment by planting nursery seedlings of selected species. Most of the species used are indigenous, but in some cases exotic tree species have been introduced. Local communities in the area were organized to form Community Based Organizations (CBOs) that determined collective roles, responsibilities and rights to the forest.

The Humbo Project is considered the first large-scale Clean Development Mechanism (CDM) forestry project in Africa. Carbon credits were purchased by the World Bank Bio-Carbon Fund, creating important revenue for local communities and providing an example for similar projects. The expected total carbon sequestration of the project is about 200,000 million tons of CO_2e by 2017. The World Bank Bio-Carbon Fund has purchased 165,000 tons of carbon credits and will provide an income of more than US$700,000 to local communities over a minimum of 10 years. The sale of timber products and carbon credits not purchased by the World Bank can generate extra revenue for participating communities (World Vision 2010).

Forest restoration provides environmental services beyond the sequestration of greenhouse gas emissions, such as greater biodiversity and connectivity among fragmented forest resources in the project area and refuge for local and migratory species. The fragile water catchment area is also protected, and the project prevents water and soil erosion and flooding. The sediment runoff that currently threatens the fragile ecosystem of Lake Abaya (30 km downstream) is being reduced, which helps maintain the springs and subterranean streams that provide water for the region (World Vision 2010).

Direct income for communities through sustainable harvesting of forest resources constitutes another key social benefit. Local communities gain robust land use and land access rights, turning their lands into important, valuable and legally recognized assets. Additionally, communities can invest carbon revenues from the project in local infrastructure and food security activities, which are determined through a participatory process aimed at improving the livelihood of the entire community.

21.4.3 Forest Ecosystem Management for Climate Change Adaptation

More than 480 species of wild trees and shrubs in Ethiopian forests have been recognized as important traditional food sources (Lemenih and Woldemariam 2010). These species have multiple direct and indirect roles in sustaining local communities and safeguarding them against the impacts of climate change. Forests and other

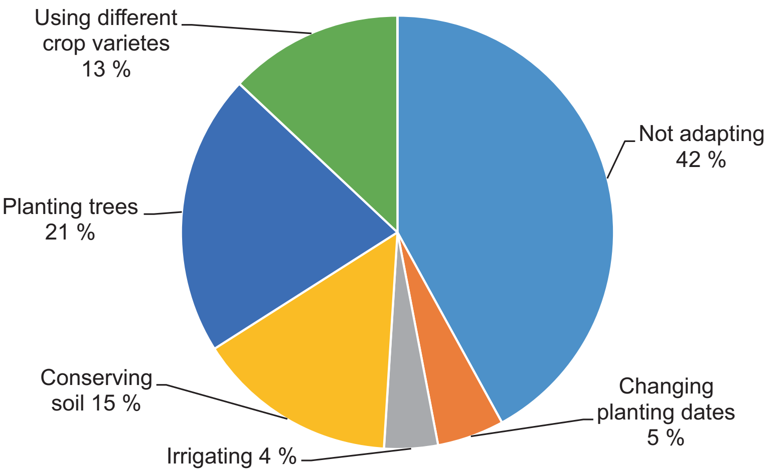

Fig. 21.1 Farmers' preferred options for adapting to climate change in Ethiopia (Source**:** Deressa et al. 2008)

vegetation resources provide consumable (edible) forest products and/or products and services that can be sold to meet local needs.

Across Ethiopia, forests and trees outside forests are among the options available for adapting to climate change (e.g. Deressa et al. 2008; World Bank 2010). In their study involving the Blue Nile basin, Deressa et al. (2008) found that planting trees was the number one adaptation measure proposed by local communities (Fig. 21.1).

Different studies (e.g. Bluffstone et al. 2008; World Bank 2010; Lemenih 2011) have illustrated the growing importance of trees for household income and asset building, thereby reducing community vulnerability to the effects of climate change in Ethiopia. During times of stress, vulnerable households look to the income diversification potential of forests and tree resources (along with temporal or permanent migration, charcoal or timber sales, changing consumption patterns, and drawing on livestock or savings). Box 21.1 presents illustrates the role of trees and forests in Ethiopia's Central Rift Valley.

Box 21.1 Role of forests in coping with drought and crop failure in Central Rift Valley The Central Rift Valley (CRV) of Ethiopia covers an area of 1400 km^2 and about 80 % of the population is rural (CSA 2006). The area is a semi-arid dryland with erratic rainfall. Mixed farming – crop production (primarily rain-fed) and livestock rearing – are the mainstays of local populations. The major crops grown in the area are maize, wheat, teff (*Eragrostis tef*) and barley. Recurrent droughts affect crop yields and animal productivity, due to effects on fodder and feed availability. The southern most part of the CRV, near Lake Abijata-Shalla and Lake Langano, suffer especially from high levels of food insecurity, which has been increasing in recent decades for a variety of reasons, including changing climatic patterns.

(continued)

Box 21.1 (continued)

As a result, the once densely-wooded area has been almost completely deforested (Garedew et al. 2009). Food production is inadequate to cover the annual needs of most households in the area, even in normal rainfall conditions. This is mainly because of the poor agriculture performance (low productivity), population growth and lack of productive assets to access food. Garedew (2010) identified eight types of non-farm activities that supplemented income from the constrained agriculture. Non-farm income sources mainly consist of forest-based activities (sale of fire wood, charcoal and thatching grass), fishing, sand sale, safety-net transfer, sale of labor, sale of salt-rich soil, petty trading and sale of traditional drinks. Households were involved on average in three types of non-farm activities. Forest-based, non-farming activities were the most common, while fishing and sale of sand were other frequently mentioned activities. Reduction of food portions and meal frequency or selling charcoal/firewood to buy food were the coping mechanisms used by most households during periods of chronic food insecurity. The most common coping activity of the farmers during crop failures has been the burning and sale of charcoal. Forest based income was the second largest after farming, accounting for 13.3 % of the total and more than 50 % of household incomes.

Forests also assist adaptation to climate change through provision of ecosystem services that support the sustainability of other sectors. For instance, the main problems pre-disposing Ethiopian agriculture to climate change impact are soil degradation such as poor soil organic matter content. Other serious and related issues include fertility degradation due to excessive soil-water erosion, the use of crop residue and animal dung as fuel rather than recycling them to maintain soil fertility and inadequate water retention in a watershed that could be used for irrigation. The presence of forests in a landscape mitigates these multiple problems by providing ecosystem services such as erosion control and regulation of water flow (Box 21.2). Watersheds with good forest cover have a regulated flow of low-sediment water that makes small- to medium-scale irrigation possible. Moreover, animal husbandry in Ethiopia relies on free grazing, mainly forest grazing, for fodder. Forests therefore support animal husbandry and income diversification.

Ethiopia has one of the highest erosion rates in the world (two billion metric tons of soil per year). Erosion caused soil degradation reduces primary productivity by 2–3 % in terms of annual agricultural gross income (Yesuf et al. 2005). Deforestation increases soil erosion and significantly reduces biodiversity. Inadequate forest cover magnifies the effects of floods or soil erosion on crops.

At present, only 22 % of Ethiopians (86 % of which live in cities) have access to an improved water source. Ethiopian forests play an essential role in water supply. This role is threatened by deforestation that could aggravate water shortages. The lakes and rivers that provide vital drinking water for people and livestock are being polluted by sedimentation from soil erosion. Rural springs and wells could dry up as the water tables fail to regenerate due to lack of plant cover. Currently, many water reservoirs, dams and canals are drying up. Water scarcity for agriculture is not always caused by low absolute rainfall; the problem stems mainly from rapid runoff and poor infiltration due to poor ground cover. This situation exposes crops and livestock to ever greater risks, as farmers must travel greater distances to access water for their households and animals. The Ethiopian government is encouraging the construction of more irrigation systems, but maintenance of irrigation infrastructures is becoming costly due to excessive sedimentation/siltation. Unless watershed and land management plans are properly integrated, water scarcity and sedimentation will intensify and opportunities for cost-effective climate change adaptation will be greatly diminished.

Box 21.2 Examples of socio-economic and environmental roles of forests and woodland vegetation in Ethiopia that can enhance adaptation to climate change

Socio-economic functions	
Food security	Forests and trees outside forests (TOF) play multiple food security roles in Ethiopia. They provide diverse edible wild food. More than 480 species of wild trees and shrubs have been recorded as important traditional food sources in Ethiopia, most of which are forest plants (Lemenih 2011). Most of these species (around 72 %) have edible fruits and/or seeds, while vegetative parts (leaves, stems and tubers/roots) can be eaten from others. *Moringa stenopetala*, for example, provides edible, nutrient- and vitamin-rich leaves and shoots, which also have medicinal properties. Moringa is widely used as a household food source in the semi-arid regions of southern Ethiopia, particularly in Konso (Tadesse 2010). Fruits from *Cordia africana*, *Balanites aegyptiaca*, *Dovyalis abyssinica, ficus spp., Carissa edulis* and *Rosa abyssinica* are commonly consumed in rural Ethiopia. Likewise, fruits from *Opuntia ficus-indica* and *Borassus aethiopum* are consumed and sold in markets for cash in the Tigray and Afar regions. Similarly, many wild animal species including fish, mammals and birds, are utilized for food or as hunting trophies. Forests and TOF also provide cash income through employment or trading forest products for food. More than 35,000 women harvest fuelwood in Addis Ababa alone, with 82 % of them fully dependent on the business (WBISSP 2004). Forest products sold from natural or farm trees are reported to provide between one-third and one-half of household income. Women and the poor are the biggest beneficiaries in this regard (Lemenih 2011). Forest grazing plays a key role supporting livestock, providing 10–60 % of livestock feed during dry and wet seasons, respectively.

(continued)

Box 22.2 (continued)

Socio-economic functions	
Health	Medicinal plants are an important component of the healthcare system in Ethiopia, and widely used for human and livestock care across the country. About 600 medicinal plant species are recorded in Ethiopia. The plant-based health industry provides around 346,000 employment opportunities through the use of 56,000 tons of medicinal plants per year in Ethiopia, most of which are harvested largely from wild plant stocks. In Ethiopia approximately 48 million inhabitants (80 % of the population) use more than 700 medicinal plants, with an annual value of US $74 million (Mender et al. 2006).
Energy	Biomass is the major source of energy, accounting for 97 % of total domestic energy consumption, 78 % of which is woody biomass (WBISPP 2004). Demand for fuelwood in Ethiopia is nearly 20 times greater than the demand for all other forest products together. In the 1990s, the demand was 80 million m^3 per year but recently demand has been estimated at 109 million m^3 per year (FAO 2005).
Employment	About 50 % of the labor force is employed in fuelwood and charcoal production and collection, 34 % in forest plantation work, and 2 % in forest industries (Bekele 2011). In Addis Ababa alone, over 35,000 women engage in commercial fuelwood harvesting, with 82 % of them fully dependent on income from this activity (WBISSP 2004). Additionally, the important gum and incense sub-sector employs between 20,000 and 30,000 seasonal workers each year, while as many as 80,000 traditional healers (9000 officially organized and registered) are assumed to exist in the country (Lemenih 2011).
Environmental functions	
Watershed	The Ethiopian landscape is hilly and mountainous. Forests help conserve soil and water by improving and regulating water flow. Deforestation is responsible for the excessive erosion of nearly two billion tons of soil annually. Forests and vegetation significantly reduce soil erosion and sedimentation downstream (Descheemaeker et al. 2006). After area exclosure in Tigray, for example, specific sediment yield was reduced by over 50 %. At the catchment scale, expanding exclosure areas will decrease runoff and increase infiltration and evapotranspiration (Descheemaeker et al. 2006). Trees on farmland (agroforestry) also create favorable microclimate conditions. They provide shade zones several degrees cooler than un-shaded areas, thereby moderating the effects of increasing temperature associated with global climate change, and reducing soil erosion and run-off from cultivated lands. With the increasing emphasis of the Ethiopian government on developing irrigation, the watershed protection effect of forests and vegetation should be given high priority. Vegetation plays several hydrological functions, such as reducing surface run-off and increasing infiltration; improving soil porosity and organic matter content, and improving soil-water content (green water availability) and regulated water distribution (watershed protection).

(continued)

Box 22.2 (continued)

Socio-economic functions	
Carbon storage	Ethiopian forests and woodlands carry a large carbon stock and high sequestration potential. Estimated carbon values are expected to reach 123.3 M Birr per year for the Munessa-Shashamane forest, 16 M USD for the Bale eco-region; and 139 M Birr per year for the Sheka forest (Tadesse 2008). Most importantly, these forests will assist the Ethiopian government's green economy development strategy by sequestering and buffering greenhouse gas emissions in a growing and expanding economy.
Biodiversity	The forests and woodlands of Ethiopia house a rich biological diversity that is also economically important. Coffee serves as an example of this. Forest conservation is the nation's insurance for a future that involves changing climate.

21.5 Conclusion

The Sahel is well-known for its repeated drought episodes. Climate change and aridity increase the vulnerability of the region. Unsustainable use and subsequent loss of vegetation, along with land degradation expose the Sahel to socio-ecological disasters. However, ecology holds the potential to reverse the situation and offers significant opportunities for climate change mitigation and adaptation through improvement of vegetation resource management. Despite the generally low stature of its vegetation (compared to rain forests) the Sahel possesses significant vegetation resources and unique biodiversity. Vegetation has been playing a tremendous role as a safety net, buffering society against the impacts of climate change by filling food and cash gaps. Sahelian vegetation also stores a considerable carbon stock in soil and biomass, helping to regulate the terrestrial carbon cycle. The potential for carbon sequestration in the Sahel is as vast as its geography. Recent successful experiences in restoration of the vegetation resources of the Sahelian region of West Africa through farmer-assisted regeneration give good testimony to this potential. For possibility to become reality, however, institutional and technical changes that fit specific contexts must be investigated, developed and implemented.

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Index

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