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SUSTAINABLE FOREST MANAGEMENT – CURRENT RESEARCH

Edited by **Jorge Martín García**
and **Julio Javier Diez Casero**

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Contributors

Manuel Marey-Pérez, Carlos Jose Álvarez-López, Luis Franco Vázquez, Mario Sporcic, Stephane Couturier, Alessandro Paletto, Isabella De Meo, Fabrizio Ferretti, Paolo Cantiani, Marco Focacci, Anusha Rajkaran, Janine Adams, Tohru Nakajima, Nikolay Strigul, Peter Newton, Joseph Molnar, John Schelhas, Pablo MartÃn López Bernal, Guillermo Emilio Defossé, Claudia Pamela Quinteros, José Omar Bava, Teresa Fonseca, Bernard R Parresol, Carlos Marques, François De Coligny, Goio, Geremia Gios, Alessandro Gretter, Rocco Scolozzi, Jožica Gričar, Ludmila La Manna, Anne Stenger, Serge Garcia, Max Bruciamacchie, Andre Silva Monteiro, Gustavo Perez-Verdin, Jose Navar, Yeon-Su Kim, Ramon Silva-Flores, Jacek Siry, Neelam C Poudyal, J. M. Bowker, Nicholas Clarke, Jørgen Bo Larsen, Roman Gebauer, Jindřich Neruda, Radomír Ulrich, Milena Martinková, Frederick Cubbage, Steverson Moffat, Kathleen McGinley, Liwei Lin, Guy Robertson, Jorge Martín García, Julio J. Diez

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Meet the editors

Dr Jorge Martín García is Assistant Professor in the Forest Engineering and Environmental Department at the University of Extremadura (Spain). He has both experience in International and Spanish Forestry Environmental Sustainability. He has carried out research stays at the University of Aberdeen (UK), Finish Forest Research Institute (Finland), French National Institute for Agricultural Research (France) and University of San Luis (Argentina) along with some other research universities. He has also worked as a Lecturer at the college of Forestry (León, Spain) and as a Forest Manager for the Government of Spain and for some private companies. He holds a Forest Engineering Degree (Best Student Record Award) and a Master in Conservation and Sustainable Use of Forest Systems at the University of Valladolid (Spain), and his PhD Topic is "Sustainable Forest Management: in search of indicators".

Professor Julio J. Diez is a Professor at the University of Valladolid, Sustainable Forest Management Institute located at Palencia city. He has 17 years of teaching experience in the field of forestry. He holds a PhD in Forestry from the Polytechnic University of Madrid, and a Master's Degree in Biology from the University of Salamanca. He has published more than 50 scientific papers in peer-reviewed journals, and served as an invited keynote speaker at international symposia and congresses around the world. He has coordinated different research projects in forestry, including some international projects in Sustainable Forest Management.

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Preface

Sustainable forest management (SFM) is not a new concept. However, its popularity has increased in the last few decades because of public concern about the dramatic decrease in forest resources. SFM is generally implemented using criteria and indicators (C&I) that define forest management standards, and several countries have established their own sets of C&I within the framework of different international or regional processes. Nevertheless, none of the C&I systems have been universally accepted and future research should consider the current and future indicators.

This book summarises some of the recent research carried out to test the current indicators, to search for new indicators and to develop new decision-making tools that can be used in forest management to assess and implement SFM. The book is divided into seven sections, including a brief introduction and six thematic blocks (carbon and forest resources, forest health, productive and protective functions, biological diversity, socioeconomic functions and decision making tools).

The Introduction provides an overview of SFM and forest certification. A brief analysis of the current state of the World's forests is presented, followed by a broad summary of the past and current situation of SFM, C&I and forest certification, concluding with future challenges.

The section on carbon and forest resources includes four articles. In the first paper, Couturier describes the status of accuracy assessment of land use and land cover maps and National Forest Inventory maps, and considers the usefulness of such maps for implementing SFM in high biodiversity areas. The author also analyzes the accuracy assessment methods used for four regions of Mexico. López-Bernal et al. contribute with an interesting study of the evolution of Lenga forests in the Argentinean Patagonia and the applicability of a selective silvicultural system, the "Group Selection System". These authors conclude that this system is a valid tool for making two key aspects of SFM compatible. These aspects are optimal regeneration and the current local production system, which is characterized by lack of financial and technological capacity. Probably the most important challenge as regards SFM is the deforestation and degradation of Amazon forests. Great efforts have been invested in deforestation monitoring programs, although the high costs make this system unviable. Monteiro and Souza Jr suggest the use of remote sensing techniques to detect, map and monitor logging activities at the scale of the Amazon, which would help improve forest management, reduce illegal logging and improve the quality of harvesting. In the final paper in this section, Nakajima discusses the effects of the Japanese carbon offsetting system, with respect to carbon price, on the regional carbon stock and timber production. The study uses simulations to investigate the effects of carbon price on timber production and carbon stock, and examines the consequences for harvesting strategies in the actual forest area formally identified in the Japanese Verified Emissions Reduction system.

The section on forest health includes three papers. Gričar presents an interesting review of the potential of xylem, phloem and cambium parameters as indicators of tree vitality status. This author concludes that the ratio between xylem and phloem, and to a lesser extent the widths of xylem, phloem and dormant cambium, are related and indicate the health condition of a tree, and therefore may be used as indicators of forest health. Traditionally forest health has been assessed at stand level. However, entomologists and pathologists are conscious of the importance of landscape level for detecting and preventing the spread of pests and diseases. In this regard, La Manna describes some useful methods of evaluating the effects of abiotic factors on forest diseases at landscape level and of developing risk models as tools for forest management. The study by Schelhas and Molnar examines how sociological perspectives on collective action and common-pool resource theory can contribute to the health and management of Southern pine forests. Some implications for the motivation of non industrial private forest owners and communication between them are discussed.

The fourth section combines protective and productive functions, because a good balance between the benefits of both is key to the success of SFM. This section comprises two papers that evaluate the effect of harvesting intensity on water and soil. Traditional forest management is changing due to a boom in renewable energy sources, particularly forest biomass. The review by Clarke addresses the current state of knowledge regarding sustainable removal of forest residues (branches and tops) for bioenergy purposes, and the author concludes that this practice may increase the risk of adverse effects on soil and water, among other effects. Soil compaction caused by forestry machines is the subject of a paper by Gebauer et al. These authors determine that the use of harvesters and forwarders without any prior site preparation is detrimental to soil properties and plant growth, and they propose some options to minimize such effects.

In the section on biological diversity, Larsen describes the history of nature-based forest management, suggesting this as the best option for attaining the most natural conditions in European forests, and discusses the Danish experience. The subject of the paper by Paletto et al. is deadwood. These authors studied the effect of management strategies on quantitative and qualitative features of deadwood, and report some results that may be very useful in helping forest managers to meet SFM demands.

From the previous sections it is clear that forests provide tangible and intangible benefits. The latter have generally not been valued economically, and therefore were underestimated until a few decades ago. In this regard, the different SFM initiatives (ITTO, MCPFE, the Montreal Process, etc.) have established a criterion involving socio-economic functions. The sixth section of the book, called Socioeconomic Functions, consists of four papers that expect to advance in the economic assessment of intangible benefits. The main objective of the paper by Goio et al. is to define the management policies that maximise the use of goods and services, ensuring that forests are managed sustainably. These authors focus on landscape and recreational function and show the experiences from the Alps, in particular the Logarska Dolina valley (Trento, Italy). The study by Perez-Verdin et al. focuses on hydrological services, specifically the economic valorization of watershed services as a means of achieving SFM. The authors analyze a case study in Mexico, where an incentive-based instrument (payment for ecosystem services) was implemented. They conclude that although this instrument is not the panacea for problems related to water quality and deforestation problems, it should be considered in designing SFM policies. The paper by Poudyal et al. provides a holistic view of the market potential and opportunities for making urban forest projects financially self-reliant and more sustainable. This information could be used to expand existing market protocols for carbon credits sourced from urban forestry projects, and to develop new protocols. Finally, the paper by Cubbage et al. deals with the legal, institutional, and economic C&I established for SFM in the U.S.

The section on decision-making tools includes seven papers. The development of models to predict the effect of different silviculture scenarios is the subject of four of the studies. Bruciamacchie et al. describe an economic model based on maximization of incomes from harvesting in relation to biological diversity, and analyze the demands for species diversity and timber supply and the link between timber production and diversity. In the same vein, Fonseca et al. present the ModisPinaster model as a useful tool for implementing SFM in maritime pine forests. This model enables simulation of different silviculture scenarios, thus providing forest managers with valuable information enabling them to achieve SFM standards. Moreover, Rajkaran and Adams developed a model for determining the harvesting intensity in mangrove forests, thus ensuring the viability of the tree population. The paper by Strigul describes different models ranging from individual to stand level, which incorporate the implications of crown plasticity for the optimization of the forest resources as a novel aspect. These models enable prediction of the effects of different management strategies or natural disturbances and provide a useful tool for forest managers in the decision-making process. The study by Marey-Pérez et al. considers a platform for Decision Support Systems in Galicia (Northern, Spain), which has proven quite useful and has been directly applied to SFM. The paper by Šporčić describes how multi-criteria methods can be used to analyze the choice of the best or at least satisfactory decision and thus contribute to more reliable planning and more objective decision making in forestry. The study by Newton describes an enhanced stand-level decision-support model for managing upland black spruce stand-types and demonstrates its operational utility in evaluating complex density management regimes involving initial spacing, precommercial and commercial thinning.

The papers included in the book should shed light on the current research carried out to provide forest managers with useful tools for choosing between different management strategies or improving indicators of SFM. We are indebted to all authors who submitted papers for consideration for publication in this book. We would also like to thank the editorial team at Intech for their assistance and support.

Jorge Martín-García[1,2] and Julio Javier Diez[1]
[1]Sustainable Forest Management Research Institute,
University of Valladolid – INIA, Palencia,
[2]Forestry Engineering, University of Extremadura, Plasencia,
Spain

Section 1

Introduction

1

Sustainable Forest Management: An Introduction and Overview

Jorge Martín-García[1, 2] and Julio Javier Diez[1]
[1]*Sustainable Forest Management Research Institute, University of Valladolid – INIA, Palencia*
[2]*Forestry Engineering, University of Extremadura, Plasencia*
Spain

1. Introduction

It is well known that forests provide both tangible and intangible benefits. These benefits may be classified according to ecological values (climate stabilization, soil enrichment and protection, regulation of water cycles, improved biodiversity, purification of air, CO_2 sinks, potential source of new products for the pharmaceutical industry, etc.), social values (recreational and leisure area, tradition uses, landscape, employment, etc) and economic values (timber, non wood forest products, employment, etc.). Although forests have traditionally been managed by society, it is expected that the current growth in the world population (now > 7,000 million people) and the high economic growth of developing countries will lead to greater use of natural resources and of forest resources in particular.

2. Global forest resources

The total forest area worldwide, previously estimated at 4 billion hectares, has decreased alarmingly in the last few decades, although the rate of deforestation and loss of forest from natural causes has slowed down from 16 million hectares per year in the 1990s to around 13 million hectares per year in the last decade (FAO, 2011). Nevertheless, the loss of forest varies according to the region, and while the forest area in North America, Europe and Asia has increased in the past two decades (1990-2010), it has decreased in other regions such as Africa and Central and South America, and to a lesser extent Oceania (Fig. 1)

There is growing public concern about the importance of the environment and its protection, as manifested by the fact that the total area of forest within protected systems has increased by 94 million hectares in the past two decades, reaching 13% of all the world's forests. Moreover, designated areas for conservation of biological diversity and for protection of soil and water account for 12 and 8% of the world's forests, respectively (FAO, 2010, 2011). Nevertheless, other statistics such as the disturbing decrease in primary forests[1] (40 million hectares in the last decade) and the increase in planted forests (up to 7% of the

[1] Forest of native species where there are no clearly visible indications of human activities and the ecological processes have not been significantly disturbed (FAO, 2010)

world's forests) (FAO, 2011) appear to indicate that to achieve forest sustainability, we must go beyond analysis of the changes in the total forest area worldwide.

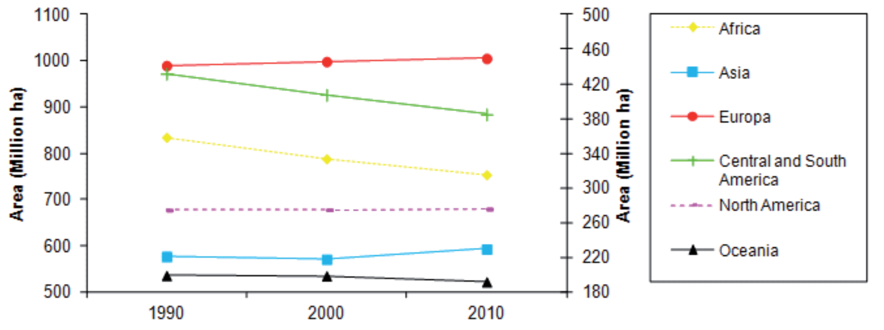

Fig. 1. State of World's Forests 2011 – subregional breakdown (Source: FAO, 2011). Africa, Asia, Europe, Central and South America and North America are represented in the left axis and Oceania in the right axis.

3. Sustainable forest management

The concept of sustainability began to increase in importance at the end of the 1980s and at the beginning of the 1990s with the Brundtland report (1987) and the Conference on Environment and Development held in Rio de Janeiro, Brazil, in 1992 (the so-called Earth Summit), respectively. Nevertheless, the need to preserve natural resources for use by future generations had long been recognised.

The negative influence of past use of forest resources, as well as the needs for continued use of these resources for future generations was already noted as early as the 17th century (Glacken, 1976, as cited in Wiersum, 1995). However, it was not until the 18th century that the concept of sustainability was specifically referred to, as follows: "every wise forest director has to have evaluated the forest stands without losing time, to utilize them to the greatest possible extent, but still in a way that future generations will have at least as much benefit as the living generation" (Schmutzenhofer, 1992, as cited in Wiersum, 1995). This first definition was based on the principle of sustainable forest yield, with the main goal being sustained timber production, and it was assumed that if stands that are suitable for timber production are sustained, then non wood forest products will also be sustained (Peng 2000). This assumption focused on the sustainability of the productive functions of forest resources, while other functions such as ecological or socio-economic functions were largely overlooked. This occurred because social demands for forests were mainly utilitarian. However, increased environmental awareness and improved scientific knowledge regarding deterioration of the environment have changed society's values and the global structural policy, which in turn have significantly influenced forest management objectives in 20th century (Wang & Wilson, 2007). Nevertheless, nowadays more and more researchers think climate change is changing the paradigm and sustainability shouldn't be referred to what we had before.

Although there is no universally accepted definition of SFM, the following concepts are widely accepted: "*the process of managing permanent forest land to achieve one or more clearly specified objectives of management with regard to the production of a continuous flow of desired forest products and services without undue reduction of its inherent values and future productivity* *and without undue undesirable effects on the physical and social environment*" (proposed by International Tropical Timber Organization: ITTO, 1992), and "*the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems"* (proposed by the second ministerial conference for the protection of the forest: MCPFE, 1993). The latter concept harmonizes ecological and socio-economic concerns at different scales of management and for different time periods. Nevertheless, both concepts are just refining the definition of sustainable development gave by the Brundtland Commission (1987) "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" to apply it to forests.

4. Criteria and indicators

The implementation of SFM is generally achieved using criteria and indicators (C&I). Criteria are categories of conditions or processes whereby sustainable forest management can be assessed, whereas quantitative indicators are chosen to provide measurable features of the criteria and can be monitored periodically to detect trends (Brand, 1997; Wijewardana, 2008) and qualitative indicators are developed to describe the overall policies, institutions and instruments regarding SFM (Forest Europe, 2011).

Different studies have pointed out the main characteristics of a good indicator. Thus, Prabhu et al. (2001) suggested seven attributes to improve the quality of indicators (precision of definition, diagnostic specificity, sensitivity to change or stress, ease of detection, recording and interpretation, ability to summarize or integrate information, reliability and appeal to users), whereas Dale & Beyeler (2001) established eight prerequisites to selection (ease of measurement, sensitivity to stresses on the system, responsive to stress in a predictable manner, anticipatory, able to predict changes that can be averted by management actions, integrative, known response to disturbances, anthropogenic stresses and changes over time, and low variability in response).

Although several criticisms have been launched against the C&I system (Bass, 2001; Gough et al., 2008; Poore, 2003; Prabhu et al., 2001), the popularity of the system is evident from the effort invested in its development in recent decades and from the large number of countries that are implementing their own sets of C&I within the framework of the nine international or regional process (African Timber Organization [ATO], Dry Forest in Asia, Dry Zone Africa, International Tropical Timber Organization [ITTO], Lepaterique of Central America, Montreal Process, Near East, Pan-European Forest [also known as the Ministerial Conference on the Protection of Forest in Europe, MCPFE] and Tarapoto of the Amazon Forest). Nevertheless, three of these processes stand out against the others[2], namely the ITTO, MCPFE and Montreal processes. The first set of C&I was developed by ITTO (1992) for sustainable management of tropical forest, and subsequently an initiative to develop C&I for sustainable management of boreal and temperate forests took place in Canada, under the supervision of the Conference on Security and Cooperation, in 1993. This first initiative reached a general consensus about the guidelines that should be

[2] Together, these three international C&I processes represent countries where more than 90% of the world's temperate and boreal forests, and 80% of the world's tropical forests are located.

followed by all participating countries. It was then decided that the countries should be split into two groups: European would establish the MCPFE and non-European countries the Montreal processes. The MCPFE process adopted a first draft of C&I in the first expert level follow-up meeting in Geneva in June 1994, which took shape in Resolution L2 adopted at the third Ministerial Conference on the Protection of Forest in Europe held in Lisbon (MCPFE, 1998), and improved at the subsequent Ministerial Conference held in Vienna (MCPFE, 2003). On the other hand, the Montreal process established its set of C&I in the Santiago Agreement (1995), with Criteria 1-6 improved at the 18th meeting in Buenos Aires, Argentina (TAC, 2007) and criterion 7 improved at the 20th meeting in Jeju, Republic of Korea (TAC, 2009).

Although the different processes have very different origins and have developed their own criteria, there are some similarities between the three major SFM programs (Table 1). The main difference concerns criterion 7, developed by the Montreal process (Legal, policy and institutional framework), which was imbedded within each of the criteria in the MCPFE process (McDonald & Lane, 2004) and the concept of which is similar to criterion 1 in the ITTO process (Enabling condition). One important difference between ITTO and the other two processes is that the former does not consider maintenance of the forest contribution to global carbon cycles.

ITTO process	MCPFE process	Montreal process
C1. Enabling condition	C1. Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles	C1. Conservation of biological diversity
C2. Extent and condition of forests	C2. Maintenance of forest ecosystem health and vitality	C2. Maintenance of productive capacity of forest ecosystems
C3. Forest ecosystem health	C3. Maintenance and encouragement of productive functions of forests (wood and non-wood)	C3. Maintenance of forest ecosystem health and vitality
C4. Forest production	C4. Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems	C4. Conservation and maintenance of soil and water resources
C5. Biological diversity	C5. Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water)	C5. Maintenance of forest contribution to global carbon cycles
C6. Soil and water protection	C6. Maintenance of other socioeconomic functions and conditions	C6. Maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies
C7. Economic, social and cultural aspects		C7. Legal, policy and institutional framework

Table 1. Criteria for sustainable forest management: comparison of three major programs

Other differences in indicators developed by the different processes have become apparent, and e.g. Hickey & Innes (2008) established more than 2000 separate indicators using the context analysis method. There are also substantial differences as regards the three major processes: the MCPFE process has 52 indicators (MCPFE, 2003), whereas the Montreal process has reduced the number of indicators from 67 (Santiago Agreement, 1995) to 54 (TAC, 2009), and the ITTO process has reduced the number of indicators from 66 in the first revision (ITTO, 1998) to the 56 considered at present (ITTO, 2005).

In light of the proliferation of C&I processes, the need to achieve harmonization has been widely recognised (Brand, 1997; Castañeda, 2000). Although the concept of harmonization is subject to several interpretations, harmonization should not be mistaken for standardization (Rametsteiner, 2006). Köhl et al (2000) has claimed that "harmonization should be based on existing concepts which should be brought together in a way to be more easy to compare, which could be seen as a bottom up approach starting from an existing divergence and ending in a state of comparability". Although there is not yet a common approach, considerable efforts have been made since the first expert meeting on the harmonization of Criteria and Indicators for SFM, held in Rome in 1995 (FAO, 1995), towards the search for a harmonization/collaboration among C&I processes through the Inter-Criteria and Indicator Process Collaboration Workshop (USDA, 2009). Advances in harmonization will minimise costs (avoiding duplication and preventing overlap), facilitate comparisons between countries and, overall, improve the credibility of SFM.

Although indicators are increasingly used, their utility is still controversial. Some authors have pointed out several weaknesses of the indicators, e.g. that they are often highly idealistic (Bass, 2001; Michalos, 1997), that they are a pathological corruption of the reductionist approach to science (Bradbury, 1996) or even that the same indicator may lead to contradictory conclusions according to the criterion and the scale. Nevertheless, there is general agreement that the advantages of the approach outweigh these limitations and that researchers should focus their efforts on testing the current indicators and searching for new indicators.

There are two key aspects involved in improving the current and future indicators, the use of a suitable scale and the establishment of a specific interpretation of each indicator. Although these have mainly been implemented at a national level, sub-national and forest management unit (FMU) levels are essential to assess SFM (Wijewardana, 2008). The FMU level has been considered as the finest scale in C&I processes. However it is well-known that for some indicators (mainly biodiversity indicators), another subdivision within this level may be necessary, such as plot, landscape and spatial levels, for correct interpretation (Barbaro et al., 2007; Heikkinen et al., 2004). In light of this level of precision and the fact that values of indicators are sometimes correlated with several different scales, managers and researchers should establish the most effective scale in each case, to avoid additional charges. Moreover, good indicators are not always easy to interpret in terms of sustainability, because most indicators do not exhibit a clear distinction/threshold between sustainability and unsustainability. In such cases, the achievement of sustainability should be considered on the basis of relative improvement in the current status of the indicator in question (Bertrand et al., 2008).

On the other hand, the scientific community must search for new indicators. Gaps in knowledge have been identified, and as these mainly involve ecological aspects, researchers should go further in investigating the relationships between type of forest management and ecological and socioeconomic functions. Thus, managers and researchers, with the support of scientific knowledge and public consultations, should be able to determine feasible goals, from socioeconomic and scientific points of view, since goals that are too pretentious may lead to a situation whereby SFM will not be promoted (Michalos, 1997). Only then can successful selection of new indicators of SFM be achieved.

5. Forest certification

In addition to the efforts of different states to develop C&I in the last two decades, a parallel process has been developed to promote SFM. This process is termed "forest certification". Forest certification can be defined by a voluntary system conducted by a qualified and independent third party who verifies that forest management is based on a predetermined standard and identifies the products with a label. The standard is based on the C&I approach and the label, which can be identified by the consumer, is used to identify products. Therefore, the two main objectives of forest certification are to improve forest management (reaching SFM) and to ensure market access for certified products (Gafo et al., 2011).

The first certification was carried out in Indonesia in 1990 by the SmartWood programme of the Rainforest Alliance (Crossley, 1995, as cited in Elliot, 2000). However forest certification became popular after The Earth Summit in Rio de Janeiro in 1992. Although important advances were reached at this summit, the failure to sign a global convention on forestry led environmental and non-governmental organizations to establish private systems of governance to promote SFM. In 1993, an initiative led by environmental groups, foresters and timber companies resulted in creation of the Forest Stewardship Council (FSC). Subsequently, other initiatives at international and national levels gave rise to many other schemes, e.g. the Programme for the Endorsement of Forest Certification (PEFC, previously termed Pan European Forest Certification), the Canadian Standards Association (CSA), the Sustainable Forestry Initiative (SFI), the Chile Forest Certification Corporation (CERTFOR) and the Malaysian Timber Certification Council, among others.

The area of certified forest increased rapidly in the 1990s and from then on more gradually, reaching 375 million hectares in May 2011 (UNENCE/FAO, 2011), which represents almost 10% of the global forest area. Although many forest certification systems were developed in the 1990s, only two schemes (PEFC and FSC) have been used for most of the forest currently certified throughout the world. The FSC scheme was established in 1993 to close the gap identified after the Earth Summit, and with more than 140 million hectares is the first program in terms of number of certified countries (81 countries) and the second system in terms of certified area at the moment (FSC, 2011). The PEFC scheme was established in 1999 as an alternative to the FSC scheme, and was led by European forest owners, who considered that FSC standards mainly applied to large tropical forests, but were inappropriate for small forest owners of European temperate forests. The PEFC scheme has gained importance because it endorses 30 national forest certification systems (Australian Forestry Standard, CSA, SFI, CERTFOR, etc.), and with more than 230 million hectares of certified forests is currently the largest forest certification system (PEFC, 2011). Although several authors have reported significant differences between FSC and PEFC (Clark & Kozar, 2011; Rotherham, 2011; Sprang, 2001), detailed analysis has revealed that FSC and PEFC are highly compatible, despite having arrived at their C&I by different routes (ITS Global, 2011).

Although forest certification began in tropical forests, the trend has changed and the scheme is now carried out in boreal and temperate forests. Almost 90% of forests certified by the two major programs (FSC and PEFC) are located within Europe and North America (Figure 2). More than half (54%) of the forests in Europe (excluding the Russian Federation) have already been certified, and almost one third of the forest area in North America has been certified (Figure 3). On the contrary, only about 1.5% of the forests in Africa, Asia, and Central and South America have been certified (Figure 3), despite the fact that more of half of the world's forests and almost 60% of primary world forests are located in these countries. The FSC and PEFC schemes display similar patterns of certification, since both mainly certify forests in Europe and North America. However, although the percentage of forest area certified by FSC in Africa, Asia, and Central and South America is only 16% of all certifications carried out by this scheme, this represents 75% of the forest areas certified in these regions. Furthermore, almost all certifications carried out in the Russian Federation are carried out by the FSC, whereas the PEFC has certified very few forests in this region. On the other hand, most forest certifications in Europe (excluding the Russian Federation) and North America have been carried out by PEFC (Figure 2).

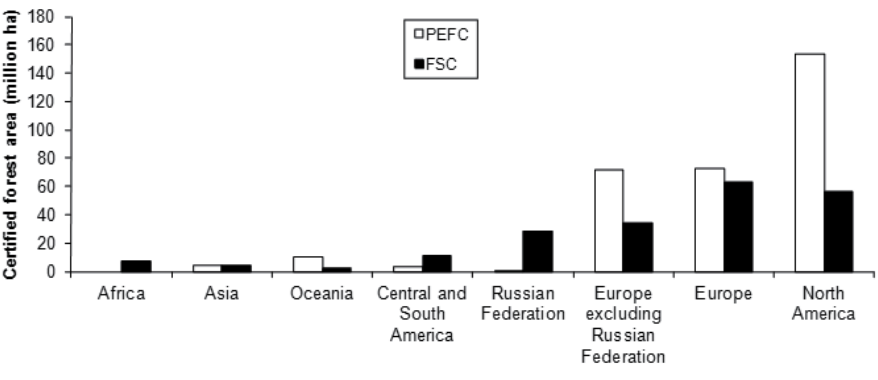

Fig. 2. Global FSC and PEFC certified forest area November 2011 – subregional breakdown (Source: FSC, 2011; PEFC, 2011)

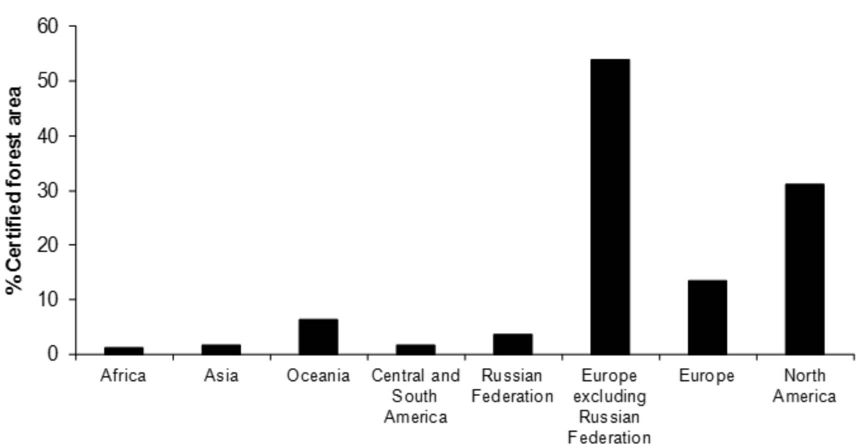

Fig. 3. Percentage of certified forest area, by both FSC and PEFC schemes, November 2011 – subregional breakdown (Source: FSC, 2011; PEFC, 2011)

Forest certification has became very popular, mainly because it is regarded it as a tool whereby everyone should benefit (win-win situation): forest owners should have an exclusive market with premium prices, the forest industry should improve its green corporate image, should not be held responsible for deforestation, and should have available a market tool, consumers should be able to use forest products with a clear conscience, and overall, forests should be managed sustainably.

The concept of forest certification is based on an economic balance, where forest owners and the forest industry place sustainable products on the market in the hope that consumers will be willing to pay the extra cost implied by SFM. Nevertheless, forest certification is still far from reaching its initial goal (win-win), since the expected price increases have not occurred (Cubbage et al., 2010; Gafo et al., 2011). In practice, only consumers and the forest industry have benefited; consumers use certified forest products with a clear conscience, and the forest industry has ensured market access without any great extra cost because this has mainly been assumed by forest owners.

This leads to a difficult question, namely, are forests benefiting from forest certification? It appears logical to believe that forest certification is beneficial to forests, since forest owners must demonstrate that the forests are being managed sustainably. Nevertheless, in depth-analysis reveals a different picture. As already noted, forest certification began in tropical forests with the aim of decreasing deforestation. However, nowadays almost all certified forests are located in developed countries. Furthermore, most of these forests are productive forests, such as single-species and even-aged forests or plantations, in which only small changes must be made to achieve forest certification, while primary forests have largely been ignored. The fact that foresters are able to place certified products from productive forests on the market, with a small additional charge compared to the extra charge involved in certifying products from primary forests hinders certification of the latter, which are actually the most endangered forests. Moreover, this disadvantage may favour unsustainable management, such as illegal logging or in extreme cases conversion of forest land to agricultural land, to favour market competitiveness. Against this background, other initiatives beyond of forest certification has been implemented, such as the FLEGT (Forest Law Enforcement, Governance and Trade) Action Plan of the European Union that provides a number of measures to exclude illegal timber from markets, to improve the supply of legal timber and to increase the demand for wood coming from responsibly managed forests (www.euflegt.efi.int) or the REDD (Reducing Emissions from Deforestation and Forest Degradation) initiative of the United Nations to create a financial value for the carbon stored in forests, offering incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to sustainable development, including the role of conservation, sustainable management of forests and enhancement of forest carbon stocks (www.un-redd.org).

In addition, some environmental organizations now consider that plantations should not be certified, since they consider that plantations are not real forests. Such organizations also denounce the replacement of primary forests with plantations in developing countries (WRM, 2010). Although the replacement of primary forests with plantations is a damaging process, replacement of degraded areas such as abandoned pasture or agricultural land provides obvious advantages from economic and ecological points of view (Brockerhoff et al., 2008; Carnus et al., 2006; Hartley, 2002). The two most important schemes (FSC and PECF) approve the certification of forest plantations because they believe that the promotion of wood products from plantations will help to reduce the pressure on primary forests. The

FSC has added another principle (Principle 10: Plantations) in an attempt to ensure SFM in plantations, while the PECF considers that its criteria and indicators are sufficient to ensure the sustainability of planted forests. The FORSEE project was carried out in order to test the suitability of MCPFE indicators (which are used as the basis for PEFC certification in Europe) for planted forests at a regional level in eight Atlantic regions of Europe (Tomé & Farrell, 2009). This project concluded that with few exceptions, the MCPFE criteria and indicators appear suited to assess the sustainable management of forests, although it was pointed out that they should be considered as a blueprint for true SFM and adaptations are needed at the local level (Martres et al., 2011).

The viability of tropical forest certification will depend on forest owners obtaining premium prices that at least cover the certification costs, taking into account that these costs vary according to the type of forest (primary forest, plantations, etc.) and that consumers' willingness to pay premium prices will also differ. It should be possible for consumers to distinguish the origin of each product, and in other words different labels are required. Nevertheless, the use of different eco-labels is controversial, since many labels may confuse rather than help consumers. Teisl et al (2002) noted that consumers "seem to prefer information presented in a standardized format so that they can compare the environmental features between products" and highlighted "the need for education efforts to both publicize and inform consumers about how to use and interpret the eco-labels". Both of these are difficult tasks when different certifiers are rivals in the market place.

Without standardization and a powerful information campaign, most environmentally concerned consumers will probably demand wood from sustainably managed forests, without taking into account the type of certification label, and will choose the least expensive product (Teisl et al., 2002). This may entail a new associated problem, since producers and industries will probably also choose the bodies that certify forests most readily and at the lowest cost. This may lead to a situation where the certification schemes would tend to compete with each other and standards would be reduced to attract producers, as pointed out by Van Dam (2001).

6. Conclusion

Sustainable forest management is evolving with public awareness and scientific knowledge, and the sustainability concept must be revised to reflect the new reality generated by climate change, where a past reference point shouldn't be considered. Therefore, C&I should be updated continuously to be able to cope with the climate change challenge and assess sustainability of changing ecosystems. Furthermore, harmonization of C&I processes would be the most desirable outcome, since this would improve the credibility of the schemes.

On the other hand, forest certification has failed to avoid deforestation and has got two main challenges;

(1) to certify the forests that are most important in ecological terms and that are most susceptible to poor forest management, such as tropical forests and, to a lesser extent, non productive forest in boreal and temperate regions, and (2) to achieve a market with premium prices, in which the win-win concept will prevail. This will require educational campaigns and a higher level of credibility for labels. Moreover, parallel initiatives, such as FLEG and REDD, considering outside forest sector drivers leading to deforestation should be taking into account to limit this process.

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Section 2

Carbon and Forest Resources

2

The Quality of Detailed Land Cover Maps in Highly Bio-Diverse Areas: Lessons Learned from the Mexican Experience

Stéphane Couturier
Laboratorio de Análisis Geo-Espacial, Instituto de Geografía, UNAM
Mexico

1. Introduction

The production of Land Use and Land Cover (LULC) maps is essential to the understanding of landscape dynamics in space and time. LULC maps are a tool for the measurement of human footprint and social processes in the landscape and for the sustainable use of finite resources on the planet, a growing challenge in our densely populated societies. LULC maps with detailed forest taxonomy constitute a basis for sustainable forest management, especially in highly biodiverse areas.

However, these maps are affected by misclassification errors, partly due to the intrinsic limitations of the satellite imagery used for map production. Misclassification occurs especially when categories of the classification system (classes) are not well distinguished, or ambiguous, in the satellite imagery. Therefore, statistical information on the quality, or accuracy, of these maps is critical because it provides error margins for the derived trends of land cover change, biodiversity loss and deforestation, these parameters being some of the few means that governmental agencies can provide as a guarantee of sustainable forest management practices associated with international conservation agreements.

Assessing the accuracy of LULC maps is a common procedure in geo-science disciplines, as a means, for example, of validating automatic classification methods on a satellite image. For regional scale LULC maps, because of budget constraints and the distribution of many classes over the large extension of the map, the complexity of accuracy assessments is considerably increased. Only relatively recently have comprehensive accuracy assessments, with estimates for each class, been built and applied to regional or continental, detailed LULC maps. However, the quasi totality of the cartography that has been assessed is for countries located in mainly temperate climates with low biodiversity. Instead, LULC maps in highly bio-diverse areas still lack this information, partly because their assessment faces uncertainty due to a high taxonomic diversity and unclear borders between forest classes.

This research focuses on the evaluation of the accuracy of detailed LULC regional maps in highly bio-diverse regions. These are provided by agencies of countries located in the sub-tropical belt, where no such comprehensive assessment has been done at high taxonomic resolution. This cartography is characterized by a greater taxonomic diversity (number of classes) than the cartography in low biodiversity areas. For example, in the United States of Mexico (USM, thereafter 'Mexico'), located in a 'mega-diverse' area, the map of the National

Forest Inventory (NFI) contains 75 LULC classes, including 29 forest cover classes, at the sub-community level of the classification scheme. Taxonomically, the NFI sub-community level in the USM is comparable to the subclass level of the National Vegetation Classification System (NVCS) of the USA, which contains 21 LULC classes, including 3 forest classes.

Higher taxonomic diversity, combined with highly dynamic landscapes, has several implications on the evaluation of errors. First, the numerous sparsely distributed classes represented in the classification scheme pose additional difficulties to the accuracy assessment of the map in terms of representative sampling. Second, thematic conceptual issues impact the way maps should be assessed, because more diversity introduces more physiognomic similarity among taxonomically close classes. As a result, more uncertainty is introduced in each label of the map as well as in each line of the map.

Confronted with these difficulties, this research presents a recently developed accuracy assessment framework, adapted to maps of environments with high biodiversity and highly dynamic landscapes. This framework comprises two methods derived from recent theoretical advances made by the geo-science community, and has been applied recently to the assessment of detailed LULC maps in four distinct eco-geographical zones in Mexico. The first method is a sampling design that efficiently controls the spatial distribution of samples for all classes, including sparsely distributed classes. The second method consists in a fuzzy sets-based design capable of describing uncertainties due to complex landscapes.

This chapter first describes the status of the accuracy assessment of LULC maps, an emerging branch of research in Geographical Information Science. Another section is focused on the methods employed for accuracy assessments of LULC maps and on the challenges related to the taxonomic diversity contained in maps of highly biodiverse areas. The next section focuses on the case of the Mexican detailed LULC cartography, as well as the framework that has been developed recently. Special emphasis lies on the distinctive features which make this case a pioneer experience for taxonomically detailed map assessments as well as a possibly valuable benchmark for other cartographic agencies dealing with biodiversity mapping in other regions of the world. Finally, the accuracy indices found for detailed LULC cartography in Mexico are presented and compared with the accuracy of other assessed international cartography. A major objective of this chapter is to appeal for the inclusion of accuracy assessment practices in the production of cartography for highly bio-diverse areas, because this kind of practice is still nearly absent to date.

2. Quality, or *accuracy* of land cover maps

2.1 Why is it important to measure the quality, or *accuracy* of a map?

A series of important applications typical of the sustainable management of land cover in bio-diverse areas relies on the information content of detailed Land Use/ Land Cover (LULC) maps: forest degradation and regeneration, biodiversity conservation, environmental services, carbon budget studies, etc. In many or all of these applications, map reliability and quality are usually unquestioned, given for granted, just as if each spatial unit on the map perfectly matched the key on the map, which in turn perfectly matched reality. The minimum mapping unit, which defines the scale of the map, is commonly the only information available about the spatial accuracy of a map and no statistically grounded reliability study is applied as a plain step of the cartographic production process.

In general, the comprehensive LULC cartography of a region is obtained through governmental agencies of a country or group of countries, at regional scale, intermediate between local (> 1:50,000) and continental (1:5,000,000). Since the 1990s, the classification of satellite imagery has become the standard for LULC mapping programs at regional scale. However, the classification process is affected by different types of error (Couturier et al., 2009a; Green & Hartley, 2000) related in part to the limited discrimination capacity of the spaceborne remote sensor. The difficult distinction, on the satellite imagery, between categories (or 'thematic classes') of a cartographic legend can cause a high percentage of errors on the map (see next subsection), especially on maps with high taxonomical detail (high number of thematic classes). This is why a forest management policy or a biodiversity monitoring program whose strategy is simply 'process map information and rely on the quality of the map' is highly questionable.

For example in highly bio-diverse regions within Mexico, typical comprehensive database and cartographic products, such as the cartography generated by the National Institute of Statistics, Geography and Informatics (INEGI) and CONAFOR (the National Commission for Forests), are obtained at scale 1:250,000. However all of these products remain deprived of statistical reliability study. This is most unfortunate since the latter governmental agency produces statements on recent deforestation rates based on these maps (online geoportal: CONAFOR, 2008), and these statements, because of the absence of statistical reliability study, remain the focus of distrust and controversial academic and public discussions. It is worth stating that the online availability of the satellite imagery – a feature advertized by this governmental agency - does *not* make an index derived from the imagery more reliable. The extraction of the index based on colour tones of the satellite imagery available online is far from trivial and it is simply impossible for a user to quantitatively derive the global reliability of the cartography out of internet access to the imagery.

An error bar is sometimes present aside the legend of INEGI maps and indicates an estimate of positional errors in the process of map production. However, the procedure leading to this estimate is usually undisclosed, and any objective interpretation of this estimate by the user is thus discouraged (Foody, 2002). Moreover, such error bar indicates a very reduced piece of information with respect to the thematic accuracy of the map.

Instead, the *accuracy* of a cartographic product is a statistically grounded quantity which gives the user a robust estimate of the agreement of the cartography with respect to reality. Such estimate is essential when indices derived from cartography – i.e. spatial extent statistics, deforestation rates, land use change analysis - are released to the public or to intergovernmental environmental panels, while the absence of such estimate indicates that these indices stand without error margins, and as such, without statistical validity. The accuracy of a map also serves as a measurement of the risk undertaken by a decision maker using the map. Besides, this information allows error propagation modeling through a Geographical Information Systems or GIS (Burrough, 1994) in a multi-date forest monitoring task, for example. The construction of the accuracy estimate is generally named 'accuracy assessment' and is explained in section 3.

2.2 Status of the measured accuracy of land cover maps

Assessing the accuracy of LULC maps is a common procedure in geo-science disciplines, as a means, for example, of validating automatic classification methods on a satellite image. For regional scale LULC maps, because of budget constraints and the distribution of many classes over the large extension of the map, the complexity of accuracy assessments is considerably increased. Only relatively recently have comprehensive accuracy assessments, with estimates for each class, been built and applied to regional or continental, detailed

LULC maps. In Europe, Büttner & Maucha (2006) reported the accuracy assessment of 44 mapped classes (including 3 forest classes) of the CORINE Land Cover (CLC) 2000 project. In the United States of America (USA), Laba et al. (2002) assessed the accuracy of 29 LULC classes and Wickham et al. (2004) the accuracy of 21 classes in maps of year 1992 from, respectively, the Gap Analysis Project (GAP) and the National Land Cover Data (NLCD). As a part of the Earth Observation for Sustainable Development (EOSD) program of Canada, Wulder et al. (2006) provide a plan for the future accuracy assessment of the 21 classes in the 2000 Canadian forest cover map, and the accuracy of this program is assessed in the Vancouver Island for 18 classes (Wulder et al., 2007).

These studies reveal the presence of numerous confusions between classes, which yield a global accuracy index (percent area of the map with correct information) of between 38 and 70%. Consequently, these reliability studies constitute very valuable information in terms of the practical use of the assessed maps as well as in terms of enhanced map production strategies in the future.

The cartography of countries situated in areas of high bio-diversity is characterized by a greater taxonomic diversity, i.e. a greater number of classes for a given taxonomic level, than the above cited cartography. However, as is currently the case of the quasi totality of the countries situated in areas of high bio-diversity, the Mexican NFI map, for example, was until recently deprived of statistically grounded information on its reliability. Table 1 reports a collection of 9 studies in the world where a statistically grounded accuracy assessment has been applied to regional LULC cartography. This collection is thought to be relatively representative of existing studies and therefore reflects the status of international accuracy assessments of regional LULC maps to date. The studies which employ a probabilistic sampling design in the sense of Stehman (2001) over the entire area and not just a partial sampling are highlighted in bold. The list of studies was sorted according to the thematic richness of the assessed map (total number of classes).

Some findings can be derived from this table; for example, at first sight, the assessment efforts seem to be greater on the American continent than in other places. The LULC cartography on the African continent is represented by a study with partial assessment in Nigeria; the regional cartography derived from the Africover 2000 project (part of the Global Land Cover, or GLC, project) has not yet been submitted to a probabilistic accuracy assessment to date. In terms of taxonomic diversity (number of mapped classes), the 2000 NFI map in Mexico ranks second after the Southwest USA map, and ranks first of the mega-diverse areas. Therefore, the study on the 2000 NFI map in Mexico stands out as especially important in the world. Among the probabilistic assessments, the study assesses the highest number of classes (32 assessed classes vs 22 in Europe which holds the second ranking). For comparison purposes, we indicated the equivalent taxonomic level of each map, with respect to the four aggregation levels (*biome*, *type*, *community*, *community with alteration*, also known as *sub-community*) considered for the classification system of the IFN 2000 (Palacio-Prieto et al., 2000), plus two more detailed levels (*community with density grades* and *association with alteration*). The taxonomic level of the maps is generally relevant to applications of regional forest management and biodiversity monitoring (7 studies involve maps of levels *community*, *community with alteration*, *community with density grades*, *association with alteration*, which are the most detailed taxonomic levels). However, the study on the NFI 2000 map is the only accuracy assessment *per class* of these levels of taxonomic detail in a mega-diverse area (the other detailed assessments are in the USA, Europe and Canada), a level of detail which actually allows statistically- based cartographic management schemes in terms of bio-diversity dynamics. Another noteworthy study in a mega-diverse area is the one in South and Southeast Asia (Stibig et al., 2007), but its accuracy assessment was only obtained at the biome level. A study at the biome level does allow a deforestation study (forest – non forest change) with error margins, but does not allow a land cover change study with more detailed processes (e.g. 'forest to forest with alteration'), also important in sustainable management.

However, the assessment of the NFI 2000 cartography in four eco-geographical areas only constitutes a pilot study, confined to a limited extension, in a mega-diverse area. The spatial extent subject to the assessment is about 19,500 km^2, much smaller than the majority of the other studies (seven of the nine studies involve extents of more than one million km^2). Indeed, the enhanced taxonomic diversity, combined with highly dynamic landscapes, increase the difficulty of the accuracy assessment of maps in mega-diverse areas (Couturier et al., 2007), a fact that probably contributes to explain the lack of studies in such areas.

3. How can I measure the quality, or *accuracy* of land cover maps in biodiverse areas?

Generally, map accuracy is measured by means of reference sites and a classification process more reliable than the one used to generate the map itself. The classified reference sites are then confronted with the map, assuming that the reference site is "the truth". Agreement or disagreement is recorded in error matrices, or confusion matrices (Card, 1982), on the base of which various reliability indices may be derived. For regional scale LULC maps, the abundance and distribution of classes over the large extension of the map, confronted with tight budget constraints, add complexity to accuracy assessments. Only relatively recently have comprehensive accuracy assessments, with estimates for each class, been built and applied to regional or continental LULC maps (e.g. Laba et al., 2002; Stehman et al., 2003; Wickham et al., 2004; Wulder et al., 2007). Because of the high complexity of these products, detailed information on the assessment process itself is needed for the reliability figures to be interpreted properly (Foody, 2002). With this understanding, Stehman & Czaplewski (1998) have proposed a standard structure for accuracy assessment designs, divided into three phases:

1. Representative selection of reference sites (sampling design),
2. Definition, processing and classification of the selected reference sites (verification design),
3. Comparison of the map label with the reference label (synthesis of the evaluation).

Wulder et al. (2006) provide a review on issues related to these three steps of an accuracy assessment design for regional scale LULC cartography. We indicate in the next sub-section the features and techniques most commonly employed in the literature for phases 1-3.

3.1 Methods employed in the accuracy assessment of LULC maps in the world

3.1.1 Sampling design

The first phase of the accuracy assessment is the sampling design. The selection of the reference sites is a statistical sampling issue (Cochran, 1977), where strategies have varied according to the application and complexity of the spatial distribution. Stehman (2001) defines the probability sampling, where each piece of mapped surface is guaranteed a non-null probability of inclusion in the sample, as being a basic condition for statistical validity. In most local scale applications, reference sites are selected through simple random

Region of the world	Acronym of project and year of cartography	Prevailing biotic environment	Assessment design	Number of classes		Equivalent taxonomic level	Spatial resolution and satellite sensor	Spatial extent of the map effectively assessed	Reference publication
				Total	Forest				
Southwest USA	GAP 2000	Temperate-dry	Partial (near to roads)	125 (85 assessed)	27 (18 assessed)	Association with alteration	30m (Landsat TM)	? (*1.4 M km²*)	Lowry et al. (2007)
Mexico (4 areas)	**NFI 2000**	**Mega-diverse**	**Probabilistic**	**75 (32 assessed)**	**29 (19 assessed)**	**Community with alteration**	**1km (Landsat TM)**	**0.0195 M km² (1.95 M km²)**	**Couturier et al. (2010)**
European Union	**CorineLC 2000**	**Temperate**	**Probabilistic**	**44 (22 assessed)**	**3**	**Community**	**30m (Landsat TM)**	**2.68 M km²**	**Buttner and Maucha (2006)**
South and Southeast Asia	**TREES 2000**	**Mega- diverse**	**Probabilistic for *biome level***	**40 (= 4 *biome* classes assessed)**	**17 (only 'forest' class assessed)**	**Community with alteration**	**1km (SPOT-VEGETATION)**	**4.5 M km²**	**Stibig et al. (2007)**
India	ISRO-GBP 1999	Mega- diverse	Partial (in 3 states of the country)	35	14	Community	188m (WiFS, IRS)	? (*3.3 M km²*)	Joshi et al. (2006)
USA	**NLCD 1992**	**Temperate**	**Probabilistic**	**21**	**3**	**Community**	**30m (Landsat TM)**	**9.1 M km²**	**Stehman et al. (2003)**
Canada (1 area)	**EOSD- Forest 2000**	**Temperate**	**Probabilistic**	**18**	**10**	**Community with density**	**25m (Landsat TM)**	**0.031 km²**	**Wulder et al., (2007)**
Nigeria	1990	Tropical humid	Partial	8	3	Biome	1km (NOAA AVHRR)	? (*0.904 km²*)	Rogers et al. (1997)
Legal Amazon, Brasil	GLC 2000	Tropical humid and dry	Partial	5	3	Type	1km (SPOT-VEGETATION)	*? (5.0 M km²)*	Carreiras et al. (2006)

USA: United States of America.
Prevailing biotic environment: If large areas of different environments exist, e.g. temperate and sub-tropical, dry and/or humid, we indicated 'mega-diverse'.
Assessment design: 'probabilistic' if the design is associated with the total area mapped, and 'partial' if not; the design is probabilistic according to criteria of statistical rigor established by Stehman (2001).
Equivalent taxonomic level: equivalence in terms of the classification system of the National Forest Inventory 2000 in Mexico. Taxonomic levels are *Biome ('Formación' in Spanish)*, *Type*, *Community*, *Community with alteration* (or *sub-community)*, sorted from the most general to the most detailed (Palacio-Prieto et al.: 2000). Two additional more detailed levels were considered: *Community with density* (vegetation density levels) and *Association with alteration*.
Spatial extent of the map effectively assessed: M km2 = millions of square kilometers. In case of partial assessment designs, many authors do not report sufficient information that indicate the effective area actually assessed; if this is the case we indicate '?', and the figure between parenthesis represents the total extent of the map (not the one actually assessed).

Table 1. List of major published studies on assessments of regional land use/ land cover maps in the world. The list is relatively exhaustive of institutional programs which aim a probabilistic sampling design. The studies are sorted according to the total number of classes contained in the legend of the map.

sampling. Two stage (or double) random sampling has been preferred in many studies in the case of regional cartography; in a first step, a set of clusters is selected through, for example, simple random sampling. This technique permits much more control over the spatial dispersion of the sample, which means much reduction of costs (Zhu et al., 2000), and was adopted for the first regional accuracy assessments in the USA, for LULC maps of 1992 (Laba et al., 2002; Stehman et al., 2003).

A random, stratified by class sampling strategy means that reference sites are sampled separately for each mapped class (Congalton, 1988). This strategy is useful if some classes are sparsely represented on the map and, therefore, difficult to sample with simple random sampling. This strategy was adopted at the second stage of their double sampling by Stehman et al. (2003) and Wickham et al. (2004).

Systematic sampling refers to the sampling of a partial portion of the mapped territory, where the portion has been designed as sufficiently representative of the total territory. This strategy, adopted as a first stratification step, is attractive for small scale datasets and reference material of difficult access: Wulder et al. (2006) define a systematic stratum for the future (and first) national scale accuracy assessment of the forest cover map in Canada.

3.1.2 Verification design

For regional scale detailed land cover maps, the frame for reference material of phase 2 is typically an aerial photographic coverage (e.g. Zhu et al., 2000), and ground survey is only occasional. For all studies cited in the text of section 3 so far, the classification of reference sites was based on more precise imagery i.e. imagery with higher spatial resolution, than the imagery that was employed during the map production process. In these cases the map was produced using Landsat imagery (spatial resolution of 30m), and was assessed using aerial photographs (spatial resolution better than 3m) or aerial videography (Wulder et al., 2007). The map of South and Southeast Asia (Stibig et al., 2007, table 1) was produced using the SPOT-VEGETATION sensor (spatial resolution of 1km) and assessed using Landsat imagery (resolution 30m). An alternative reference material for recent LULC cartography could be a wide coverage of very high resolution satellite imagery such as the one available on the online Google Earth database. For all studies, remote sensing based reference data has been preferred as the primary material instead of ground survey for its cost-effectiveness in large areas.

Double sampling techniques are effective at controlling the spatial dispersion of the sample among image/ photograph frames if these are taken as the cluster, or Primary Sampling Unit (PSU), for first stage sampling (see previous subsection).

Congalton & Green (1993) relate errors of the map to imprecise delineation and/or misclassification. Additionally, the imperfect process of the assessment itself also generates erroneous statements on whether the map represents reality or not. A main topic is the positional error of the aerial photograph with respect to the map. To this respect, a procedure ensuring geometric consistency must be included in the evaluation protocol. For example, the procedure of visually locating sample points on the original satellite imagery, described in Zhu et al. (2000), reduces the inclusion of errors due to geometric inconsistencies. Other sources of fictitious errors occur in phase 3 (labeling protocol), and are related to the thematic and positional uncertainties of maps. This topic is introduced in section 3.2 and fully devised in Couturier et al. (2009a).

3.1.3 Synthesis of the evaluation

The comparison between the information contained on the map and the information derived from the reference site yields an agreement or a disagreement. Typically, the numbers of agreements and disagreements are recorded and form a confusion matrix. However, these numbers are reported in the matrix with weights that depend on the probability of inclusion of the reference site in the sample (Stehman, 2001). This probability of inclusion is defined by the sampling design. For example, a simple random selection is associated with a uniform (constant) inclusion probability among all reference sites. For a two stage sampling, the probability of inclusion follows Bayes law: The probability of inclusion p_{2k} of a reference site at the second stage is a multiplicative function of the inclusion probability p_{1k} of the cluster it pertains to, and of the inclusion probability of the reference site, once the cluster has been selected $p_{2|1}$ (conditional inclusion probability)(equation 1):

$$p_{2k}=p_{2|1} * p_{1k} \tag{1}$$

Accuracy indices per class are derived from these calculations: 'user's accuracy' of class k is the account of agreements from all sites of mapped class k while the 'producer's accuracy' of class k counts agreements from all reference sites labeled as class k. The respective disagreements correspond to 'commission errors' and 'omission errors' (Aronoff, 1982). The global accuracy index, or proportion correct index, which indicates the accuracy of the map as a whole (all thematic classes), integrates the accuracy level of all classes, weighted by the probability of inclusion specific to each class. In this calculation, weights usually correspond to the relative abundance of the class on the map. Other reliability indices are popular, such as the Kappa index, which takes into account the contribution of chance in the accuracy (Rosenfíeld and Fitzpatrick-Lins, 1986). However, in regional scale accuracy assessments, the proportion correct indices are preferred, because they are coherent with the interpretation of confusions according to area fractions of the map (Stehman, 2001).

A confidence interval of the accuracy indices can be estimated, although only few accuracy assessments provide this information. A popular estimate of the confidence interval is based on the binomial distribution theory: the confidence interval of the accuracy estimate depends on the sample size and on the reliability value (accuracy estimate) in the following manner (Snedecor & Cochran, 1967, cited by Fitzpatrick-Lins, 1981)(equation 2):

$$d^2 = t^2 \, p \, (1-p) \, / \, n \tag{2}$$

where d is the standard deviation (or half the confidence interval) of the estimate, t is the standard deviate on the Gaussian curve (for example, t = 1.96 for a two-sided probability of error of 0.05), p is the reliability value, and n is the number of sampled points. Although most accuracy assessments refer to it, this binomial distribution formula is only valid for simple random sampling. For more sophisticated sampling designs (e.g. two stage sampling) the confidence interval is influenced by the variance of agreements among clusters. Estimators integrating inter-cluster variance (Stehman et al., 2003) are seldom employed in map accuracy assessments because of their complexity (Stehman et al., 2003). For the cartography assessment in Mexico, an estimator which includes an inter-cluster variance term was used in Couturier et al. (2009a). The estimate was built on a stratified by class selection in the second-stage of the sampling design (Särndal et al., 1992).

3.2 Methodological challenge for the accuracy assessment of detailed LULC maps

The detailed cartography of highly bio-diverse regions is characterized by a greater taxonomic diversity (number of classes) than the cartography of regions in mainly temperate climates. Greater taxonomic diversity, combined with highly dynamic landscapes, has several implications on the evaluation of errors.

First, the numerous sparsely distributed classes represented in the classification scheme pose additional difficulties to the accuracy assessment of the map in terms of representative sampling.

Second, thematic conceptual issues impact the way maps should be assessed, for reasons illustrated in three cases:

- More diversity introduces more physiognomic similarity among taxonomically close classes: for example, cedar forest is an additional conifer forest class in sub-tropical environments, so mixed conifer forest patches are more difficult to classify, and boundaries between conifer forests are more difficult to set. As a result, more uncertainty is introduced in each label of the map as well as in each line of the map.
- Highly dynamic landscapes mean more classes placed along a continuum of vegetation, where some classes are a temporal transition to other classes. For example, the sequence of classes 'pasture to secondary forest to primary forest' is characteristic of sub-tropical landscapes. The extremes of such sequence may be spectrally distinct and easily separated, however boundaries between intermediate classes are difficult to interpret.
- More diversity combined with highly dynamic landscapes means more fragmented landscapes composed of small patches of different classes. The interpretation of these results in heterogeneous patches is difficult to assess.

Third and last, ambiguity between classes on satellite imagery, related to the above situations, becomes more likely. In these conditions, the information on spectral separability could be a systematic tool to prioritise future cartographic efforts (Couturier et al., 2009b).

Confronted with the three implications, we developed two methods based on recent theoretical advances made by the geo-science community.

The first method comprised a sampling design that efficiently controlled the spatial distribution of samples for all classes, including sparsely distributed (or fragmented) classes. Previous assessments have relied on two-stage sampling schemes where simple random or stratified by class random sampling was employed in the first stage. Couturier et al. (2007) demonstrated that these strategies fail in the context of the Mexican NFI. Section 3.2.1 presents a two-stage hybrid scheme where proportional stratified sampling is employed for sparsely distributed (rare) classes. This scheme was applied to four areas in distinct eco-geographical zones of Mexico (see section 4.3).

The second method was to design a fuzzy sets-based design capable of describing uncertainties due to complex landscapes. We will see in section 3.2.2 that it is traditionally possible to incorporate a thematic fuzzy component in accuracy assessment designs, but this component, as well as positional uncertainty, are implicitly fixed by the map producer, with no possible change after the design has been applied. Recently, advances in fuzzy classification theory have permitted the comparison of maps incorporating thematic and positional uncertainties.

3.2.1 The sampling design for fragmented (rare) classes

In order to find a sampling design well suited to an abundant set of fragmented, sparsely distributed (or rare) classes, several double sampling designs (DS) were previously tested in a pilot study, the closed watershed of the Cuitzeo lake in Mexico (Couturier et al., 2007);

DS1 was defined as the simple random selection of the Primary Sampling Units (PSUs), as in Laba et al. (2002).
DS2 was characterized by the random, stratified by class, selection of PSUs, as in Stehman et al. (2003).
DS3 was defined as a proportional, stratified by class, selection of PSUs. For the latter design, not applied in previously published research, the probability of inclusion of a PSU is proportional to the abundance of the class in the PSU. The abundance of a class equates its area fraction, easily obtainable via attribute computation in a GIS. Then, the probability of inclusion of Secondary Sampling Units (SSUs) at the second stage was defined as being inversely proportional to the abundance of the class in the PSU. Proportional sampling is a known statistical technique (Cochran, 1977) and some characteristics of its application to map accuracy assessment are devised in Stehman et al. (2000). However, DS3 had never been applied in published studies, maybe because it was not necessary for maps with classification systems of mainly temperate countries.
Finally, an entirely novel, hybrid design (DS4) includes a simple random selection of PSUs (as in DS1) for common classes (area fraction above 5%, 7 classes in Cuitzeo), and a proportional stratified selection of PSUs (as in DS3) for rare classes (area fraction below 5%, 14 classes in Cuitzeo). After selection of the PSUs, the sample size of SSUs was fixed at 100 per mapped class, a value widely adopted in similar assessments (Stehman & Czaplewski, 1998).
With fixed operational costs, the only design that systematically provided statistically representative estimates for all classes was the hybrid design DS4 (Couturier et al., 2007). Additionally, the hybrid design achieved a spatial dispersion of the sample similar to the dispersion achieved by DS1, with simple random selection of Primary Sampling Units (PSUs). DS1 is known for generating a good dispersion of the sample in regional map assessments. For this reason, DS1 was successfully applied in the accuracy assessment of the NLCD project in the USA (Stehman et al., 2003). However, DS1 was discarded in our pilot study because it was not able to handle the high number of rare classes of the NFI. Instead, the hybrid design maintains simple random selection of PSUs for common classes, but applies a proportional stratified selection of PSUs for rare classes. This way, DS4 cumulates the advantages of a wide-spread sample dispersion for common classes, and the advantages of a sufficient sample size and easy estimate calculation for rare classes.

3.2.2 The fuzzy approach for positional and thematic uncertainties

In traditional accuracy assessment, the labeling protocol (phase 3 of the accuracy assessment) consists in attributing one and only one category of the classification scheme to each reference site. However, this procedure assumes that each area in the map can be unambiguously assigned to a single category of the classification scheme (or LULC class). In reality, the mapped area may be related to more than one LULC class because of the characteristics of the landscape in the reference site. This conceptual difficulty is ignored in the traditional (or Boolean) labeling protocol, and may conduce to an under-estimation of map accuracy (Foody, 2002). In particular, this difficulty arises in the following cases:

- The landscape in the reference site has physiognomic similarities with more than one LULC class. For example, a one hectare forest patch containing oak trees and two or three pine trees has physiognomic similarity with forest class 'oak forest' and forest class 'oak-pine forest'. As a result, the map label for this site is affected with uncertainty. The reference site could be in a transition zone between an oak forest and a oak-pine forest.

- The landscape in the reference site is a patch within a continuum of vegetation, where the LULC classes represented are a temporal transition to other classes. For example, the sequence of classes 'pasture to secondary forest to primary forest' is characteristic of some sub-tropical landscapes. As a result, the map label for this site is affected with uncertainty. The extremes of such sequence may be easily identified on the ground, however boundaries between patches of intermediate classes are difficult to set. As a result, lines between mapped objects for this site are affected with uncertainty.
- The landscape in the reference site is a fragmented landscape, composed of small patches (below minimum mapping unit) of different land use or land cover. The interpretation of this mixed reference site, because of the scale of the map, must be a non unique label. As a result, the map label for this site is affected with uncertainty.

Due to the above described continuous or fragmented aspects of land use and land cover in a landscape, maps with discrete representation (discrete, or crisp, class assignation) and infinitely small line features (crisp boundaries of objects) necessarily describe reality with a certain margin of uncertainty. In order to take this uncertainty aspect into account, it has been referred to the concept of fuzzy sets (Zadeh, 1965).

In the crisp approach, an element x of the map X belongs totally to a class k of the set C or does not belong to it. A way of representing this is to define a membership function μ, which takes the value '1' if the element x belongs to class k and '0' otherwise. This assignation process can be called Boolean labeling. In a typical case of photo-interpretation for map accuracy assessment, a forest reference site with a crown cover close to 40% may pertain to a transition zone between closed forest (crown cover > 40%) and open forest (crown cover < 40%). If the photo-interpreter characterizes this site as closed forest and the corresponding label on the map is open forest, then this site is interpreted as an error on the map.

In fuzzy sets theory, an element belongs to a set or class with a certain degree of similarity, probability or property, some of these notions being contained in a 'degree of membership', depending on the application. One element x may belong to various classes at a time with different degrees of membership $\mu_k(x)$. For example, quantitative degrees of membership take a value between 0 and 1 to express the partial membership to various classes of the set. With this approach, the reference site with a tree cover close to 40%, would be characterized for instance by a 0.5 degree of membership in both classes (open and closed forest).

Many authors have rejected the term "fuzzy set theory" to characterize landscape interpretation, in favor of "soft" or "continuous" classification. Critiques have noted that the use of a continuous range of membership values does not entail employment of the concepts of fuzzy logic (Haack, 1996). Nevertheless, the term "fuzzy classification" will be used here as a compromise, recognizing the heritage of these techniques but emphasizing the classification process over the logic of set theory.

Cartographical models that present a fuzzy classification approach were developed (Equihua, 1990, 1991; Fisher & Pathirana, 1990; Foody, 1992; Wang, 1990). These models allow the representation of the landscape features previously enumerated in this subsection. Despite the perspective of a more lawful representation of real landscapes, these models present two limitations:

- The interpretation and manipulation of fuzzy classified maps by users already accustomed to crisp maps is still a pending challenge; each point on the map represents various LULC classes with different degrees of membership. The vast majority of maps, including the existing LULC maps in Mexico and in territories with high biodiversity, are crisp.

- The coherent production of fuzzy classified maps with quantitative degrees of membership is not possible in all mapping situations. One of the situations where such fuzzy classified map can be easily produced is a binary map of, for example, forest/non-forest where percent crown coverage represents one of the fuzzy labels. A second situation is a map made of ordinal categories, where uncertainty between categories can be modeled by a fuzzy matrix (illustrated in Hagen, 2003). A third situation occurs when automatic processing is constructed so as to generate the quantitative fuzzy labels. A typical example of this third situation is a map of unmixed fractions of LULC classes, extracted from automatic spectral mixing analysis, where the classes are represented by pure end-member pixels. However, the assignment of quantitative fuzzy labels during visual interpretation, for example, can be affected by subjectivity. This is possibly a reason why quantitative fuzzy labeling has generally not been adopted in mapping situations with visually interpreted material.

Consequently, for the challenge concerning land cover over highly bio-diverse regions, the focus was made on assessing a crisp map with fuzzy classified reference material. As mentioned in section 3.1, the typical reference material of regional accuracy assessments is aerial photographs. We were confronted with the subjectivity of interpreters in preliminary attempts at classifying the material with quantitative degrees of membership. For these reasons, we settled for the fuzzy classification technique expressed by linguistic rules, introduced for visual interpretation by Gopal & Woodcock (1994), and commonly employed. This technique of fuzzy classification is described in the verification design of Couturier et al. (2008) for the case of detailed land cover map assessment in Mexico.

The use of fuzzy classification techniques in the labeling protocol permits the reduction of fictitious errors in the process of map assessment, fictitious errors being due to the thematic uncertainty of maps. However, as said earlier, maps are also characterized by positional uncertainty. This uncertainty may also affect the accuracy results when the assessed map is compared with the reference material. As a result of advances in fuzzy classification theory, much research have focused on the comparison of fuzzy classified maps and on the multi-scale comparison of maps (Pontius & Cheuk, 2006; Remmel & Csillag, 2006; Visser & de Nijs, 2006). In Couturier et al. (2009a), the systematic inclusion of positional uncertainty within regional accuracy assessments is proposed, formalized, and applied to the case of land cover maps of highly bio-diverse regions.

4. Mexican detailed land cover cartography and the application of the methods developed recently

4.1 Mexican detailed LULC cartography

As a consequence of its extension over a wide range of physio-graphical, geological and climatic conditions, the Mexican territory is composed of a remarkably large variety of ecosystems and diversity of flora (Rzedowski, 1978), is among the five richest countries in biological diversity and therefore considered as a mega-diverse area (Velázquez et al., 2001). In turn, this range of environmental conditions predetermined transformations of the landscape by humans in a variety of ways. The intensification of land uses over the last century and the response of the eco-systems to this intensification altogether shaped the complex landscapes in the contemporary Mexico.

In the past three decades, governmental agencies in the North American sub-continent have promoted the production of geographic information at a regional scale, which we define intermediate between continental (1:5 000 000) and local (> 1:50 000). The major historical data set of regional scale (1:250 000) LULC maps in Mexico was developed by the National Institute of Statistics, Geography and Informatics (INEGI). In the nineteen eighties, the first set of 121 LULC maps was published for the entire territory, based on the interpretation of aerial photography collected from 1968 to 1986 (average date 1976) and considerable ground work (INEGI, 1980). This dataset was part of the INEGI first series ('INEGI serie I', in Spanish) cartography. In the mid nineteen nineties, INEGI produced the second series cartography ('INEGI serie II') in a digital and printed format. The LULC maps of INEGI serie II were elaborated using the former series I maps, and visual interpretation of Landsat Thematic Mapper (TM) images acquired in 1993, printed at scale 1:250 000. The INEGI cartography legend included 642 categories to consistently describe LULC in the entire country. For land cover categories, or classes, this detailed classification scheme was based on physiognomic, floristic and phenological attributes of plant communities (table 1) and degrees of anthropic modification.

Formation	Vegetation Types
Temperate Forest	1. Cedar forest , 2. Fir forest, 3. Pine forest, 4. Conifer scrubland, 5. Douglas fir forest, 6. Pine-oak woodland, 7. Pine-oak forest , 8. Oak-pine forest, 9. Oak forest, 10. Mountain cloud forest, 11. Gallery forest.
Tropical forest	*Perennial & sub-perennial tropical forests:* 12. Tropical evergreen forest, 13. Tropical sub-evergreen forest, 14. Tropical evergreen forest (medium height), 15. Tropical sub-evergreen forest (medium height), 16. Tropical evergreen forest (low height), 17. Tropical sub-evergreen forest (low height) , 18. Gallery forest.
	Deciduous & sub-deciduous forests: 19. Tropical sub-deciduous forest (medium height), 20. Tropical deciduous forest (medium height), 21. Tropical sub-deciduous forest (low height), 22. Tropical deciduous forest (low height), 23. Tropical spiny forest.
Scrubland	24. Sub-montane scrubland, 25. Spiny Tamaulipecan scrubland, 26. Cacti-dominated scrubland 27. Succulent-dominated scrubland, 28. Succulent-cacti-dominated scrubland, 29. Sub-tropical scrubland, 30. Chaparral, 31. Xerophytic scrubland, 32. Succulent-cactus-dominated cloud scrubland,, 33. Rosetophilous scrubland, 34. Desertic xerophytic rosetophilous scrubland, 35. Desertic xerophytic microphilous scrubland,, 36 Propospis spp.-dominated, 37. Acacia spp.-dominated, 38. Vegetation of sandy desertic conditions.
Grassland	39. Natural grassland, 40. Grassland-huizachal, 41. Halophilous grassland, 42. Savannah, 43. Alpine bunchgrassland, 44. Gypsophilous grassland.
Hygrophilous vegetation	45. Mangrove, 46. Popal-Tular (hygrophilous grassland), 47. Riparian vegetation.
Other vegetation Types	48. Coastal dune vegetation, 49. Halophilous vegetation.

Table 2. Classification scheme of the INEGI land use and vegetation cartography (only natural land cover categories are indicated):

In the year 2000, the Ministry of the Environment in Mexico (SEMARNAP) attributed the task of updating the LULC map of the country (at scale 1:250 000) to the Institute of Geography of the Universidad Nacional Autónoma de México (UNAM). This task was intended as an academic-driven methodological proposal for rapid nation-wide detailed forest assessments. In this perspective, the cartographic project was named the National Forest Inventory (NFI) map of year 2000. An important objective of the project was the compatibility with previous cartography in view of LULC change studies. Rapidity (8 months) and low cost of execution were constraints that guided the planning of the project.

Visual interpretation of satellite imagery, with the aid of INEGI previous LULC digital cartography, was selected as the best classification strategy. However, the classification scheme was adjusted to the capacity of the Landsat Enhanced TM plus (ETM+) imagery at discriminating classes, according to previous classification experience in Mexico (e.g. Mas & Ramírez, 1996). The 642 categories of the INEGI cartography legend (including 49 vegetation types in table 2) were aggregated into 75 thematic classes (community level, with the inclusion of two levels of human induced modification) and further into three coarser levels of aggregation.

Visual interpretation was done on ETM+ imagery of the drier season, acquired between November 1999 and April 2000. The best option for interpretation was visually selected among various colour composites. The methods and results of the IFN 2000 cartographic project have been published (Mas et al., 2002a; Palacio-Prieto et al., 2000; Velázquez et al., 2002). Figure 1, taken from Mas et al. (2002a), illustrates the 2000 NFI map at formation level (coarsest level of aggregation). The present research focuses on the cartographic product with the finest level of aggregation (community level, with the inclusion of degradation levels), because of the availability of abundant quasi synchronous aerial photograph cover all throughout the country which can be used as independent reference data for accuracy assessment.

Since 2001, the National Commission of Forests (CONAFOR), an agency dependent of the National Environmental Agency in Mexico (SEMARNAT), is in charge of updating the vegetation cover change in Mexico, in parallel with the INEGI regional LULC cartography (year 2002: 'Serie III' map, and year 2007: 'Serie IV' map). None of this cartography so far has been generated with an international standard accuracy assessment scheme as described in this chapter. Since 2004, CONAFOR has established a 5 year repeat forest inventory of the Mexican territory ('Inventario Nacional Forestal y de Suelos', INFyS, 2008), based on a systematic grid of ground plots over the entire vegetation cover of Mexico.

4.2 Developing the framework for assessing the Mexican detailed LULC cartography

As stated previously, if we except the material presented in this chapter, all detailed LULC cartography in Mexico is characterized by the absence of quantitative, reliable information on its quality. Consequently, only qualitative statements can characterize the reliability of archive and recent Mexican cartographic products, based on a judgment on the quality of the data that was employed in the map production process. For example, the INEGI serie I data (1968-1986) are expected to be very reliable in terms of thematic accuracy, because of the quality of the field reference data, but their temporal coherence (accuracy) is low. Conversely, the LULC maps of INEGI serie II are characterized by a high temporal coherence. However, because the visual interpretation of only one colour composite of

Landsat imagery (bands 4,3,2) was used to update a map with very high taxonomic precision (INEGI legend of 642 classes), the thematic accuracy of INEGI serie II is likely to be poorer than that of INEGI serie I (Mas et al., 2002b).

In the case of the National Forest Inventory (NFI) map of Mexico, a preliminary accuracy assessment was conducted immediately after map production in year 2000. A systematic sampling of the entire country was planned, but the assessment could only take place on a small portion of the planned coverage, in the Northern part of the country (Mas et al., 2002a). The assessment yielded reliability levels for a few homogeneously distributed classes, and was not designed to attend, in a cost-effective way, the high number of classes of the NFI map and their complex distribution over the entire territory.

In 2003, a research project was initiated at the Institute of Geography, UNAM, with the proposed tasks of building academic capacity for the assessment of LULC maps in Mexico and developing a framework for future accuracy assessments of the INEGI cartography. Such a framework was built in accordance with the typically available verification materials, skills and resources in Mexico. In order to implement the methodology, a pilot study was launched over a set of four distinct eco-geographical areas described in the following section. The accuracy assessment fulfilled the following desirable criteria (see section 3): 1) a probability sampling scheme (sensu Stehman & Czaplewski 1998), comprising a sampling design, a response design and the synthesis of evaluation; 2) an operational design for future INEGI map updating missions; 3) a reasonable compromise between the precision (standard error) of accuracy estimates and operational costs.

4.3 An accuracy assessment in four eco-geographical areas in Mexico

We fixed a set of eco-geographical areas (located on figure 2) that captured parts of the mega-diversity of the Mexican territory, with special focus on some of the main forest biomes (see Tables 2 and 3). They also included contrasted levels of modification of the original vegetation cover.

Two areas are located on the transversal volcanic chain and contiguous altiplano in central western Mexico. These are the closed watershed of the Cuitzeo Lake, later referred as Cuitzeo, and an area encompassing both the natural reserve of the Tancítaro peak and the Uruapan avocado production zone, later referred as Tancítaro. Both areas are included in the state of Michoacán and are covered with temperate sub-humid and tropical dry vegetation (Table 3). A third area includes the core and buffer zones of the biosphere reserve of Los Tuxtlas, in the state of Veracruz. This area is mainly characterised by tropical humid conditions although temperate humid micro-climates prevail on the relief of the two coastal volcanic chains. The fourth area corresponds to the Mexican side of the Candelaria river watershed in the state of Campeche. This area includes a portion of the Calakmul forest reserve and is mainly characterized by tropical sub-humid conditions.

The Candelaria and Tancítaro areas comprise extensive forests (of low and high levels of human management, respectively) while most of Cuitzeo and Los Tuxtlas is covered with non forested agriculture land (crop and grazing land, respectively). Apart from the informative contrast in LULC, the selection and definition of these areas were guided by the availability of reference data for independently verifying the NFI-2000 map. These reference data are detailed in table 4.

Within each eco-geographical region (stratum), the sampling design incorporated a two-stage sampling design where aerial photographic frames formed the Primary Sampling Units (PSUs), as in most regional accuracy assessments of Landsat-based maps (Wulder et al. 2006). A regular 500 m-spaced two dimensional grid (hereafter referred to as the 'second stage grid') formed the set of points, or Secondary Sampling Units (SSUs) of the second stage. Indeed, a scale criterion used during map production was to leave out polygons less than 500 meters wide.

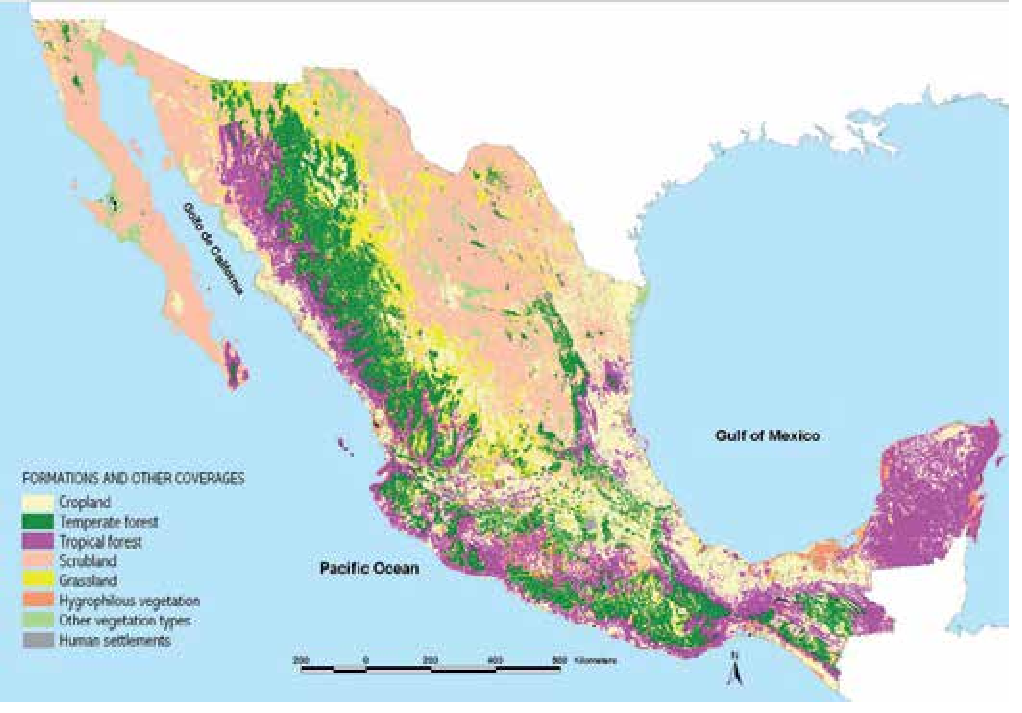
The first stage of the sampling design consisted in the selection of two subsets of PSUs. The first subset of PSUs was obtained with a simple random selection and was used for the assessment of common classes (classes whose area fraction is above 5%, a total of 7 classes in Cuitzeo, for example). The second subset of PSUs was obtained with a proportional random selection of PSUs, and was used for the assessment of rare classes (classes whose area fraction is below 5%, a total of 14 classes in Cuitzeo, for example). In the latter selection, the probability of selection attributed to each PSU was proportional to the abundance of the rare class in that PSU, as described in Stehman et al. (2000, further discussed via personal communication); this mode of selection was retained as an appropriate way for including all scarcely distributed (or 'rare') classes (a frequent occurrence in our case), in the sample while maintaining a low complexity level of statistics (i.e. standard stratified random formulae to compute estimators of accuracy). As a compromise between the precision of the estimates and our budget for undertaking the pilot research, the number of selected PSUs approached but was maintained below one quarter of the total number of PSUs in each area. According to this scheme, the PSU selection process is made independently for each rare class and a given PSU can be potentially selected multiple times (for rare classes as well as for the common classes). This hybrid selection scheme, differentiated according to 'common' and 'rare' classes, was proposed and detailed in Couturier et al. (2007), where its potential advantages with respect to sampling designs formerly applied in the literature were evaluated.

Once the sample PSUs were selected, all points of the second stage grid included within these PSUs were assigned the attribute of their mapped land cover class. The full second stage sample consisted of the selection of 100 points (SSUs) for each class mapped in the area. For each common class, the selection was a simple random sorting of points within the second stage grid in the first subset of PSUs. For rare classes, the selection of points was obtained via proportional random sampling in the second subset of PSUs, this time with a probability inversely proportional to the abundance of the class. This mode of selection could preserve equal inclusion probabilities at the second stage within a rare class (see the option of proportional stratified random sampling advocated in Stehman et al. 2000). A sequence of ArcView and Excel-based Visual Basic simple routines, for easy and fast repeated use on vector attributes of each class, was specifically designed to perform this proportional selection at both stages.

5. Quality of detailed LULC cartography in Mexico vs. quality of international cartography

5.1 Accuracy indices of the National Forest Inventory map in Mexico

Global and per class accuracy indices are presented in table 5 for each eco-geographical area. Confusion patterns among classes were presented in error matrices by Couturier et al. (2010) and permitted a detailed study of the quality of the cartography in terms of biodiversity representation. The global accuracy indices ranged from 64 per cent (Candelaria) to 78 per cent (Los Tuxtlas). Accuracy levels were lower in forest-dominated Candelaria (64 per cent) and Tancítaro (67 per cent) areas than in nonforest-dominated Cuitzeo (75 per cent) and Los



Source: Mas et al. (2002a)

Fig. 1. National Forest Inventory map of Mexico in year 2000 (NFI-2000 map)

Fig. 2. Location (shaded in grey) of the four eco-geographical areas in Mexico.

Class	Name	Biome	Cuitzeo		Tancítaro		Tuxtlas		Candelaria		Total
			Area frac	Area (km^2)	Area frac	Area (km^2)	Area frac	Area (km^2)	Area frac	Area (km^2)	area per class (km^2)
100	Irrigated crop	Cropland	0.1411	564.97	0.0106	13.45					578.42
110	Hygrophilous crop		0.0048	19.04							19.04
130	Cultivated grassland						0.6058	1839.75	0.1708	1908.25	3748.01
200	Perennial crop		0.0021	8.27	0.2904	367.70	0.0129	39.16			415.13
210	Annual crop		0.2356	943.14	0.0803	101.69	0.1765	535.84	0.0070	77.85	1658.51
300	Forest plantation		0.0071	28.24							28.24
410	Fir forest	Temperate forest	0.0037	14.72							14.72
420	Pine forest		0.0041	16.32	0.1658	209.99	0.0011	3.36			229.67
421	Pine forest & secondary vegetation		0.0036	14.31	0.0634	80.23	0.0011	3.37			97.90
510	Oak-pine forest		0.0958	383.34	0.1907	241.47					624.82
511	Oak-pine forest & secondary vegetation		0.0325	130.29	0.1284	162.54	0.0028	8.48			301.31
600	Oak forest		0.0232	92.88			0.0011	3.44			96.32
601	Oak forest & secondary vegetation		0.0553	221.54	0.0017	2.16	0.0041	12.49			236.20
700	Cloud forest	Tropical forest	0.0029	11.73			0.0035	10.78			22.51
800	Median/high perennial tropical forest						0.1213	368.43			368.43
801	Median/high perennial tropical forest & secondary vegetation						0.0292	88.56			88.56
820	Median/high subperennial tropical forest								0.5010	5595.31	5595.31
821	Median/high subperennial tropical forest & secondary vegetation								0.0880	982.82	982.82
830	Low subperennial tropical forest								0.1765	1971.60	1971.60
831	Low subperennial tropical forest & secondary vegetation								0.0025	27.57	27.57
920	Subtropical scrubland	Scrubland	0.0194	77.58							77.58
921	Subtropical scrubland & secondary vegetation		0.0768	307.25							307.25
1000	Mezquital		0.0004	1.51							1.51
1200	Chaparral										0.00

1320	Savanna	Grassland							0.0108	120.13	120.13
1330	Induced grassland		0.1594	638.04	0.0032	4.02	0.0004	1.08	0.0039	43.65	686.80
1400	Mangrove	Hygrophil ous vegetation					0.0066	20.15	0.0060	66.93	87.08
1410	Hygrophilous grassland		0.0209	83.50			0.0019	5.86	0.0225	251.24	340.60
1510	Halophilous vegetation	Other vegeta-tion types	0.0069	27.78					0.0039	43.56	71.34
1600	No apparent vegetation cover	Other cover types			0.0390	49.43	0.0007	2.11			51.54
1700	Human settlement		0.0250	100.02	0.0265	33.59	0.0065	19.82	0.0009	10.19	163.62
1800	Water		0.0796	318.75			0.0244	74.01	0.0063	70.11	462.87
	All		1.0000	**4003.23**	1.0000	**1266.28**	1.0000	**3036.69**	1.0000	**11 169.21**	**19 475.41**

Area frac: Fraction of the eco-geographical area. The 'community with alteration' taxonomic level refers to the 'sub-community' level in Palacio-Prieto et al. (2000).

Table 3. Class distribution (subcommunity and biome aggregation levels) of the NFI-2000 map in the four ecogeographical areas

Aerial photography	Data type/ interpretation	Scale/resolution	Year	Number of photographs
Cuitzeo	Prints/stereoscopic	1:37 000	1999	244
Tancítaro	Prints/stereoscopic	1:24 000	1996	152
Tuxtlas	Digital/on screen	1:75 000 /	2000	12
		1.5 m grain	1996	14
Candelaria	Prints/stereoscopic	1:75 000	Jan 2000–Mar 2002	174

Table 4. Aerial photography used for the accuracy assessment of the NFI-2000 map.

Tuxtlas (78 per cent), possibly because of the higher (confusion prone) diversity of forest classes than nonforest classes in the NFI classification scheme.

For 'other cover types' ('no vegetation cover', 'water' and 'human settlement'), a high accuracy (79 per cent and above) was registered, with the only exception of 'water' in Candelaria, where water bodies are small, dispersed and often seasonal. Visually, the spectral separability of these land covers within their group and with respect to other groups is indeed among the highest on conventionally used Landsat colour composites (e.g. 342). The mangrove class also recorded high accuracy in both Candelaria and Los Tuxtlas areas where mangroves are present. Conversely, very high interconfusion within aquatic non tree vegetation covers is evident when hygrophilous grassland and halophilous vegetation are both present (Cuitzeo and Candelaria). We also found high levels of commission error in hygrophilous grassland at the expense of induced grassland in Los Tuxtlas and Candelaria. The spectral ambiguity and variability (across inundation phases) of these aquatic vegetation types is probably one of the key explanations for this observed high confusion. Former INEGI maps mostly confirm the reference data in exhibiting such errors of the NFI-2000 map. Finer trends registered for forest types and land use categories vary according to the ecogeographical area as described in Couturier et al. (2010).

By contrast with the relatively high levels of accuracy of vegetation cover with little modification (classes without 'secondary vegetation'), many errors were reported for classes

Class	Taxonomic name		Cuit-zeo		Tancí-taro		Tux-tlas		Cand-elaria		Total area per
Code	(Community with alteration)	(Biome)	User's	Prod-ucer's	User's	Prod-ucer's	User's	Prod-ucer's	User's	Prod-ucer's	class (km2):
100	Irrigated crop	Cropland	87	90	22	23					578.42
110	Hygrophilous crop		63	75							19.04
130	Cultivated grassland						83	90	69	78	3748.01
200	Perennial crop		99	100	86	84	57	9			415.13
210	Annual crop		71	78	87	64	52	99	75	9	1658.51
300	Forest plantation		83	33							28.24
410	Fir forest	Temperate forest	76	100							14.72
420	Pine forest		79	59	41	44	85	31			229.67
421	Pine forest & sec veg		12	5	8	44	0	-			97.90
510	Oak-Pine forest		96	92	77	67					624.82
511	Oak-Pine forest & sec veg		45	68	56	55	6	83			301.31
600	Oak forest		92	40		-	28	32			96.32
601	Oak forest & sec veg		46	95	5	100	70	82			236.20
700	Cloud forest	Tropical forest	0	-			100	100			22.51
800	Median/high perennial trop forest						92	66			368.43
801	Median/high perennial trop forest & Sec Veg						63	42			88.56
820	Median/high subperennial trop forest								70	89	5595.31
821	Median/high subperennial trop forest & Sec Veg								55	45	982.82
830	Low subperennial trop forest								52	61	1971.60
831	Low subperennial trop forest & Sec Veg								32	1	27.57
920	Sub-tropical scrubland	Scrubland	78	29							77.58
921	Sub-tropical scrubland & Sec Veg		88	63							307.25
1000	Mezquital		0	-							1.51
1200	Chaparral			-							0.00
1320	Savanna	Grassland							22	-	120.13
1330	Induced grassland		60	91	36	66	69	11	67	26	686.80
1400	Mangrove	Hygrophilous vegetation					86	99	87	96	87.08
1410	Hygrophilous grassland		47	68			53	100	70	44	340.60
1510	Halophilous vegetation	Other vegetation types	25	21					9	41	71.34
1600	No apparent vegetation	Other cover types			82	92	87	100			51.54
1700	Human settlement		100	63	97	88	92	92	80	72	163.62
1800	Water		89	92			100	98	48	96	462.87
	Total		**74.6**		**67.3**		**77.9**		**64.4**		**19475.41**

Same conventions as table 2. Trop: Tropical; Sec Veg: Secondary Vegetation; Taxonomic level *Community with alteration* refers to level *Sub-community* in Palacio-Prieto et al. (2000)

Table 5. Accuracy indices (user's and producer's) per class of the National Forest Inventory (*Community with alteration*) in the four eco-geographical areas.

of highly modified vegetation cover (classes 'with secondary vegetation'). For instance in Cuitzeo, the accuracy of sub-tropical scrubland (78%), oak-pine forest (97%), pine forest (79%) and fir forest (76%) contrast with the accuracy of highly modified oak forest (46%), highly modified pine forest (12%) and highly modified mixed forest (45%). From both the taxonomical and landscape points of view, a class of highly modified vegetation cover is close to a wide set of land use classes as well as low modification vegetation cover classes, which makes it prone to more confusions than a class of low modification vegetation cover. These low accuracy levels, however, appear as a real challenge for improving the quality of future cartography because degradation studies are an important part of forest management and biodiversity monitoring.

Region of the world	Acronym of project and year of cartography	Prevailing biotic environ-ment	Assessment design	Number of assessed classes		Global accuracy index	Reference publication
				Forest	Total		
Mexico (4 areas)	**IFN 2000**	**Mega-diverse**	**Probabilistic**	**19 (29)**	**32 (75)**	**64-78%**	**Couturier et al. (2010)**
Southwest USA	GAP 2000	Temperate dry	Partial (near to roads)	18 (27)	85 (125)	*61%*	Lowry et al. (2007)
India	ISRO-GBP 1999	Mega-diverse	Partial (in 3 states of the country)	14	35	*81%*	Joshi et al. (2006)
Canada	**EOSD-Forest 2000**	**Temperate**	**Probabilistic**	**10**	**18**	**67%**	**Wulder et al., (2007)**
European Union	**CorineLC 2000**	**Temperate**	**Probabilistic**	**3**	**22 (44)**	**74.8%**	**Buttner & Maucha (2006)**
USA	**NLCD 1992**	**Temperate**	**Probabilistic**	**3**	**21**	**46-66% (per administrative region)**	**Stehman et al. (2003)**
Nigeria	1990	Tropical humid	Partial	3	8	*74.5%*	Rogers et al. (1997)
Legal Amazon, Brasil	GLC 2000	Tropical humid and dry	Partial	3	5	*88%*	Carreiras et al. (2006)
South Southeast Asia	**TREES 2000**	**Mega-diverse**	**Probabilistic for biome level**	**1 (17)**	**4 (40)**	**72% (biome level)**	**Stibig et al. (2007)**

Same conventions as table 1.

Table 6. Global accuracy indices of regional Land Use Land Cover cartography, derived from major published assessment studies in the world. The list is sorted by the number of assessed forest classes.

5.2 Comparison with other assessed international cartography

Table 6 presents the global accuracy indices found in each study listed in table 1. As a means of acknowledging the difficulty of mapping forest classes, the list in table 6 was sorted by the number of forest classes actually assessed in the study. With the exception of the GAP2000 very detailed study, the partial (non probabilistic) assessments yield higher accuracy indices (from 74.5 to 88%) than probabilistic assessments (in bold; from 46 to 74.8%). However, a partial assessment is possibly optimistically biased because it is not representative of the quality of the entire map, although it is impossible to estimate the magnitude of this bias (Stehman & Czaplewski, 1998). Among probabilistic assessments, the accuracy index in both densely forested areas (Tancítaro: 64.4% and Candelaria: 67.3%) is comparable with the results of assessments with a high amount of forest classes, en Canada (67%). Likewise, the accuracy index in areas where land use classes prevail (Cuitzeo: 74.6% and Los Tuxtlas: 77.9%) is comparable with the results of other assessment, such as the CorineLC 2000, mainly focused on land uses in Europe (74.8%)and with TREES2000 in South and Southeast Asia (72%). The accuracy indices in Cuitzeo and Los Tuxtlas, nevertheless, are higher than the range of results in other probabilistic assessments (46-66%). The NFI and the TREES 2000 cartographies have similar spatial detail (1km2 resolution) although the assessment of TREES 2000 was at biome level (only 4 assessed classes). The cartographic challenge of the NFI 2000 was greater at taxonomical detail of 'community with alteration' (32 classes).

The NFI map is also characterized by a higher taxonomic diversity than the other probabilistically assessed maps in the USA, Canada and Europe. However, the Minimum Mapping Unit (MMU) of those maps is much smaller (approximates the Landsat pixel size) than the MMU of the NFI, which in turn is a greater challenge for mapping accuracy. Considering these compensating factors (taxonomic richness but poorer spatial precision), the NFI map achieves comparable or better accuracy indices than the cited cartography, in a limited extent of the Mexican territory but in an extent that may reflect several scenarios and complexity of the national LULC.

The low accuracy registered for highly modified vegetation classes has been observed in the EOSD Canadian experience for forest covers of various density grades. Wulder et al. (2007) conclude that the highest source of errors in their map is caused by confusions among density grades. The confusion among density/ alteration classes caused by ambiguity on the Landsat imagery could be related, in the case of the NFI map, to the inclusion of the secondary vegetation in a great number of forest classes. This inclusion may be simpler and less confused in other projects such as GAP2000, TREES2000, or the NFI of year 1994 in Mexico where in spite of many forest classes, the presence of secondary vegetation is aggregated in very few classes.

A possible improvement of the detailed LULC cartography in Mexico could derive, therefore, from aggregating secondary vegetation classes into, for example, forest subtypes such as 'temperate forest with secondary vegetation' and 'tropical forest with secondary vegetation'. Such grouping could reflect a better matching of the classification system with the discrimination capacity of Landsat-like sensors in complex forest settings.

6. Conclusion

Land cover maps with detailed forest taxonomy are an essential basis for sustainable forest management at regional scale. This cartography is especially useful in highly biodiverse areas. A deforestation rate, a biodiversity conservation program or a land use change study critically depend on the quality of such cartographical datasets. Yet, for the overwhelming majority of governmental agencies in the world, the quality of the cartography is easily confounded with the spatial resolution, or temporality of the satellite imagery used in the map production process. Confusions between thematic classes on the imagery that lead to errors on the map are simply ignored, so that the derived deforestation rates, forest extent baselines, etc. are quantities without error margins and therefore these quantities lack statistical validity.

Based on a review on accuracy assessment studies in the world, this chapter first reports the occurrence of substantial errors in detailed regional land cover maps. The chapter then reports the recently developed research on the quality assessment of the LULC cartography in Mexico. A probabilistic accuracy assessment framework was developed for the first time in a mega-diverse area for taxonomically detailed maps and applied to four distinct eco-geographical areas of the Mexican NFI map of year 2000.

As a first feature of the accuracy assessment, a two-stage hybrid sampling design was applied to each of the four eco-geographical areas. Proportional stratified sampling was employed for sparsely distributed (rare) classes. This design had been fully tested and compared with existing designs in Couturier et al. (2007).

Second, with the utilization of reference maplets and GIS techniques, this research incorporated thematic and positional uncertainty as two parameters in the design, which created the possibility for a map user to evaluate the map at desired levels of positional and thematic precision. Couturier et al. (2009a) illustrated the practical usefulness of this possibility in the case of the NFI map, with landscapes composed of intricate tropical forest patches.

The accuracy of the NFI map was then compared with published error estimates of regional LULC cartographic products. We found that the quality of the NFI 2000 map (accuracy between 64% and 78%) is of international standards. This information is valuable given that the taxonomical diversity enclosed in the NFI is much higher than the currently assessed international cartography. Additionally, we found that the majority of land use classes and of low modification vegetation cover classes in the NFI are characterized by accuracy indices beyond 70%. By contrast, the NFI map registers low accuracy for highly modified vegetation cover classes. It is suggested that the quality of the cartography could be improved in the future by grouping categories containing secondary vegetation.

The assessment of the NFI 2000 cartography in four eco-geographical areas still constitutes a pilot study, confined to a limited extension, in a mega-diverse area. Since 2003, the monitoring of vegetation cover in Mexico is partly ensured using the MODIS sensor (CONAFOR, 2008), which is comparable with the SPOT-VEGETATION sensor used by Stibig et al. (2007) in Asia. We recommend the method presented here be extended to the national level for comprehensive accuracy assessment of future INEGI Serie V or vegetation cover annual maps of SEMARNAT. This method would ensure very reasonable costs and would contribute to solve the polemical discussions on the reliability of deforestation rates and land use change rates in the country.

We conclude that the work presented here sets grounds, as the first exercise of its kind, for the quantitative accuracy assessment of LULC cartography in highly bio-diverse areas. Among assets of this work is the knowledge, for the first time in a highly bio-diverse region, of the LULC quality that can be expected from the interpretation of medium resolution satellites.

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3

Sustainable Management of Lenga (*Nothofagus pumilio*) Forests Through Group Selection System

Pablo M. López Bernal[1,2,3], Guillermo E. Defossé[1,2,3],
Pamela C. Quinteros[1,2] and José O. Bava[2,3]
[1]*Consejo Nacional de Investigaciones Científicas y Técnicas – CONICET*
[2]*Centro de Investigación y Extensión Forestal Andino Patagónico – CIEFAP*
[3]*Universidad Nacional de la Patagonia San Juan Bosco – UNPSJB*
Argentina

1. Introduction

1.1 Distribution and environmental gradients of lenga forests

Lenga (*Nothofagus pumilio* (Poepp. *et* Endl.) *Krasser*) is a native tree species widely distributed in the Andean forests of Patagonia. In Argentina, lenga forests cover almost the entire length of the sub-Antarctic forests on the eastern slopes of Andean Cordillera, from the 35 º 35 'S latitude parallel in the province of Neuquén to the 55 º S in the province of Tierra del Fuego (Figure 1). This species usually occupies the upper portion of the altitudinal limit of woody vegetation (up to 2000 masl) in its northern distribution area, while it grows near sea level in its southern distribution area in Tierra del Fuego (Donoso Z., 1995, Tortorelli, 2009, Veblen et al., 1977).

Lenga is adapted to grow under a great variety of soils, environmental conditions, and disturbance regimes (Schlatter, 1994). In fact, this species could be found in areas in which average annual precipitation may reach 500 mm year^{-1} (under a Mediterranean type of climate), to others reaching 3,000 mm year^{-1} (under either iso-hygro or Mediterranean type of climates). Lenga is also capable of supporting extreme temperatures, from mean annuals of 3.5 to 4 °C in upper altitudinal areas (Schlatter, 1994) to 7 to 9 °C in milder areas at lower altitudes or also near sea level.

In the northern part of its distribution, lenga grows under a typical Mediterranean climate, with precipitation concentrated during winter and early spring as either rain or snow, followed by a dry and mild period during summer and early fall. Going south, this regime gradually changes to more iso-hydric conditions, being precipitation more evenly distributed along the year. In the northern part of its distribution and up to the 52 º S, however, the amount of annual precipitation is greatly influenced by the barrier that imposes the Andean Cordillera, which creates one of the most spectacular precipitation gradients of the world. There, the western humid winds coming from the South Pacific Ocean discharge most of the precipitation as they go upward to the upper parts of the Andes, passing to the eastern slopes as more dry air masses that rapidly lose their humidity content. This makes that upper mountain ranges near the border with Chile may receive 5000 mm of precipitation per year, while in less than 50 to 80 km toward the Patagonian steppe, precipitation sharply diminishes to ca 500 mm annually (Barros et al., 1983, Jobbágy et al., 1995, Veblen et al., 1977). To the South of the 52 and up to the 55 º S parallel in the island of Tierra del Fuego, a regular rainfall pattern occurs, with rainfall evenly distributed throughout the year (Burgos, 1985);

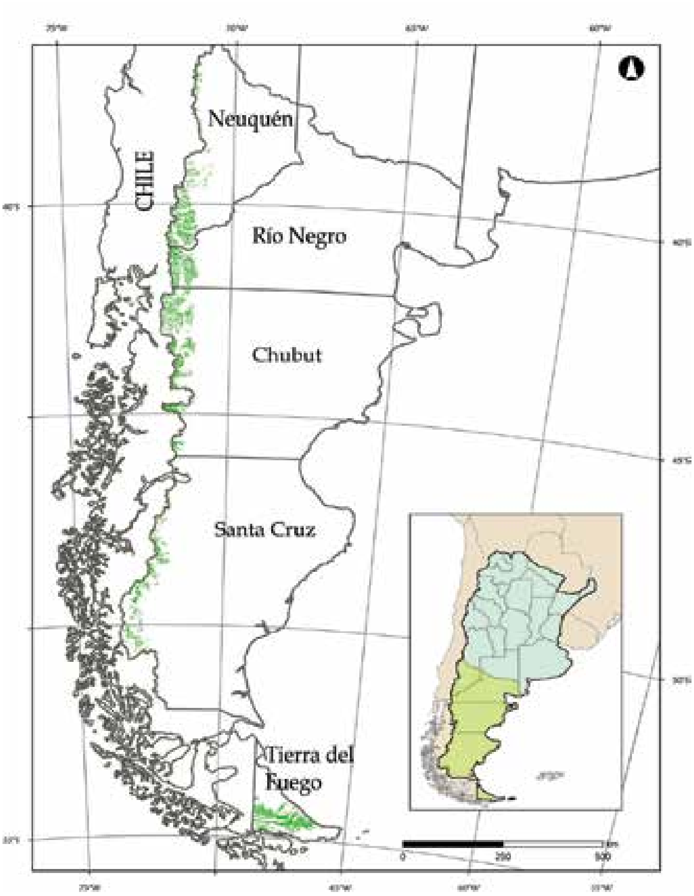

Fig. 1. Distribution of *N. pumilio* forests (green shading) in Argentinean Patagonia.

1.2 Disturbance regime

Throughout its wide distribution area, lenga stands are clearly distinguishable from other component of the Andean forests, being composed of simple monospecific structures with narrow ecotones (Donoso Z., 1995). However, given the different environments in which it develops, lenga presents different structures and regenerative dynamics, mainly associated with the frequency, magnitude and severity of disturbances such as windstorms, fires, avalanches, landslides, or the falling of senescent trees (Donoso Z., 1995, Veblen et al., 1996). As a consequence of these disturbances, lenga stands may present either even or uneven-aged structures, both situations representing extremes in a range of different possible structures. At the southern end of lenga distribution in Tierra del Fuego, tree falls usually occur due to wind storms, and this result in even-aged young structures (Rebertus et al., 1997). Furthermore, the same wind storms that cause large falls may also lead to formation of small gaps in the forest canopy. Uneven-aged structures are usually originated in mature forests located in favorable sites at low altitudes having low frequencies of catastrophic disturbances or human interventions. In these areas, the falling out of senescent trees may promote the opening of gaps or patches of about 0.1 ha (Bava, 1999, Veblen & Donoso Z., 1987) in which regeneration begins. These patches generally possess favorable undergrowth conditions which allow the formation of small clumps of saplings. The result of this process is a multi-aged and multi-strata forest, even when the formation of these gaps may be an episodic phenomenon (Rebertus & Veblen, 1993).

In relation to its tolerance to shadow, lenga has been classified as either "purely heliophilous" (Mutarelli & Orfila, 1971), "semi-heliophilous" (Tortorelli, 2009), "medium tolerant" (Rusch, 1992) to "semi-tolerant" (Donoso Z., 1987). These controversial or even opposite classifications are probably due to the different habitats these descriptions came from. It has been well established that for many species, proper development may depend on the limiting resources a given environment may have (Choler et al., 2001) so the radiating needs of lenga regeneration may significantly vary depending on a set of other environmental factors (Rusch, 1992).

In sites with high rainfall levels (i.e. South of Tierra del Fuego and West of Chubut province), lenga regeneration is established even after major disturbances affecting up to hundreds or even thousands of hectares (Rebertus et al., 1997, Veblen et al., 1996). The same occurs in forests affected by intensive forest harvesting (Gea Izquierdo et al., 2004, Mutarelli & Orfila, 1971, Rebertus & Veblen, 1993, Rosenfeld et al., 2006). By contrast, in sites with water deficit during the summer, as in the northern sector of lenga distribution in Río Negro, Neuquén and Chubut provinces, regeneration cannot be established with low canopy coverage (Bava & Puig, 1992). In these areas, regeneration establishment is strongly influenced by water availability and usually occurs in small gaps caused by falling trees (Rusch, 1992).

Recent studies have analyzed the effects of micro-environmental factors on the establishment and growth of lenga seedlings in natural gaps. These studies showed that in the driest sites of lenga distribution, the shade generated by individuals from the edge of the gaps and the presence of coarse woody debris, produce a facilitator effect on seedling establishment (Heinemann & Kitzberger, 2006, Heinemann et al., 2000). Seedling survival in these xeric sites have been positively related to water availability, while in mesic sites survival seems to be controlled by both water availability and light (Heinemann & Kitzberger, 2006, Heinemann et al., 2000).

2. History of productive use of *N. pumilio* forests in Argentinean Patagonia

Most of lenga productive forests in Argentinean Patagonia, owned by either private or state sectors, started to be exploited at the beginning of the XXth century, but did not reach significant levels of harvest until mid-century, with the emergence of large sawmills that used almost exclusively high quality timber. Since then, the closing down of these large sawmills and the gradual installation of small and medium sawmills generated changes in harvesting techniques, extraction rates, and final products. These changes were usually marked by the lack of effective control policies by the state administration, which lead to the absence of sustainable management practices. Furthermore and to worsen this situation, these forests have been traditionally used as summer pastures for cattle, which in many cases has caused the degradation of the understory, with long delays in, and even preclusion of, regenerative processes.

The objective of this chapter was to analyze the evolution of productive schemes of lenga forests along their history of use, which will help us understand the overlap of strains on this resource, impacts on their conservation status, and the difficulties that currently have the implementation of sustainable management systems. For this purpose, we got information derived from published analyses, statistical records of the Forest Administration, analyses of historical harvesting, and of the impacts of livestock on forest regeneration. This information is presented for two contrasting situations, located one in the northern lenga distribution area in Chubut province and the other in its southern distribution area in Tierra del Fuego.

2.1 Pre-industrial

The original inhabitants of continental Patagonia were mostly nomadic Indian tribes that depended largely on the guanaco (*Lama guanicoe Muller*) for their livelihood. These tribes used ecotone and steppe areas of and did not settled in the Andean forests, although there are some examples of communities who lived associated with *Araucaria araucana (Molina) K. Koch* forests in northern Patagonia. Lenga forests, located at higher altitudes, were only occasionally used as firewood in the journeys crossing the Andes (Musters Chaworth, 1871). In Tierra del Fuego, unlike continental Patagonia, guanaco used lenga forests as part of its habitat, perhaps due to the absence of its natural predator, the puma (*Puma concolor* Linnaeus). Some Indian tribes lived much of the year in these interior forests, while others were established on the shores of the Beagle Channel, all surrounded by lenga forests. In this region, the use of lenga for small constructions and canoes, although in small scale, has been reported (Bridges, 2000). The major effect of indigenous peoples on Patagonian forests has been the recurrent employment of fire, either for hunting purposes or used as a communications signal (Kitzberger & Veblen, 1999).

During white settlement, cracked poles, rustic tables, or shingles, were widely used products from lenga forests, but undoubtedly, fire was the most devastating factor affecting them. In the Argentine sector of Tierra del Fuego, an estimated 20,000 ha were burnt in the early twentieth century. Contemporarily and in an attempt to open land for sheep raising, pioneers in the Chilean Patagonia initiated what could now be called catastrophic fires, burning large portions of lenga woodlands (2.8 million ha, Fajardo & McIntire, 2010), reducing their original area by a half (Otero Durán, 2006). In the rest of its distribution area, thousands of hectares were also burnt, although reliable data are not available (Willis, 1914). The recovering of lenga forests after those fires depended on a multiplicity of factors, among which the availability of safe sites (*sensu* Harper, 1977) for seed germination and seedling establishment, and the grazing pressure exerted on the burned sites played a crucial role. The outcomes in former lenga forests were then open fields to raise sheep or the slow recovery of lenga forests. After that beginning and in the mid 40´s, factors such as the strengthening of national protected areas, the decline of sheep production and the displacement of rural populations modified this process of impoverishment or forest clearance, at least at regional level. Though, the lenga forests that were formerly used as summer ranges for sheep gradually changed to cattle grazing areas. It is interesting to note that the introduction of cattle ranching in the area has a vague origin, as the early explorers (Musters Chaworth, 1871), cite the existence of wild cattle in the forests of the region already in 1870, possibly coming from Valdivia, Chile (settled around 1600), or escaped from the cattle drives that native communities transported from Argentina to Chile.

2.2 The beginnings of industrial use: High grading

The first forest industries installed in the early twentieth century, either for medium or small sawmills, or for wood veneer production, were characterized by softly logging, cutting only healthy, medium-sized trees. Stem rots, caused mostly by fungi of the genera *Postia* and *Piptoporus*, is a very common phenomenon that affects lenga trees, being very important in old age trees. This determined that in virgin forests, only a small proportion of trees contained good quality timber. For that reason and in general, forest workers used to cut down only trees in good health status, medium-sized (40 - 50 cm DBH), which generally did not exceed 10% of forest trees in a stand. This type of "soft logging" was locally known as "floreo" (high grading).

2.3 Forest management plans

The first Argentine Forest Law was put in force in 1948. While the concept of forest management, as a synonym of timber production, was prevalent in that law, it included articles about protection of soil, water and biodiversity. Although it mandated for the implementation of Forest management plans, its principles and regulations were applied sparingly. As a result, logging continued in public forests in an unplanned way. In the mid 50´s of the XXth century, the first forest management plans were designed and applied by Croat forest engineers, who arrived to Argentina after the World War II. These plans represented a breakthrough for the understanding of lenga forests, but had little practical effects on forest management due to the weaknesses of the Administrative forestry services of the Patagonian provinces.

By the 80's, the practice of giving access to cut lumber in public or private forests depended on the approval of forest management plans by the provincial forest service, practice that became usual. However, these were just cutting plans, without long term planning horizon and being controlled at different levels of implementation.

Silvicultural aspects were changing over time with the evolution of knowledge about forests dynamics, from the early experiences on shelterwood systems in Chile (Cruz M. & Schmidt, 2007), clear-cut in Argentina (Mutarelli & Orfila, 1971), to the currently used alternatives, ranging from a group selection system (Bava & López Bernal, 2005) up to a variation of shelterwood systems with dispersal and-or aggregate retention (Martínez Pastur et al., 2009).

2.4 Cattle

As already mentioned, the first activity developed in Patagonian was sheep ranching. Near the Andes, the usual ranching scheme was a system that alternated winter grazing (locally called invernadas) in low areas with summer grazing areas at higher altitudes in the forest (called veranadas). There are plenty of examples of this system in mountain areas around the world. In the mid-twentieth century, with increasing population established in the area, cattle raising was becoming important, with the same production scheme.

In lenga forest ecosystems of Patagonia, herbivory causes severe impacts, because this species is palatable to both wild (camelidae, deer and leporidae), and domestic livestock, and heavy grazing can prevent forest regeneration (Veblen et al., 1996). In Argentina, lenga forests suitable for timber production are mainly concentrated in the provinces of Chubut and Tierra del Fuego. Lenga forests in the province of Chubut are also a very important part of traditional cattle management, which similarly to what formerly occurred with sheep, alternates winter fields in the steppe with summer fields at the mountains (York et al., 2004).

Therefore, 19 % of the forests suitable for timber production are potentially degraded by cattle grazing (Bava et al., 2006). In Tierra del Fuego, by contrast, is the wild guanaco (*L. guanicoe*) which has a negative impact on regeneration. This impact has been reported especially in forests located northwards of Fagnano Lake and in the Chilean side of the Isla Grande de Tierra del Fuego (Cavieres & Fajardo, 2005).

It is possible to distinguish between direct and indirect effects on forest regeneration caused by large herbivores. The direct consumption of seedlings keeps the lenga regeneration stunted (Perera et al., 2004) and multi-stemmed (Bava & Rechene, 2004). The consumption of other species may cause a decrease in the diversity of the understory. The transport of seeds through feces allows the introduction of exotic species (Bava & Puig, 1992). Immersed in lenga forests is frequent to find meadows, locally called "mallines" in the province of Chubut or "vegas" in Tierra del Fuego. These meadows are highly valued by farmers because of their high productivity in forage species. Due to the existence of meadows near the forests, and the little vegetation cover that characterizes the undergrowth of lenga (Lencinas et al., 2008), an intensive use of resources and a great impact on regeneration and understory forest areas close to the meadows has been reported (Quinteros, unpublished data). The changes that livestock generated in the understory, such as increased coverage of grazing tolerant species, mainly exotic, constitutes an indirect effect on the development of lenga regeneration, because the high grass coverage competes for water resources with lenga seedlings, affecting their growth and development (Quinteros, unpublished data).

3. Group selection system in *N. pumilio* forests

3.1 General concepts

The selective silvicultural systems are characterized by generating uneven-aged stands where regeneration layer strongly interacts with the mature forest, and this interaction could either be favorable or unfavorable for seedlings or saplings of different species (Daniel et al., 1979). With this method, individual trees are removed (or small groups of them), opening small gaps that can be used by tolerant species. Harvesting procedures require frequent partial cuts, where the harvest interval is called "Cutting Cycle" and there is not a rotation age where all production is harvested, as in the even-aged methods (Daniel et al., 1979).

In the Group Selection System (GSS), harvested trees are pooled in small groups (typically up to 10 mature trees), thus creating gaps in the canopy larger than the individual selection cuttings, which are better suited to the requirements of semi-tolerant species, as is the case of lenga (Bava & Rechene, 2004). It also provides some advantages of even-aged stands, as the saplings grow in conditions of intra-cohort competition. This competition favors the production of better shaped stems, while the harvest is partially concentrated, reducing its costs and minimizing the damages from falling trees (Daniel et al., 1979).

Under this scheme, some decisions that have to be taken are (Davis & Johnson, 1987):

- Cutting cycle: time between harvest entries on each stand.
- Reserve growing stock level: residual volume or basal area (BA) immediately after harvest.
- Group, patch or gap size: defined in function of objective species requirements.

The historical origin of this type of management helps gauge its applicability and scope in different productive forest systems. Uneven-aged forest management had its origins in Central Europe, where since the twelfth century exist harvest protocols regulating the forest extraction, by limiting the numbers of individuals or the volume to be harvested (Becking, 1995). However, the real practice was a selective extraction of the best stems (high-grading) without control policies that would ensure the regeneration and future productive potential of the forest, affecting negatively the productive quality of large areas. For this reason, between the fifteenth and eighteenth centuries this type of logging was progressively sidelined (Puettman et al., 2009), and new forest practices, such as clear-cutting, emerged (Becking, 1995). In the early nineteenth century, an alternative selection system was formalized for various regions of Europe, where clear-cuts were banned and where the landowners of small forest stands were especially interested in the high frequency of harvesting (short cutting cycles) thus maintaining a continuous cash flow (Puettman et al., 2009).

As mentioned in the previous section, the historical context of lenga forests in Patagonia, and particularly in the north of its distribution, is in some ways comparable to the origins of the implementation of selective cuts. The predominance of small and medium producers, the low productivity of these forests and the low control capacity of state agencies has meant that in most cases the harvest have been a high-grading, often of low intensity. Thus the GSS, where harvesting is simultaneous with other silvicultural tasks (such as thinning or regeneration release) on the one hand represents an economically feasible objective for local producers, and on the other a simplification of the control tasks posed by the state.

Moreover, the prolonged rotation periods needed for lenga forests and the brief history of the implementation of these schemes in Patagonia prevents direct observation of long-term management examples. Given this situation, models of conservation and sustainable management based on emulation of natural disturbance regimes are very attractive for developing sustainable management practices (Perera et al., 2004).

From this view, various management systems have been proposed through intense felling as clear-cuts or shelter-wood cuttings (Arce et al., 1998, Martínez Pastur et al., 2009); with the intention of imitating mass disturbances that naturally occur, especially in southern Patagonia. However in Chubut province, the rainfall regime with wet winters and prolonged dry periods in summer, prevents the proper regeneration establishment in large areas subject to direct sunlight (Rusch, 1992). For this reason, an adaptation of Group Selection System (1997) is currently proposed for these sites, imitating the predominant disturbance of gap dynamics (Bava, 1999, Veblen & Donoso, 1987, Veblen et al., 1981, Veblen et al., 1980). This promotes the establishment of regeneration patches formed by felling from one to six trees (Antequera et al., 1999, López Bernal et al., 2003).

3.2 Adaptation of GSS to *N. pumilio* forests

3.2.1 Definition of canopy gap

The minimum unit for the application of different treatments in a group selection system is the "forest patch" or "canopy gap". However, the definition of canopy gap or its size is often unclear. On one hand, there are different definitions of the “canopy gap limit” and on the other, there are various ways to simplify its form. Additionally, there are several field methods for gap size measurement. López Bernal et al. (2010) compared different ways of this three issues, specifically:

i. Gap limit definitions: there are two main schools of thought defining this parameter. One is proposed by Brokaw (1982), who defined the gap as a "hole" in the forest that extends across all levels to an average height of two meters above the ground, and whose boundaries are defined as vertical walls. The space calculated by this method is usually called the "canopy gap" (Figure 2). However, this method has been criticized because it underestimates the area affected by the gap (Popma et al., 1988). The other definition was proposed by Runkle (1981), based on the concept of an "expanded gap" whose limits extend to the base of the bordering trees. Runkle argued that this method has the advantage of including the area where light availability is directly and indirectly influenced by the gap.

ii. Calculation methods: regardless of the gap type (i.e. the definition of its limits), there are several methods to calculate or estimate the surface area of a gap. These methods mainly differ in the degree of form simplification, i.e. how well they capture boundary irregularities, moving from ellipses to polygons, octagons or hexadecagons, either with straight sides or with sections of an ellipse (Brokaw, 1982, Green, 1996, Lima, 2005, Runkle, 1981, Zhu et al., 2009).

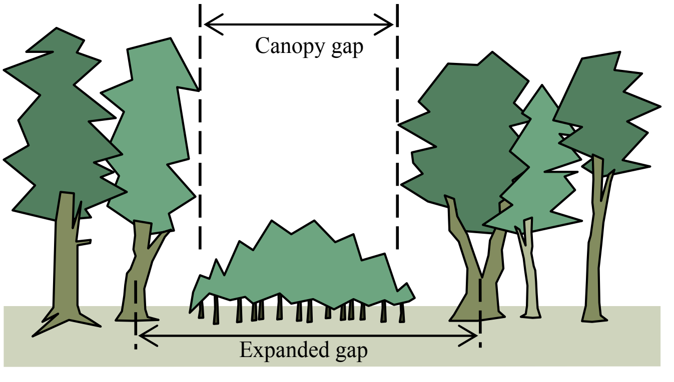

Fig. 2. Canopy and expanded gap scheme.

iii. Field Methods: Finally, different methods, such as measuring directions and distances from gap center or the triangles method (Lima, 2005), may be applied to measure the variables needed to calculate gap size. These methods may be more or less effective depending on the characteristics (such as understory density and height) of the forest being studied. The optimal field method must also be evaluated in terms of its simplicity of operation, time requirement, necessary tools, etc.

The three issues listed above are all based on the conception of a gap as a surface. The relationship between gap diameter and canopy height has also been used as a reference parameter in some studies (Albanesi et al., 2008, Minckler & Woerheide, 1965, Runkle, 1985), especially where gap creation has been used as a management activity. Canopy height is a parameter with a direct influence on the amount of received radiation. Therefore, the addition of canopy height in any calculation may lead to a significant improvement in the accuracy of gap size estimation.

López Bernal et al. (2010) concluded that the Polygonal Expanded Gap Diameter / dominant canopy Height ratio (from now on D/H) is an expeditious method to characterize gap size. This method not only allows the estimation of the incident radiation, but also the comparison of gaps of different stands and even of different species (Albanesi et al., 2008, OMNR, 2004). The method also incorporates dominant canopy height, which improves gap characterization at different sites. A range for this variable for lenga forests is between 14 and 30 m, and makes D/H an adaptable parameter. The strong correlation between D/H and incident radiation makes this parameter a good radiation predictor for gaps in a broad range of gap sizes and with canopies of different heights, and represents a useful tool, both to define silvicultural guidelines and to carry out forest ecological studies.

3.2.2 Tree marking guidelines

The main strategy proposed by Bava and Lopez Bernal (2005, 2006) for marking in virgin or high graded forests, focuses on trees from which it is currently possible to gain good quality logs, or on young healthy trees showing high timber potential. If these trees exceeded the minimum diameter at the breast height (DBH) of 35 cm (or 40 cm if they had smooth bark), they are felled in order to open or expand the gap, but if they not exceed that diameter, their growth is favored by cutting or girdling competitor trees. Thus, the procedure identifies almost homogeneous small patches within the stand, and depending on their structure, the decision between this three alternatives is made:

Gap opening: Operation for the opening of a gap by felling healthy mature trees and girdling old rotten ones, where a small regeneration patch must grow successfully.

Gap release: Operation oriented to release seedlings patches in old gaps by cutting old over-matures neighbor trees.

Thinning: Release of young healthy saplings or poles (15-30 cm DBH), by cutting competitors trees, mostly from the same cohort.

These general rules can become more specific, taking into account the rainfall level (Figure 3):

Mesic sites (Rainfall > 1100 mm/year)

Stage 0: Patch of mature trees.

i. In a patch composed of mature trees, new gaps with D/H between 1.5 -2 have to be open. We expect that after a rotation of 35 years, the regeneration here will be at least 5 m height.

Stage 1: Gap with seedlings less than 1 m height.

ii. If the available light is not enough for a successful growth, we enlarge the gap up to H/D 2. That allows maximizing the height growth up to 25-30 cm/year.

Stage 2: Gap with seedlings higher than 1 m.

iii. In this case, it is possible that the regeneration losses his form and vigor because of inappropriate light conditions, and the opening of a new gap as in situation 0 is needed. In the other case, with the regeneration in good condition, we enlarge the gap up to D/H = 2. In that case we expected height growth rates similar to that mentioned in situation 1.

iv. If the gap is colonized by seedlings with good growth, there is the possibility that some dominant seedlings with bad form are preventing the proper development of the rest; in this case they should be removed by cutting or girdling.

Stage 3: gap with saplings.

v. In this situation the gaps limits are unclear and it is possible to recognize the good quality saplings or poles with the potential to reach commercial sizes (DBH > 40 cm). They have to be released from their main competitors.

Xeric sites (Rainfall < 1100 mm/year)

Stage 0: Patch of mature trees.

vi. In this situation, where individuals from the canopy have reached an age and size that made them suitable for harvesting, new gaps should be open with D / H between 0.8

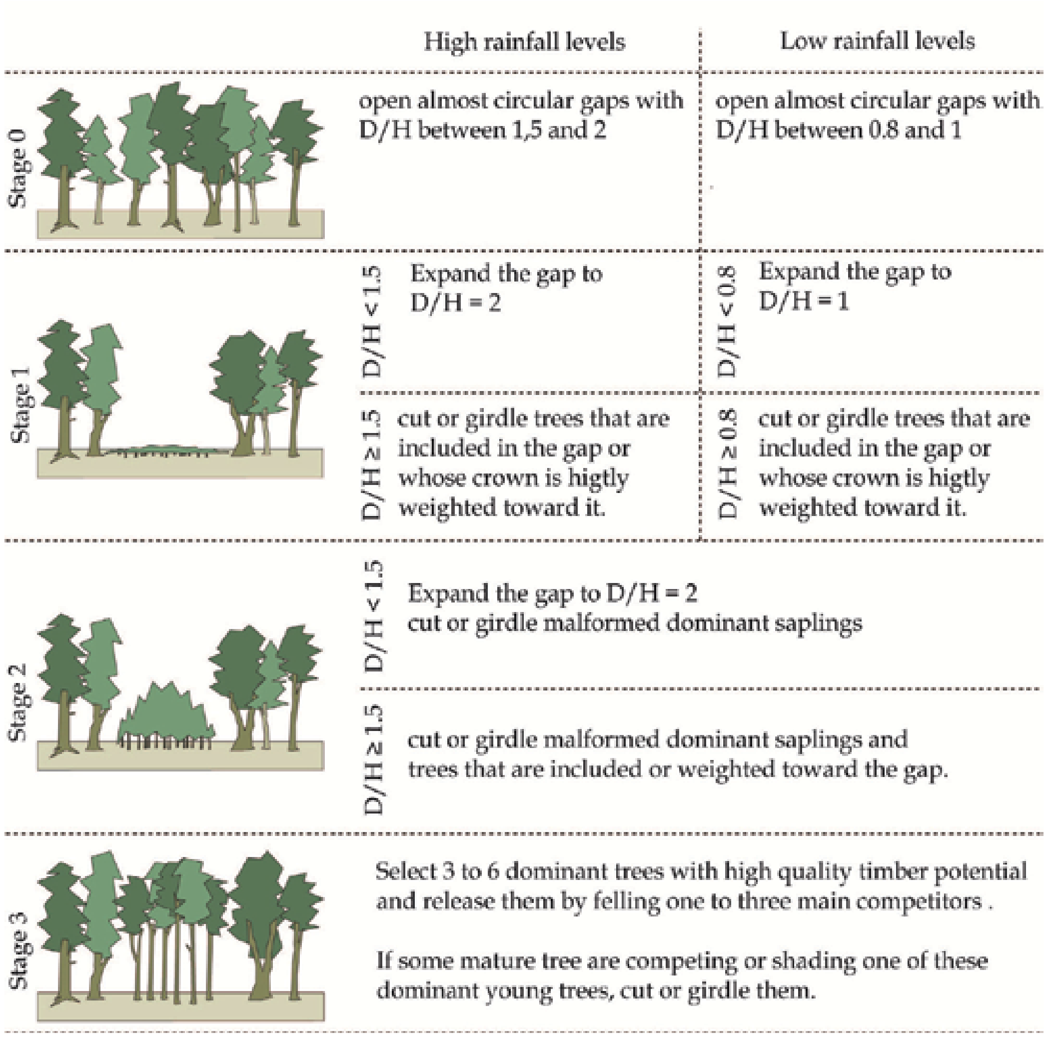

Fig. 3. Schematic representation of the marking procedure.

and 1. Situations where seedlings are already present should be preferred. Thus it is expected that after a short cycle of about 35 years regeneration has reached a height of about 3 m.

Stage 1: gap with seedlings less than 1 m height.

vii. Given this situation, it is recommended to take no action unless you notice the presence of an isolated adult tree stocked in the gap (not as border tree), which should be girdled.

Stage 2: gap with seedlings higher than 1 m.

viii. If the gap has a size from D/H < 1, it should be expanded to reach D/H = 1.5 to 2. The height growth in this condition will reach 15-20 cm/year.

ix. If the gap is colonized by seedlings with good growth, it is the possible that dominant seedlings with bad form are preventing the proper development of the rest; in this case they should be girdled.

Stage 3: gap with saplings.

x. In these cases the gap limits are unclear and individuals have reached a size that allows us to identify those with potential to reach the appropriate size for harvesting (e.g. DBH greater than or equal to 40 cm). They should be released from its major competitors to maximize their growth.

There are three key issues for the success of group selection system in lenga forests. First, regeneration must be installed and growing properly in the gap as to reach their final height with a good stem form. Second, the lateral crown growth of trees bordering the gap must not interfere with the proper development of saplings. Finally, the remaining volume stock after each intervention must maintain its stability until the next harvest. Here we review these three issues, focusing on the aspects that should be taken into account in the definition of guidelines for forest management of lenga by group selection system.

3.3 Regeneration requirements

3.3.1 Regeneration establishment

Several studies in the northern area of distribution of lenga (xeric sites) have concluded that the establishment of the regeneration of this species is strongly dependent on the availability of water during the growing season. In the drier sites located to the east, regeneration is only installed on microsites that, because of being shadier or because of the protection of course woody debris, remain wetter in summer (Heinemann & Kitzberger, 2006, Heinemann et al., 2000, Rusch, 1992). Thus, the position within the gap is a decisive factor for the recruitment of regeneration in drier sites, whereas in moist sites the position does not influence seedling survival. The same authors found that the initial growth of seedlings in the driest sites was greater in the shady parts of the gap, while in wetter sites the initial growth was higher in the center, concluding that moisture and light availability are the limiting factors for recruitment and early growth for sites with lower and higher levels of precipitation, respectively.

Thus, on the sites without drought stress during the summer, as in the western sector of the distribution of lenga in the province of Chubut, larger gaps will be more adequate, in which the interaction between the canopy and the regeneration is lower. By contrast, in sites with a high hydric deficit during the growing season, located east of the distribution of this species, the facilitating effects in microsites protected from direct sunlight and with lower evapotranspiration by the canopy, outweigh the effect of competition for other resources. As a result, smaller gaps will present the highest values of recruitment.

3.3.2 Saplings growth

Having established the regeneration, the requirements for their development change as the seedlings grow in height and their roots explore the soil profile (Callaway & Pugnaire, 2007). Figure 4 shows the values of mean annual increments in height (MAIh) for every level of precipitation and gap size. These values were estimated by a mixed ANCOVA model, in which sapling height was included as a co-variable (López Bernal, unpublished data). During the first 20 years since the gap opening, in the sites with higher levels of precipitation, the dominant seedlings located in the central sector of the gap showed higher growth in larger gaps ($p = 0.03$ and $p = 0.045$ for 0-10 and 10-20 years respectively). By contrast, in sites with lower average annual rainfall, there is a tendency for smaller gaps to show higher height growth, especially during the first 10 years.

Summarizing, we can infer that during the first 20 years since the opening of the gaps, the growth of regeneration is determined by light availability in moist sites and water availability in dry sites, with average values of about 22 cm/year and 15 cm/year, respectively, showing a decrease in the differences due to rainfall with the gap age.

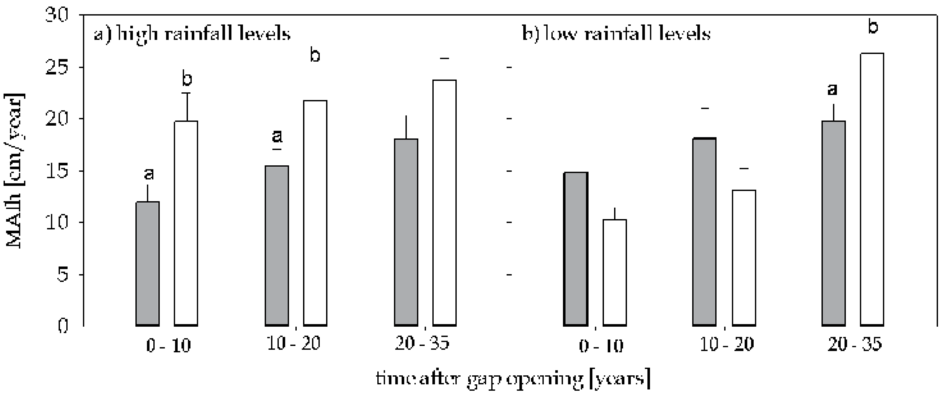

Fig. 4. Mean annual increase in height (MAIh) for each precipitation level and gap size class (small gaps = gray bars, large gaps = white bars) along a 35 years cutting cycle. Different symbols indicates significantly different means (Fisher's posthoc test, α = 0.05).

Moreover, the growth data for gaps between 20 and 35 years old shows that at this stage the saplings grew independently of the availability of water, at least enough to keep differences between the sites with higher and lower levels of precipitation. These observations are consistent with several studies which reported that the balance between facilitation and competition interactions usually tends toward negative values when the "facilitated" individual, approaches the age of maturity (for a comprehensive review of this phenomenon see Callaway & Pugnaire, 2007, pp 240).

3.3.3 Saplings density

Density of seedlings in gaps is often highly variable. During the first years after the creation of gaps, density is strongly determined by the availability of water in the soil, so in places with water deficit during the summer, a greater density is usually observed in the shady gap borders or in microsites caused by the presence of coarse woody debris (Heinemann & Kitzberger, 2006). However, with the subsequent development of the seedlings and the processes of mortality, linked to competence or because of the small disturbances that occur within the gaps (such as total or partial collapse of one of the trees limit), these patterns are lost. For example, it has been observed that in gaps between 20 and 35 years old, significant differences in saplings density between different parts of the gap are not detected (Figure 5, Lopez Bernal et al. Unpublished data). On the other hand, considering only the central part of the gap, there is also great variability, which prevents detect possible influences of gap size or rainfall levels.

3.4 Lateral crown growth of trees bordering the gap

The average closing rate of gaps due to lateral growth of bordering trees is approx. 19 cm/year. This is high enough so that can occur the gap healing before that regeneration can reach the upper stratum (López Bernal et al. unpublished data).

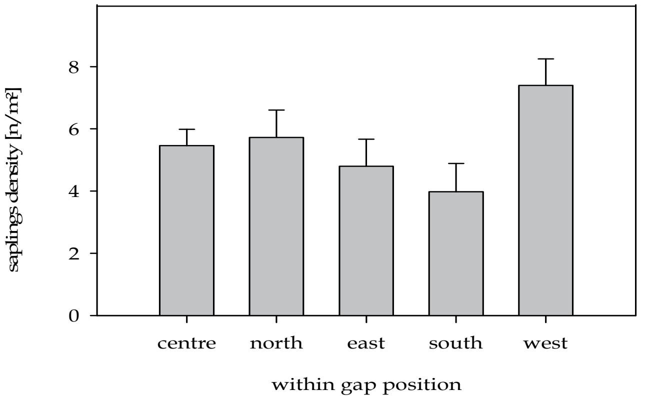

Fig. 5. Average seedling density at different locations within 20-35 years old gaps.

Figure 6 represents the two mechanisms of gap healing (i.e. lateral crown growth of bordering trees and regeneration height growth), indicating the time needed for them to close gaps of different sizes (ordinates). In general, larger gaps require more time for healing by crowns growth and less time for healing by regeneration growth. Thus, the curves representing each mechanism are cut at the point corresponding to the gap size that allows the regeneration to reach the canopy just before the crown growth of bordering trees prevents it. It can also be inferred how long will it take for this to happen (abscissa).
Thus, the ① arrow represents the development of a gap in a humid stand, where it is feasible to open a gap with D/H between 1.5 and 2, favoring the seedlings installation and saplings development until its final height. Moreover, the ② and ③ arrows represent the development of a gap in a xeric stand, where it is necessary to open smaller gaps to ensure seedling establishment, but after a 35 years cutting cycle is necessary to enlarge the gap to prevent the healing by the lateral crown growth of bordering trees.

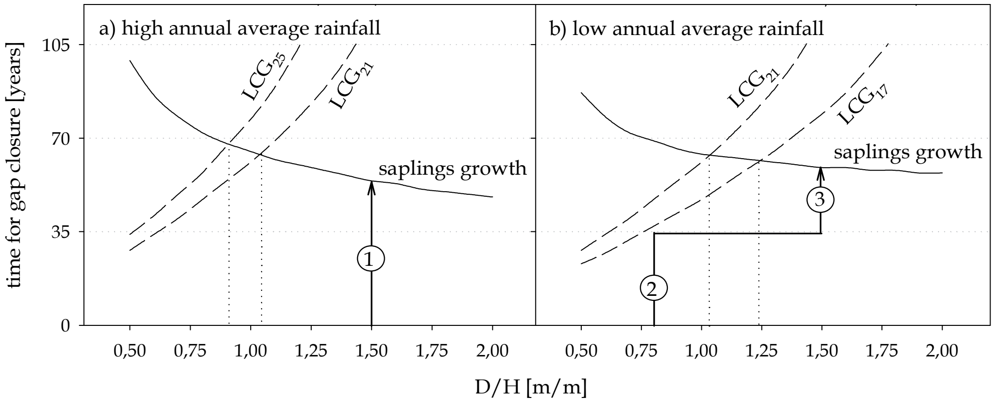

Fig. 6. Necessary time to close gaps of different sizes (D/H) through the height growth of regeneration (solid lines) or the lateral crown growth of the bordering trees (dashed lines) at sites with different dominant height ($LCG_{17, 21 \& 25}$). For references of the arrows ①, ② and ③ see above.

3.5 Adaptability of GSS to *N. pumilio* natural dynamics

Managing an uneven-aged forest through selection cuts implies a continuous production of wood, so that the remaining stand becomes very important. The regeneration which is established after each harvest and the remaining young trees with timber potential will be the wood source in the coming rotation cycles, so that they constitute the basis for the system´s sustainability (Antequera et al., 1999). That is why post-harvest mortality is a factor of utmost importance.

The harvested stands are affected in their stability, according to the original structure, topography and the type of intervention (Burschel & Huss, 1997, Smith et al., 1997). This weakening effect leads to the fall of trees after the harvest, phenomenon that can seriously affect the quality of the remnant stand. In Tierra del Fuego the windfalls occur even in virgin forests (Rebertus et al., 1997), which poses a logical doubt on the real possibility of implementing this system.

Bava & López Bernal (unpublished data) found that there is no relationship between the manner in which a tree dies (uproot, break or standing death) and the harvest intensity, site quality or stand structure. However, a higher percentage of uprooted trees were observed. The stems that break down correspond to well-anchored individuals, when the wind burden cannot be transmitted by the trunk to the root and soil (Abetz, 1991), or to trees affected by rots, as frequently happens in lenga forests. The uprooting happens when the wind burden is transmitted to the root but cannot be transmitted to the soil (Abetz, 1991). In lenga forests of Tierra del Fuego this can occur in shallow soil stands, when the root system grows superficially (Bava, 1999).

The post-harvest mortality is not significantly related with the percentage of extracted BA. However, when we compare between different stand structures, we note that uneven-aged forests presented minor damage to the even-aged, while the bi-stratified stands presented intermediate damages (Figure 7, ANOVA $p = 0.014$). These differences may have their origin in phenomena observed at two separate scales: in a stand-scale, uneven-aged forests present a more gradual decline of wind speed from the forest canopy up to the understory, allowing a better adaptation mechanics of trees to wind and giving more stability to the whole (Gardiner, 1995). On the other hand, at the individual-level, Wood (Wood, 1995) observed that the tree develops stems only with the resistance needed to support regular wind intensities, growing adaptively. In this way, the increased heterogeneity of uneven-aged stands would provide more opportunities for development of more resistant individuals, which remain after the harvest, and that play a very important role in the stand stability (Burschel & Huss, 1997, Mattheck et al., 1995, Smith et al., 1997). The structural alterations produced by the harvest causes greater exposure of individuals to wind, but in a different way for each one, and would depend on other factors besides the size of the gaps, the h/d value, the felling damages, and homogeneity of the remnant forest.

We have mentioned the importance of the forest stability for sustainability in a selection cuts system, where the productive potential for future interventions is represented by individuals which remain after harvest. In this sense, the results indicate that the post-harvest losses are a limiting factor for the implementation of this system, and which would only be advisable by uneven-aged forests. Moreover, the system success also depends on the conscientiously choosing of the trees to cut, and to carry out the harvest operations carefully. If these conditions are present, the group selection system would be a viable alternative, which would maintain the forest cover, with a cutting cycle of approximately 35 years and extracting a timber volume equivalent to the historical average.

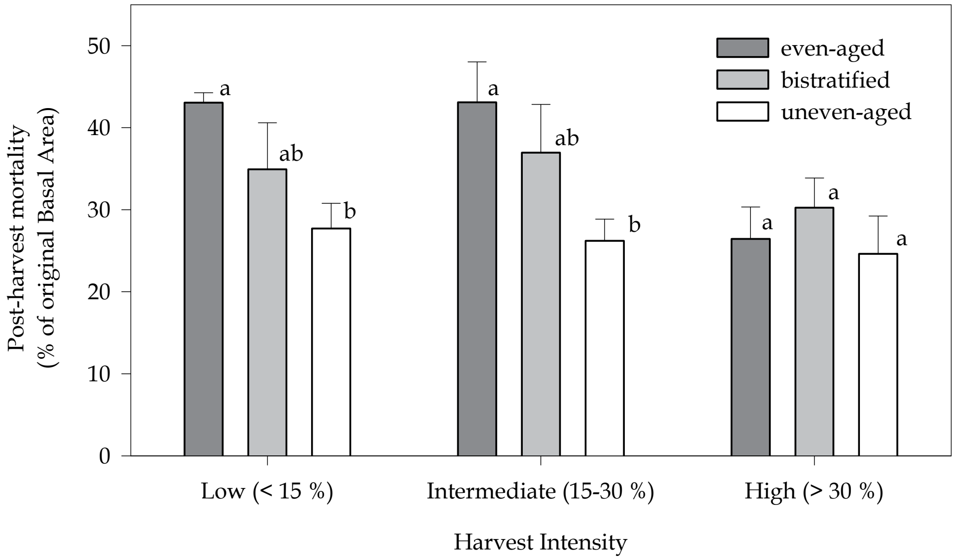

Fig. 7. Post-harvest mortality by harvest intensity and original stand structure. Different symbols indicate significantly different means (Tuckey's posthoc test, α = 0.05).

3.6 Case study

In this section we present the main results of three trials located in the province of Tierra del Fuego where group selection cut were applied (Bava & López Bernal, 2006). These were implemented in uneven-aged stands with trees from at least three generations and where it was possible to identify the natural process of gap dynamics.

The tree marking was made in November 2003 and the harvest in February 2004, which consisted of felling and bucking of complete stem. During the tree marking, DBH, height and average sawing bole diameter of all marked trees was recorded. At the same time, it was recorded if the tree was felling to open a new gap, to release existing regeneration, or to optimize the growth of young trees. The felling, skid trails opening and bole extraction were carried out in the same campaign. In all three essays harvest tasks were performed by the same team, using directional felling techniques for tree felling and a skidder for bole extraction.

After the tree marking, a forest inventory was carried out in each of the three trials. Measurements were performed in 300 m^2 circular plots spread over a 50 m x 50 m grid, representing a sampling intensity of 1.2 %. In each plot, the DBH of all individuals over 10 cm was measured, recording their sawing potential (indicating the length and medium diameter of the logging portion of the bole), and if it had been marked, whether for felling or girdling.

All three trials represented intermediate quality sites, located on gentle slopes and possessing uneven-aged structures. The trial 1 had about 360 tree per ha, a BA of 44 m^2/ha and a high proportion of overmature trees (DBH over 60 cm) with a low sawing quality. Essay 2 had 430 trees per ha, a BA of 49 m^2/ha and presents a high proportion of trees with a DBH between 40 and 60 cm. Essay 3 had 498 trees per ha, a BA of 52 m^2/ha with a high proportion of trees with DBH between 30 and 50 cm (Figure 8).

Timber stock differences between trials derived in great differences on tree marking. The marking intensity, expressed as a percentage of the original AB, was considerably higher in trial 3 than in trial 1 and 2, proportionally to the differences in timber stock. Moreover, differences in the stand structures generated varying amounts of felled and girdled trees (Table 1).

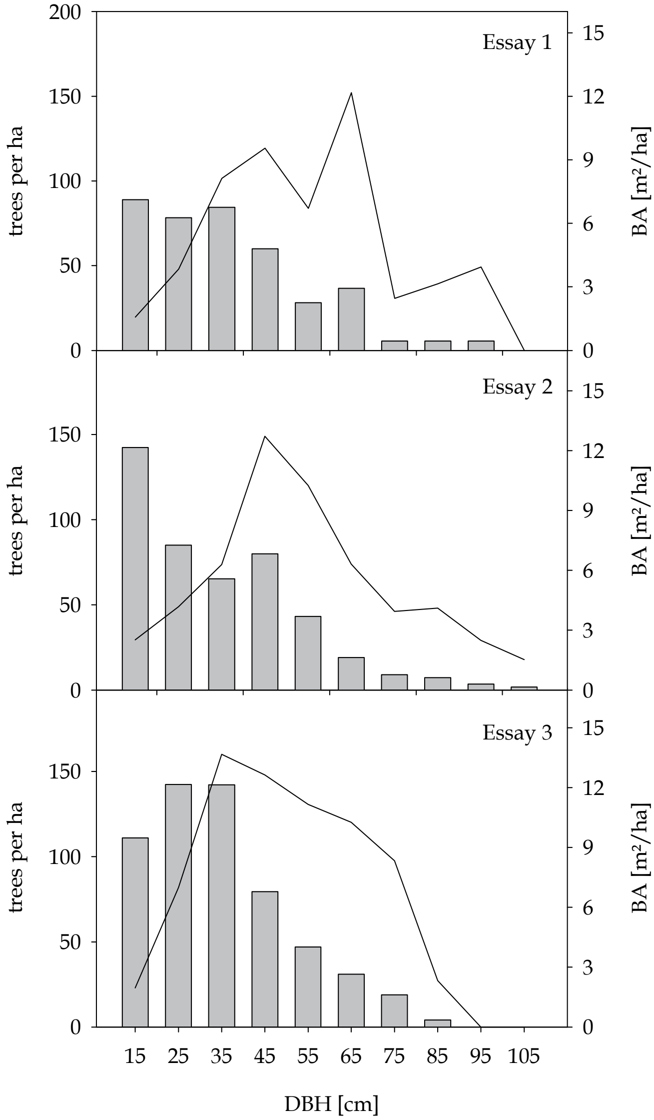

Fig. 8. Diametric frequency distribution for each trial.

The number and size of gaps or patches that were intervened were also different. In the first two trials, which showed similar productions, about 11 gaps per ha were opened by felling or girdling between 2.5 and 3 trees. In trial 3, with a much higher timber production, the number of opened gaps was also bigger, mainly due to a high proportion of patches with young trees (DBH between 20 and 40 cm), while the number of trees per gap increased to 3.4 (Table 2). Moreover, the proportion of gaps or patches with *gap opening*, *gap release* or *patch thinning* interventions differ between essays, pointing out differences in the original stand structures.

Although the three trials were conducted in similar structures, there were significant differences (up to 100%) in the amount of lumber in each. This was reflected in the number of gaps per hectare, but not in their size. The trial with highest harvest intensity (28% of BA) produced twice as sawtimber than the other two, mainly due to felling tending to release young pole trees. This is different from harvests in Chubut province, where the largest volume portion comes from gap opening cuts (Berón et al., 2003).

	Trees (N/ha)			Basal area (m^2/ha)		
Essay	felling	girdling	Total	felling	girdling	Total
1	28.0 (87%)	4.0 (13%)	32.0	4.8 (96%)	0.2 (4%)	5.0
2	24.4 (86%)	3.8 (14%)	28.2	4.6 (85%)	0.8 (15%)	5.4
3	58.5 (75%)	19.1 (25%)	77.6	11.8 (81%)	2.8 (19%)	14.6
Mean	37.0 (81%)	9.0 (19%)	45.9	7.1 (86%)	1.3 (14%)	8.3

Table 2. Number and proportion of trees and AB marked in each essay, distinguishing between felling and girdling.

Intervention objective	Essay 1	Essay 2	Essay 3	Mean
Gap opening (N/ha)	2,0	7,6	7,3	6,7
Gap release (N/ha)	1,3	0,4	5,2	2,7
Patch thinning (N/ha)	7,3	3,2	10,2	6,9
Total gaps / patches per ha	10,7	11,2	22,8	16,4
Felled trees per gap	2,6	2,2	2,6	2,5
Girdled trees per gap	0,4	0,3	0,8	0,7
Total	3,0	2,5	3,4	3,1

Table 3. Number of interventions for gap opening, gap release or patch thinning per ha, and number of marked trees per gap in each essay.

According to the remnant structures after harvesting, all three trials are able to recover the volume of extracted timber. However, the best choices to implement a group selection system are stands like in trial 3, i.e. a forest with uneven-aged structure and with a high proportion of trees with DBH between 30 and 50 cm. These structures allow a higher proportion of "gap release" and "patch thinning" interventions, which generates a bigger timber harvest in the first cycle, leaving a high number of young trees in optimal growth conditions. The harvest intensity of this trial is very similar to the historical average for Tierra del Fuego province, at about 27% of BA (Bava & López Bernal, 2004), while is much higher than the historical mean for the province of Chubut, of about 15% (Berón, et al., 2003).

4. Conclusion

The Group Selection System is a valid alternative management system for lenga forests of Argentinean Patagonia. This system emulates one of the most common natural dynamic processes in these forests and provides optimal conditions for regeneration establishment and further development. It is especially recommended for sites with medium to low rainfall levels, where the frequency of large-scale disturbances is low and where the forest presents a natural uneven-aged structure. In Argentina, these situations mainly occur in Chubut province and in the northern part of the lenga distribution in Tierra del Fuego province, where there are already experiences with this type of management.
Moreover, the GSS is compatible with the local production system, dominated by small and medium producers, without financial or technological capacity to afford the costs of intensive harvesting or long-term silvicultural investments. The GSS is adapted to these systems by splitting the turnover age in shorter cutting cycles, giving a more flexible cash flow to these systems, and by allowing that in a single intervention, different silvicultural practices can be carried out. This last point is also an advantage for state control agencies by allowing them to condition the timber extraction to the implementation of other practices that do not generate immediate benefits, such as thinning or regeneration release.
Finally, to ensure the sustainability of forests managed by the GSS, there are at least two aspects that should be especially considered. The first one is that the forester must make his proper interpretation of the natural forest dynamics to decide whether it is feasible or not the implementation this system. The second one implies that to maintain the productive potential for future interventions, logging activities should be conducted with special attention to the remaining forest, using low-impact harvesting technologies.

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4

Remote Monitoring for Forest Management in the Brazilian Amazon

André Monteiro and Carlos Souza Jr.
Amazon Institute of People and The Environment-Imazon
Brazil

1. Introduction

Timber harvesting is an important economic activity in the Brazilian Amazon. In 2009, the timber industry produced 5.8 million cubic meters of logwood and generated US$ 2.5 billion in gross income along with 203,705 direct and indirect jobs (Pereira et al., 2010). Logging in the region is predominantly predatory, and is commonly known as Conventional. Only a small proportion occurs in a managed fashion (planned), known as Reduced Impact Logging (RIL) (Asner et al., 2002; Gerwing, 2002; Pereira Jr. et al., 2002; Veríssimo et al., 1992). In the conventional method activities are not planned (opening of roads and log decks[1], tree felling and log skidding), while with RIL planned management techniques are applied at all stages of harvesting (Amaral et al., 1998).

The two methods cause impacts ranging from low to severe on the structure and composition of the remaining forest (Gerwing, 2002; John et al., 1996; Pereira Jr. et al., 2002). However, the impacts of predatory logging are two times greater than those of managed logging (John et al., 1996). Among the main impacts are: greater reduction in living aboveground biomass (Gerwing, 2002; Monteiro et al., 2004), risk of extinction for high-value timber species (Martini et al., 1994), greater susceptibility to forest fires (Holdsworth & Uhl, 1997), increase of vines and pioneer vegetation (Gerwing, 2002; Monteiro et al., 2004) and substantial reduction in carbon stocks (Gerwing, 2002; Putz et al., 2008).

The impact of timber harvesting can be described by means of forest inventories carried out in the field, with which it is possible to evaluate the structure and composition of the remaining forest (Gerwing, 2002; John et al., 1996; Monteiro et al., 2004). Another method employed is remote sensing, which has advanced over the last decade. In the Amazon, there have been successful tests with satellite images to detect and quantify forest degradation brought about by logging activities in the region (Asner et al., 2005; Matricardi et al., 2007; Souza Jr. et al., 2005). Images with moderate spatial resolution, such as Landsat (30 m) and Spot (20 m), have been used to detect types of logging, damages to the canopy and roads and log decks for harvesting (Asner et al., 2002; Matricardi et al., 2007; Souza Jr. & Roberts, 2005). As for images with high spatial resolution, such as Ikonos (1 to 4 m), they are capable of detecting smaller features of logging, such as small clearings (Read et al., 2003), as well as making it possible to determine the size of log decks and width of roads (Monteiro et al.,

[1] Clearings (500 m^2) opened in the forest for storing timber.

2007). The use of remote sensing for monitoring forest management plans is of great importance for the Brazilian Amazon, given that logging activities are predominantly predatory and occur in extensive areas that are difficult to access.

Recent studies have shown how to integrate data extracted from satellite images with biomass data collected in the field, which makes it possible to estimate the loss of biomass in the forest submitted to different levels of forest degradation (Asner et al., 2002; Pereira Jr. et al., 2002; Souza Jr. et al., 2009). Our research has made advances in applying those techniques to assess the intensity and quality of logging (Monteiro et al., 2009).

In this chapter, we demonstrate how the impacts of timber harvesting can be characterized by means of forest inventories and combined with satellite images to monitor extensive areas. We also present the remote sensing techniques utilized for detecting, mapping and monitoring logging activities. Finally, we present the results of our system for monitoring forest management plans, applied in Pará and Mato Grosso, the two largest timber-producing States in the Amazon, which respectively account for 44% and 34% of the total produced in 2009 (Pereira et al., 2010).

2. Logging impact characterization based on field surveys

2.1 Change in structure and composition as a result of forest degradation

Characterization of the impacts of logging in the field is done by means of forest inventories. To do this, transects or plots are established in the forest to quantify the damages to its structure and composition in terms of soil cover, canopy cover and aboveground live biomass (Gerwing, 2002; John et al., 1996; Monteiro et al., 2004).

In the method developed by Gerwing (2002) 10 m x 500 m transects are opened, in which all individual trees with DBH (Diameter at Breast Height) ≥ 10 cm are sampled. In 10 m x 10 m sub-parcels, located at 50 m intervals along the central line of the transect, all individuals with DBH ≤ 10 cm are sampled. In those sub-parcels the soil cover is assessed, with the percentages of intact soil, soil with residues and disturbed soils being recorded; as well as the canopy cover, with four readings in a spherical densiometer, at 90° intervals, every 50 m along the central axis of the transect (Figure 1).

Additionally, the live biomass above the ground in each transect is estimated, adding together the weight of dry matter from different forest components using alometric equations available in the literature (Table 1). The estimate of biomass for trees < 10 cm is done by multiplying the number of stems in each diameter class by the biomass corresponding to the arithmetic average of the diameter for each class.

The forest inventory was carried out in 55 transects, including 11 in intact forest (reference) and 44 in forests in different classes of degradation due to different log harvesting methods. It was done in the Paragominas and Santarém regions, in the State of Pará, in Sinop, in Mato Grosso, and in Itacoatiara, in Amazonas (Figure 2). Below is a description of intact forest and forests in different classes of degradation according to Gerwing (2002):

i. Intact forest: mature forest (> 40 years) without disturbance, dominated by shade-tolerant species.
ii. Logged without mechanization (Traditional logging): forest logged without the use of skidder tractors, that is, without impact from construction of logging infrastructure: log decks, roads and skidder trails.
iii. Managed logging (Reduced Impact Logging-RIL): forest logged selectively following planning of harvesting activities: forest inventory, opening of decks and roads, felling and skidding of trees and transport of logs.

iv. Conventional logging: forest logged selectively and not following planning of the activities mentioned above. Log decks, roads and skidder trails are opened causing severe damage to the forest.
v. Logged and burned: forest logged selectively, without planning, followed by burning.
vi. Logged intensely and burned: forest logged selectively in a conventional manner more than once, and later burned.
vii. Burned: forest burned without having been logged.

To evaluate differences between the variables in the degradation classes we employed the analysis of variance (ANOVA with Type III Sum of Squares) followed by the Tukey's HSD post-hoc test with an individual error rate of 0.05% and with an overall significance of 0.08% using the R program (R *Development Core Team*, 2010).

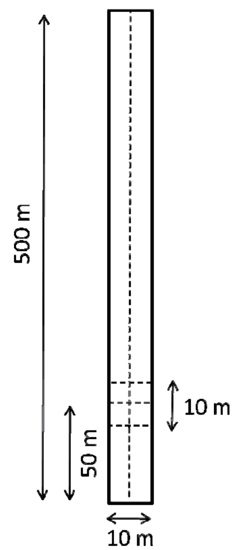

Fig. 1. Transect layout of the forest inventories.

Species group	Regression equation	Source
Forest tree species		
≥ 10 cm DBH	$DW= 0.465(DBH)^{2.202}$	Overman et al. (1994)
	$DW= 0.6*4.06(DBH)^{1.76}$	Higuchi & Carvalho (1994)
< 10 cm DBH	log(DW)= 0.85+2.57 log(DBH)	J. Gerwing (data not published)
Pioneer tree species		
Cecropia sp.	ln(DW)= -2.512+2.426 ln(DBH)	Nelson et al. (1999)
Other sp.	ln(DW)= -1.997+2.413 ln(DBH)	Nelson et al. (1999)

Table 1. Regression equations used to determine the dry weights (kg) of various forest components based on their diameters (cm).

In the subsections below, we present the results of characterizing forest degradation for the classes described above. That information is later combined with remote sensing data to evaluate the intensity and quality of logging. In the field we quantified forest degradation related to soil disturbance (intact vegetation, residues and disturbed soil), canopy cover and aboveground biomass because those indicators present a direct relation with remote sensing data (Souza Jr. et al., 2009).

Fig. 2. Location of the forest transects sites.

2.1.1 Soil disturbance and canopy cover

The evaluation of soil disturbance (intact vegetation, residues and disturbed soil) and canopy cover in the field is crucially important, since those results directly influence the results of the satellite images. The greater the soil disturbance and the smaller the canopy cover of the degraded forest, the greater will be the signal for this damage in the image. Our results show that the area of intact vegetation was smaller in the classes with greater degradation. The smallest percentage of intact vegetation was observed in the intensely logged and burned class (4%), followed by burned forest (22%), with these presenting a significant difference in relation the intact forest and to the classes with less degradation. The quantity of residues in the soil was greater in the logged and burned forest (26%), followed by the burned forest (25%), however no significant differences were found between these classes and intact forest. The area of disturbed forest was greater in the intensely logged and burned forest (96%) and the burned forest (53%), presenting a significant difference in relation to the intact forest and the other degradation classes. The logged and burned class presented the lowest canopy cover (75%), with a significant difference in relation to the intact forest and to the classes with less degradation (Table 2).

2.1.2 Change in the live aboveground biomass

The live biomass aboveground was less in the forest degradation classes compared to the biomass in intact forest; however, no significant differences were found between them. Among individuals with DBH ≥ 10 cm, the logged and burned and intensely logged classes presented 36% lower biomass than the intact forest, followed by the burned class (18%) (Table 2). The lowest biomass for individuals with DBH < 10 cm was also observed in the logged and burned (44%) and intensely logged and burned (11%) classes (Table 2). The biomass in individuals with DBH ≥ 10 cm decreased with increasing degradation. The variation in biomass for individuals with DBH < 10 cm seems to be related to the incidence of pioneer species that tolerate moderate levels of degradation (Gerwing, 2002; Monteiro et al., 2004). The greatest loss of biomass is not related only to the greatest forest degradation. The distance from the first degradation may also influence a greater reduction in biomass (Gerwing, 2002; Monteiro et al., 2004). For example, data collection in the logged and burned forest (biomass for individuals ≥ 10 cm and < 10 cm = 232 t ha^{-1} and 5 t ha^{-1}, respectively) occurred approximately 2.5 years after the first degradation event, while in the intensely logged and burned forest (biomass for individuals ≥ 10 cm and < 10 cm = 234 t ha^{-1} and 8 t ha^{-1}, respectively), it occurred 16 years after the first degradation event.

	(a) Intact (n=11)	(b) Non-mechanized logging (n=9)	(c) Managed logging (n=14)	(d) Conventional logging (n=8)	(e) Logged and burned (n=4)	(f) Heavily logged e burned (n=5)	(g) Burned (n=4)
Ground cover (total area (%))							
Intact vegetation	95 (6) a	87 (22) b	96 (20) c	92 (15) ac	75 (7) d	4 (5) ef	22 (19) f
Woody debris	5 (6) a	23 (23) a	23 (12) a	13 (11) a	26 (10) a	0 a	25 (44) a
Disturbed soil	0 a	3 (4) a	10 (9) a	13 (11) a	7 (6) a	96 (5) b	53 (49) c
Canopy cover (%)	95 (3) a	87 (9) b	96 (2) a	92 (6) a	75 (8) c	86 (2) a	88 (1) d
Aboveground live biomass (t ha^{-1})							
Live trees ≥ 10 cm DBH	365 (50) a	347 (33) a	342 (52) a	321 (12) a	232 (11) a	234 (23) a	299 (14) a
Live trees < 10 cm DBH	9 (2) a	10 (2) a	9 (1) a	11 (1) a	5 a	8 a	9 a

* Means presented with standard deviation noted parenthetically. In the ground cover, canopy cover and biomass values, different forest class letters denote significant differences among stand classes at P<0.05 utilizing Tukey's HSD post-hoc test, with a global significance level of 0.8

Table 2. Comparison of ground cover, canopy cover and biomass among intact forest and degraded forest in the States of Pará, Mato Grosso and Amazonas in Brazil*

3. Remote sensing techniques to enhance and detect timber harvesting

Moderate satellite imagery such as Landsat Thematic Mapper (30-meters pixel size) and Spot Multispectral (20 meters) has been used to detect and map the impacts of selective logging. However, the complex mixture of dead and live vegetation, shadowing and soils found throughout forest environments impose challenges to revealing these impacts, requiring advanced remote sensing techniques (Asner et al., 2005; Souza Jr. et al., 2005).

From the satellite vantage point, forest damage caused by logging seems to disappear within three years or less, making detection of previously logged forest (> 1 year) very challenging (Souza Jr. et al. 2009; Stone & Lefebvre, 1998). Remote sensing studies on logging in the Brazilian Amazon found that Landsat reflectance data have high spectral ambiguity for distinguishing logged forest from intact forest (Asner et al., 2002, Souza Jr. et al., 2005). Vegetation indices (Souza et al. 2005a; Stone & Lefebvre, 1998) and texture filters (Asner et al., 2002) also showed a limited capability for detecting logging. Improving the spatial resolution of reflectance data can help; 1-4 m resolution Ikonos satellite data can readily detect forest canopy structure and canopy damage caused by selective logging (Asner et al. 2002; Read et al., 2003; Souza Jr. & Roberts, 2005). However, the high cost of these images, and additional computational challenges in extracting information, requiring a combination of object-oriented classification with spectral information, severely limit the operational use of Ikonos and similar imagery.

Over the last two decades, the Brazilian Amazon has been a great laboratory for testing remote sensing techniques to detect and map forest impacts of selective logging (Asner et al., 2005; Matricardi et al., 2001; Read et al., 2003; Souza Jr. & Barreto, 2000; Souza Jr. et al., 2005; Stone & Lefebvre, 1998;). These techniques differ in terms of mapping objectives, image processing techniques, geographic extent, and overall accuracy. In terms of mapping objective, some image processing algorithms were proposed for the total logged area, including roads, log landings, forest canopy damaged and undisturbed forest islands, while others were focused only on the mapping of forest canopy damage. Techniques to map total logged area were based on visual interpretation (e.g., Matricardi et al., 2001; Stone & Lefebvre, 1998), combination of automated detections of log landings with buffer applications defined by logging extraction reach (Monteiro et al., 2003; Souza Jr. & Barreto, 2000), and textural filtering (Matricardi et al, 2007). More automated techniques are mostly based on SMA (spectral mixture analysis) approaches combined with spatial pattern recognition algorithms (Asner et al., 2005; Souza Jr. et al., 2005). Finally, image segmentation has been applied to very high spatial resolution imagery (Hurtt et al., 2003). Landsat images are the ones most used in the studies and in operational systems in the Brazilian Amazon.

Some research has shown that the detection of logging at moderate spatial resolution is best accomplished at the sub-pixel scale using SMA (Box 1). Images obtained with SMA that show detailed fractional cover of soils, non-photosynthetic vegetation (NPV) and green vegetation (GV) enhance our ability to detect logging infrastructure and canopy damage. For example, log landings and logging roads have higher levels of exposed bare soil with detection facilitated by Soil Fraction (Souza Jr. & Barreto, 2000). The brown vegetation component, including trunks and tree branches, increases with canopy damage, making NPV fraction useful for detecting this type of area (Souza Jr. et al., 2003; Cochrane & Souza Jr., 1998) and the green vegetation (GV) fraction is sensitive to canopy gaps (Asner et al., 2004).

A novel spectral index combining the information from these fractions, the Normalized Difference Fraction Index (NDFI) (Souza, Jr. et al., 2005), was developed to augment the detection of logging impacts. NDFI is computed as:

$$\mathrm{NDFI} = \frac{\mathrm{GV}_{\mathrm{Shade}} - (\mathrm{NPV} + \mathrm{Soil})}{\mathrm{GV}_{\mathrm{Shade}} + \mathrm{NPV} + \mathrm{Soil}} \tag{1}$$

where GV_{shade} is the shade-normalized GV fraction given by,

$$GV_{Shade} = \frac{GV}{100 - Shade} \tag{2}$$

NDFI values range from -1 to 1. For intact forests, NDFI values are expected to be high (i.e., about 1) due to the combination of high GV shade (i.e., high GV and canopy Shade) and low NPV and Soil values. As forest becomes degraded, the NPV and Soil fractions are expected to increase, lowering NDFI values relative to intact forest (Souza Jr. et al., 2005). Canopy damage detection caused by forest degradation induced by factors such as logging and forest fires can be detected with Landsat images within a year of the degradation event with 90.4% overall accuracy (i.e., for three land cover classes, Non-Forest, Forest and Canopy Damage) (Souza Jr. et al., 2005).

The reflectance data obtained from Landsat data of each pixel can be decomposed into endmember fractions, which are purest component materials that are expected to be found within the image pixels. For the purpose of detecting forest degradation, we modeled the reflectance pixel in terms of GV (green vegetation), NPV (non-photosynthetic vegetation), Soil and Shade through Spectral Mixture Analysis – SMA (Adams et al., 1993). The SMA model assumes that the image spectra are formed by a linear combination of n pure spectra, such that:

$$R_b = \sum_{i=1}^{n} Fi\, R_{i,b} + \varepsilon_b \tag{1}$$

for

$$\sum_{i=1}^{n} F_i = 1 \tag{2}$$

where R_b is the reflectance in band b, $R_{i,b}$ is the reflectance for endmember i, in band b, F_i the fraction of endmember i, and ε_b is the residual error for each band. The SMA model error is estimated for each image pixel by computing the **RMS** error, given by:

$$RMS = [n^{-1} \sum_{b=1}^{n} \varepsilon_b]^{1/2} \tag{3}$$

Identifying the correct endmembers is a crucial step in SMA model. To avoid subjectiveness in this process, we have built a generic endmember spectral library (Figure 3) as described in Souza Jr. et al. (2005).

The following steps are used to evaluate SMA results :

1. Fraction images are evaluated and interpreted in terms of field context and spatial distribution. For example, high Soil fraction values are expected in roads and log landings and high NPV in forest areas with canopy damage;
2. Fraction values should have physically meaningful results (i.e., fractions ranging from zero to 100%). Histogram analysis of fraction values can be performed to evaluate this requirement.

3. Fraction values must be consistent over time for invariant targets, i.e., that intact forest not subject to phenological changes must have similar values over time.

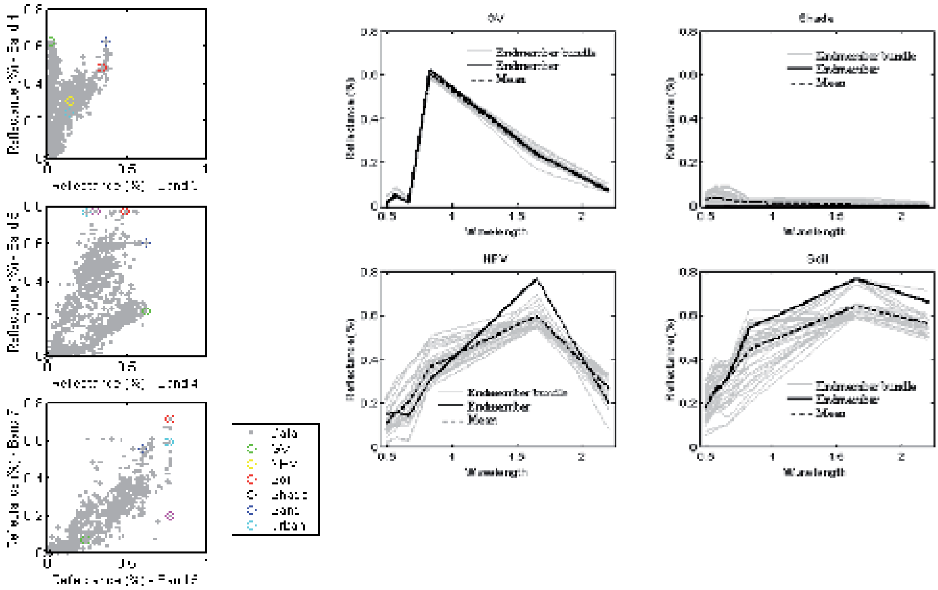

Fig. 3. Image scatter-plots of Landsat bands in reflectance space and the spectral curves of GV, Shade, NPV and Soil (source: Souza Jr. et al., 2005).

Box 1. Spectral Mixture Analysis (SMA)

4. Integrating field and remote sensing data

Assessment of the quality of timber harvesting has traditionally been done through measuring damages to the forest, e.g. quantification of the opening of log decks, logging roads and openings resulting from felling trees; and the density and biomass for remaining individuals (Gerwing, 2002; Pereira Jr. et al., 2002; Veríssimo et al., 1992). However, field surveys are expensive and lengthy, especially for extensive areas such as the Amazon. Recent studies have shown that it is possible to infer the quality of timber harvesting through satellite images that are calibrated with indicators of damages measured in the field, allowing greater speed, reduction of costs and monitoring of extensive areas. Using satellite images such as Landsat and Spot it is possible to evaluate the quality of logging activities based on mapping of roads, log decks and damages to the forest canopy (Monteiro & Souza Jr., 2006; Monteiro et al., 2009). We present below the items evaluated and the respective indicators for monitoring timber harvesting and the results of its application in order to qualify its impacts.

4.1 Roads and log decks

For the roads and log decks we evaluated the following indicators: the density of log decks and roads; the distance between secondary roads and between log decks; and spatial distribution of log decks and roads. Those indicators were tested in 43 logging areas located in regions of Pará, Mato Grosso and Amazonas. The results were validated with measurements of the same indicators in the field, in areas of conventional (predatory) logging and managed logging in the Paragominas (PA) and Sinop (MT) regions (Monteiro & Souza Jr., 2006).

To do this we used Landsat 5 TM satellite images with 30 meters of spatial resolution. We first applied geometric and atmospheric correction to those images. Next, we obtained fraction images of vegetation, soils and NPV (non-photosynthetically active vegetation), based on the spectral mixture model followed by NDFI (Normalized Difference Fraction Image) to highlight the scars caused by logging (Souza Jr. et al., 2005). Finally, we digitalized the log decks and roads in the NDFI image and inferred the density of log decks and roads and the distances between log decks and between roads. Additionally, we classified the spatial distribution of log decks and roads as systematic and non-systematic. Systematic distribution is characterized by rectilinear and parallel roads and log decks regularly distributed along the roads, while non-systematic distribution is defined by sinuous roads with log decks interlinked by their segments.

The results of evaluating the indicators presented an average density of 16 meters/hectare for roads and 3/100 hectares for log decks. The average distance between roads was 623 meters, and between log decks it was 484 meters (Table 3). Conventional logging presented a higher density of log decks and roads compared to logging with forest management (Johns et al., 1996). As for the distance between secondary roads and between log decks, they are smaller in logging with forest management compared to the distances between secondary roads in conventional logging (Monteiro, 2005).

As for the spatial distribution of log decks and roads, the majority of areas evaluated that were logged using forest management presented a non-systematic distribution of log decks (60%) and roads (58%), which indicates low quality in planning that infrastructure (Monteiro & Souza Jr., 2006).

Region		**Logging type**	**Density***		**Distance***	
			Road (m/ha)	Log deck (n/100 ha)	Secondary roads (m)	Log landing (m)
IMAGE	**Pará, Mato Grosso and Amazonas**	Managed	16 (5)	3 (2)	623 (232)	484 (148)
FIELD	**Paragominas (PA)**	Managed	23 (4)	8 (1)	469 (30)	260 (74)
		Conventional	36 (6)	15 (2)	513 (38)	301 (263)
	Sinop (MT)	Managed	32 (11)	7 (4)	455 (24)	347 (126)
		Conventional	19 (3)	1 (2)	508 (43)	512 (44)

*Mean density and distance with standard deviation within brackets.

Table 3. Comparison between the indicators (mean) measured in the images from Pará, Mato Grosso and Amazonas regions and those measures in the field in Paragominas and Sinop (Monteiro & Souza Jr., 2006).

4.2 Forest canopy

We evaluated the indicator of damages to the canopy caused by logging operations. To do that we utilized NDFI images to evaluate the area of forest affected. First, we delimited the logging area visible in the NDFI image by means of visual interpretation. Next, we selected around five samples from 100 in the NDFI image to represent logging and extracted the average values of those samples. The samples were composed of a mosaic of environments (forest, log decks, roads, skidder trails and clearings caused by felled trees).

The quality of logging is determined using thresholds obtained in the NDFI image and calibrated using field data (Monteiro et al., 2008), so that: NDFI≤ 0.84 represents low quality timber harvesting (predatory logging); NDFI= 0.85-0.89, intermediate quality harvesting (there was an attempt at adopting management, but the configuration of roads, log decks and clearings reveals serious problems with execution); and NDFI≥ 0,90, good quality harvesting (the configuration of roads, log decks and clearings is in conformity with the techniques recommended by forest management (Figure 4).

This method was tested in the States of Pará and Mato Grosso, the main timber producers, responsible respectively for 44% and 34% of the total produced in 2009 in the Brazilian Amazon (Pereira et al., 2010). We evaluated 156,731 and 177,625 hectares respectively of areas undergoing timber harvesting in the two States. In Pará, 21% of that total presented logging of good quality, 54% showed intermediate quality and 25% showed low quality (Figure 5). In Mato Grosso, only 9% of logging was of good quality, 55% showed intermediate quality and 36% showed low quality (Figure 5). In the images, the log decks appear as yellow points; and the roads as light green lines. In the areas with logging of good quality, we observed the low impact on the canopy as light green patches in the images. The medium impact on the canopy observed in areas with intermediate quality appears as intense light green patches. In low quality harvesting, the log decks and roads are mixed, with the high impact on the canopy and appearing as more intense patches, varying from light green to yellow) (Figure 4). The high percentage of areas harvested in Pará and Mato Grosso with intermediate and low quality indicates a low level of adoption of forest management. This may also point to technical deficiency among company forest management technicians.

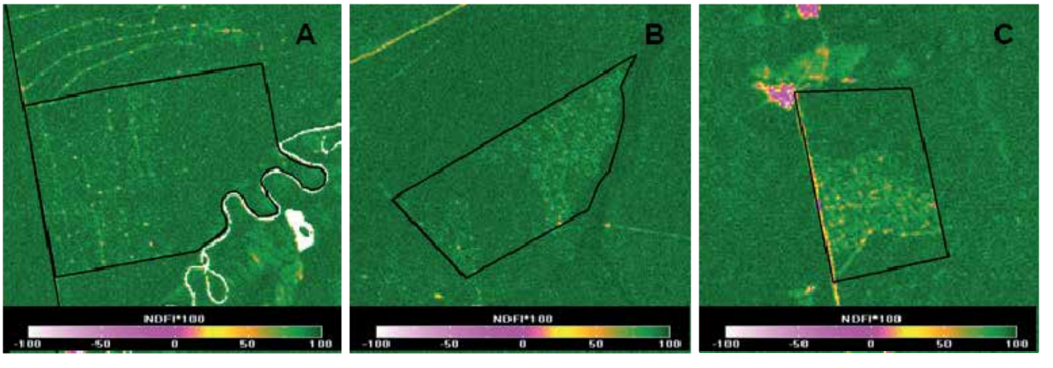

Fig. 4. Forest management of good (A), intermediate (B) and low (C) quality according to NDFI images.

To validate results of our assessment of the quality of timber harvesting as seen in satellite images, we went to the field to quantify it. To do this, we evaluated and scored impacts of logging resulting from opening of log decks and roads, felling trees and damages to remaining trees. We verified that the greater the impact, the lower the quality of harvesting and vice-versa. We thus attributed a score (from 0 to 4) and a corresponding classification (low, intermediate and good), in which: score <2 = low quality; score 2-<3 = intermediate quality; and score 3–4 = good quality. In table 4 we present the results of that validation. However, in the samples validated in the field we did not have cases of low quality management, despite having detected this standard of quality in the images. The quality standards were correctly classified in 86% and 58% of cases, as intermediate and good quality respectively (Table 4). The cases in which results in the images were different from those in the field may be related to the fact that the area evaluated in the field was geographically not the same area evaluated in the image. In the image we sampled the forest management area that visually was the most disturbed; however, because of the difficulty in accessing that area in the field, we had to evaluate another area geographically closed to the real area. With this, we verified that the quality of forest management can vary within the same licensed area, confirming the importance of monitoring forest management by satellite as a planning tool in enforcement campaigns by environmental agencies.

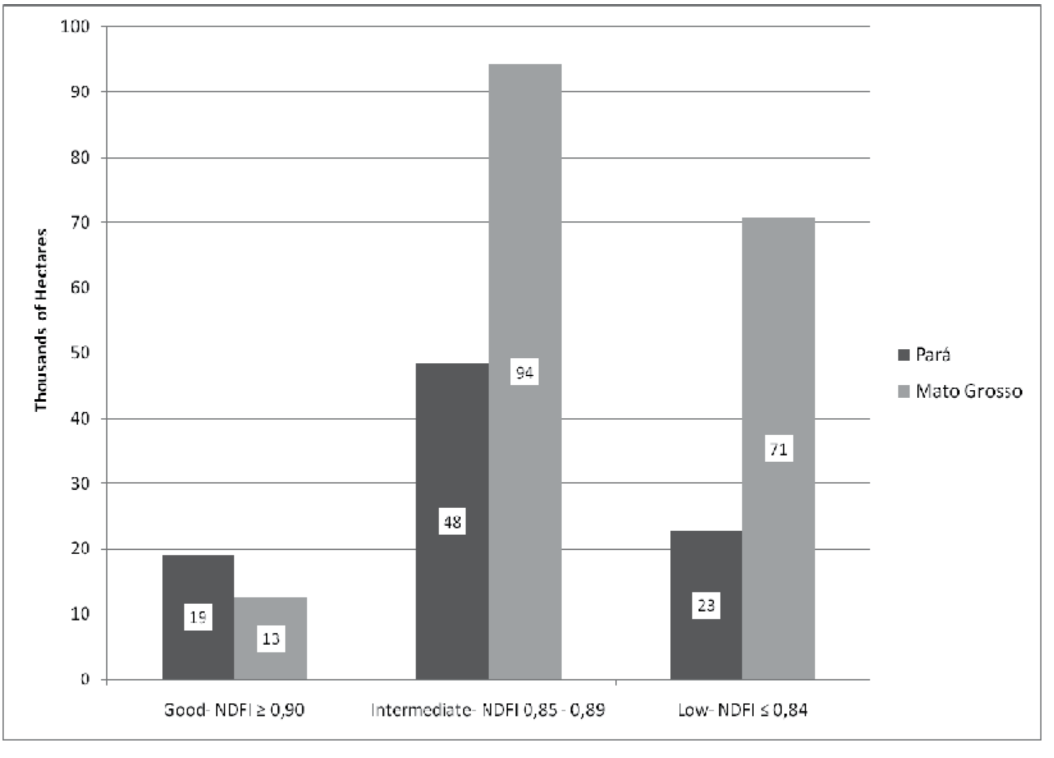

Fig. 5. Quality (in hectares) of timber harvesting in management plans in the State of Pará and Mato Grosso.

Forest Management Sample	Quality in the image		Quality in the field	
	Classification	NDFI	Classification	Scoring
1	**intermediate**	**0.86**	**intermediate**	**2.77**
2	**intermediate**	**0.89**	**intermediate**	**2.66**
3	intermediate	0.86	good	3.43
4	**good**	**0.90**	**good**	**3.18**
5	intermediate	0.85	intermediate	2.42
6	intermediate	0.88	intermediate	2.26
7	good	0.90	good	3.22
8	**good**	**0.90**	**good**	**3.00**
9	good	0.90	intermediate	2.54
10	**intermediate**	**0.88**	**intermediate**	**2.72**
11	good	0.91	intermediate	2.81
12	**good**	**0.90**	**good**	**3.09**
13	**good**	**0.91**	**good**	**3.09**
14	**good**	**0.90**	**good**	**3.36**
15	good	0.90	intermediate	2.81
16	**good**	**0.91**	**good**	**3.45**
17	good	0.90	intermediate	2.45
18	good	0.90	intermediate	2.45
19	**intermediate**	**0.89**	**intermediate**	**2.45**
20	good	0.91	intermediate	2.90
21	**good**	**0.91**	**good**	**3.27**
22	**good**	**0.91**	**good**	**3.18**
23	good	0.91	intermediate	2.45
24	good	0.90	intermediate	2.54
25	**good**	**0.91**	**good**	**3.09**
26	**good**	**0.90**	**good**	**3.27**

Table 4. Comparison of forest management quality obtained in the image and obtained in the field (Monteiro et al., 2011).

4.3 Forest biomass

We evaluated the loss of forest biomass indicator in the areas submitted to forest degradation. To do this, we first obtained an NDFI image to quantify forest degradation (Souza Jr. et al., 2005). Next, we integrated that information with the forest biomass data collected in the field (See section 2.1.2).

We observed that the NDFI value in the image diminishes with the increase in forest degradation. This means that the lower the biomass, the more degraded the forest; in other words, there is a high negative correlation between the biomass values quantified in the field with the NDFI values of the forest degradation classes (Figure 6) (Souza Jr. et al., 2009). However, that negative correlation is only observed when the NDFI image is from the same year as the occurrence of the degradation event, since beginning in the following year, the degradation signal diminishes (Souza Jr. et al., 2009).

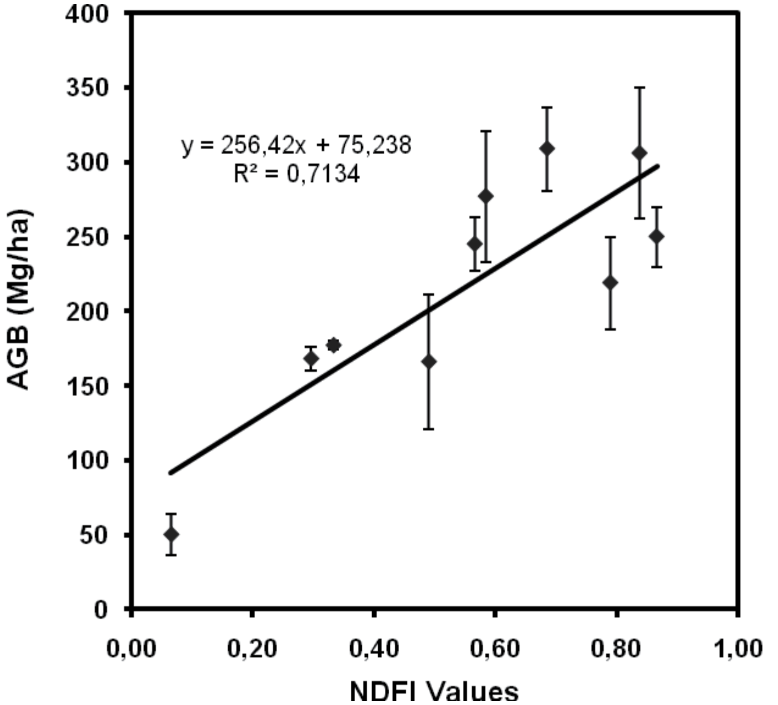

Fig. 6. Relationship between Aboveground Biomass- AGB and NDFI values for degraded forest of Paragominas and Sinop (Souza Jr. et al., 2009).

5. Applying remote sensing to monitor forest management in the Amazon

In the subsections below we present the results of remote monitoring in the timber harvesting areas of the States of Pará and Mato Grosso for the period of 2007 to 2009. We first mapped and classified timber harvesting as legal and illegal. Next, we identified the municipalities in those States where illegal forest activity is most critical. Later, we overlaid the map of illegal logging on the Protected Areas and land reform settlements so as to identify the areas under the greatest pressure from illegal timber harvesting. Finally, we integrated the information from satellite images with those of the forest control systems in those States.

5.1 Mapping of timber harvesting

We mapped logging using the NDFI images and overlaid that information on the map of forest management plans so as to identify non-authorized logging (illegal and predatory) and authorized logging (forest management). We quantified 543,504 hectares of logged forests in Pará, of which 86% were not authorized and 14% had an authorization for forest management. In Mato Grosso, we mapped 460,134 hectares of logged forests, of which 39% lacked authorization and 61% were authorized (Figure 7).

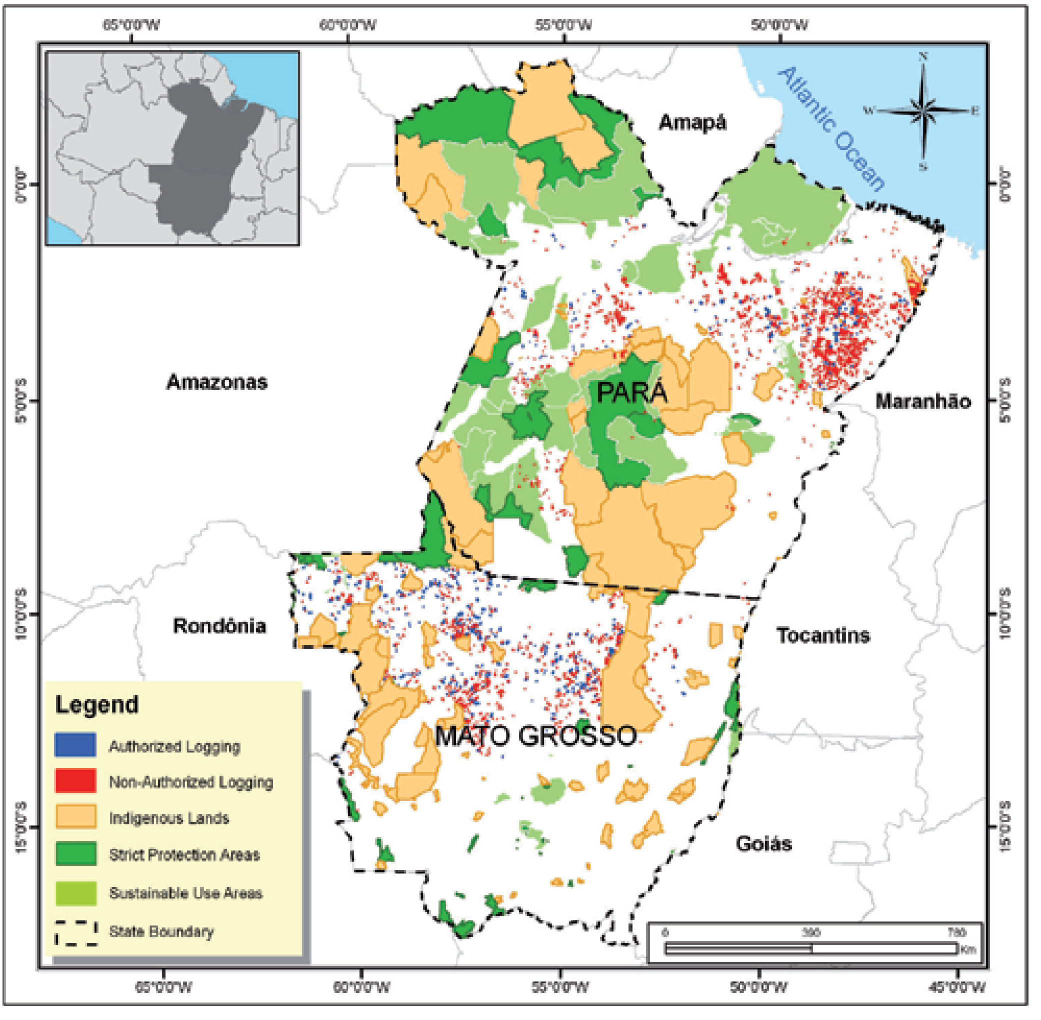

Fig. 7. Authorized and non-authorized logging from 2007 to 2009 in Pará and Mato Grosso states.

5.1.1 Non-authorized logging

Of the 466,979 hectares of forest logged without authorization in Pará between 2007 and 2009, the majority (77%) occurred in 10 municipalities. The remaining 23% were distributed more sparsely among 41 other municipalities. The municipality of Paragominas presented the largest area of non-authorized logging, followed by Rondon do Pará and Goianésia do Pará. In Mato Grosso, there were 179,155 hectares of forests logged without authorization, of which the majority (62%) occurred in 10 municipalities.

The remaining 38% were distributed more sparsely among 32 other municipalities. The municipality of Marcelândia presented the largest area of non-authorized logging, followed by Nova Maringá and Aripuanã.

From 2007 to 2009, illegal timber harvesting in Pará affected 54,874 hectares of forests in Protected Areas. Of that total, 83% was logged in Indigenous Lands (TI) and 17%, in Conservation Units (UC). TI Alto Rio Guamá was the most logged, followed by TI Sarauá and TI Cachoeira Seca. Among the Pará UCs, the National Forests (Flonas) of Jamanxim, Caxiuanã and Trairão are stand out as having the largest volume of timber harvesting. In Mato Grosso, illegal timber harvesting affected 10,524 hectares of forests in Protected Areas: 86% in TIs and 14% in UCs. TI Zoró had the highest amount of logging, followed by TI Aripuanã and TI Irantxe. Among the UCs, an Extractive Reserve (Resex Guariba/Roosevelt) and the Serra de Ricardo Franco State Park (PE) stand out with highest harvest volumes. Monitoring of Protected Areas is extremely important for guaranteeing their integrity and the sustainability of populations that depend on the forest for a living. Thus, environmental agencies can use this tool to restrain devastation of Protected Areas in the Amazon. Additionally, forest concessions in public forest areas such as Flonas need to guarantee income and employment for the population living inside and around those Protected Areas.

In the land reform settlements in Pará, timber harvesting without authorization between 2007 and 2009 affected 53,924 hectares of forests; the majority (75%) in 10 settlements. The most critical situation occurred in the Liberdade Sustainable Development Project (PDS) (50% of the total harvested), followed by the Ouro Branco I and II Collective Settlement Projects (PAC) (12% and 8%). In Mato Grosso, timber logging without authorization in the settlements affected 994 hectares of forests. The most critical situation was the Settlement Project (PA) of Pingos D'água (44% of the total harvested), followed by PA Santo Antonio do Fontoura I (33%). Rural settlement projects in the Amazon hold forest areas with great timber potential. However, in the majority of those projects logging is done in an illegal manner, meaning without a logging license. Programs that encourage forest practices through technical capacity-building for settlers can contribute towards reducing illegal timber harvesting in the settlements and generate income for those families.

5.1.2 Authorized harvesting

For the areas with authorized harvesting, in other words, with forest management, we evaluated the data contained in the Forest Harvesting Authorizations (*Autorizações de Exploração Florestal* - Autef) and in the timber credits issued from 2007 to 2009, in order to verify their conformity or consistency. That information is made available by the State Environmental Secretariats (Sema) in Pará and Mato Grosso, in their systems for forest control, Simlam (Integrated System for Environmental Monitoring and Licensing - *Sistema Integrado de Monitoramento and Licenciamento Ambiental*) and Sisflora (System for Sale and Transport of Forest Products - *Sistema de Comercialização e Transporte de Produtos Florestais*).

In Pará, 277,440 hectares of forests were licensed for management. Of that total, the majority (87%) did not present inconsistencies, while 13% revealed inconsistencies, such as: i) authorization for forest management in area totally or partially without forest cover (6% of cases evaluated); ii) area authorized for management superior to the total area for forest management (4% of cases); and iii) authorization for forest management in area already harvested through logging activities (3% of cases).

In Mato Grosso, 498,783 hectares of forests were approved for forest management, of which the majority (81%) presented no problems and 19% revealed inconsistencies. Those include: i) timber credit commercialized does not correspond to credit authorized (16% of cases); ii) area authorized in deforested area (1% of cases); iii) area authorized greater than the area for forest management (1% of cases); iv) credit issued without authorization for forest harvesting (1% of cases).

Finally, we integrated information from the Autefs with our satellite image base to assess the consistency of forest management performance. In Pará, the largest percentage (45%) of the Autefs evaluated in the satellite image presented no problems, while in 31% it was not possible to make an evaluation because of cloud cover and 24% revealed problems, such as: i) lacking signs of scarring from logging in the images for the period in which the logging authorization was in effect (11% of cases); ii) area of forest management licensed overlaying a Protected Area (5% of cases); iii) logging carried out before issuance of the forest authorization (3% of cases); iv) area licensed for forest management deforested before receiving authorization for harvesting (3% of cases); and v) area logged above the authorized limit (2% of cases).

In Mato Grosso, the same analysis revealed that the majority (78%) presented no problems in the satellite image, whereas 22% revealed problems, which were: i) area was logged above the authorized limit (16% of cases); ii) area licensed for forest management deforested before receiving authorization for harvesting (3% of cases); iii) lacking signs of scarring from logging in the images for the period in which the logging authorization was in effect (1% of cases); iv) plan overlaying a Protected Area (1% of cases); v) logging carried out before issuance of the forest authorization (1% of cases).

The method proposed in this study is capable of monitoring the performance of forest management by timber cutters and forest management licensing by the environmental agencies. This makes it possible to reduce the errors and frauds in the forest control systems during the forest management licensing process and during commercialization of timber.

6. Conclusion

Characterizing the impacts of timber harvesting in the field is essential for determining changes in the structure and composition of the forest submitted to different levels of forest degradation. However, that activity is extremely expensive and lengthy. The advance in techniques for detecting and mapping timber harvesting and integration of that information with data from the field has made it possible to monitor logging (Monteiro et al., 2011) and quantify the loss of carbon from degraded forest in the Brazilian Amazon (Souza Jr et al., 2009).

On the other hand, there is the challenge of putting into operation a system for monitoring timber harvesting at the scale of the Amazon. Logging in the region is predominantly predatory, which has contributed towards an increase in forest degradation and a reduction of the stocks of individual tree species with timber potential. Currently, the Amazon has two systems for detecting deforestation (clearcutting) the

Program for Monitoring Deforestation in the Amazon (*Programa de Monitoramento do Desflorestamento da Amazônia* - Prodes), developed by the Brazilian Space Research Agency (Inpe), and the Deforestation Alert System (*Sistema de Alerta de Desmatamento* - SAD), developed by the Amazon Institute for People and the Environment (Imazon). The method for monitoring timber harvesting proposed in this study can contribute towards reducing illegal logging and improve the quality of harvesting through forest management in the region. With that, we can reduce emissions of CO_2 (Putz et al., 2008) and guarantee the sustainability of the forest-based economy in the Amazon. That method could also contribute towards improving forest management in the Amazon by making it more efficient and transparent.

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5

Case Study of the Effects of the Japanese Verified Emissions Reduction (J-VER) System on Joint Forest Production of Timber and Carbon Sequestration

Tohru Nakajima
Laboratory of Global Forest Environmental Studies
Graduate School of Agricultural and Life Sciences
The University of Tokyo, Yayoi, Bunkyo-ku
Laboratory of Forest Management
Graduate School of Agricultural and Life Sciences
The University of Tokyo, Yayoi, Bunkyo-ku, Tokyo
Japan

1. Introduction

In the context of climate change (including global warming), the net reduction in carbon emissions as a result of forest carbon sinks and sustainable forest management are two critical issues. Recently, the benefits of carbon sequestration by forests have been highlighted and carbon sequestration has been measured throughout the world: in the United States (Sakata 2005; Calish et al 1978; Foley et al 2009; Ehman et al 2002; Im et al 2007), Europe (Backèus et al 2005; Liski et al 2001; Matala et al 2009;Pohjola and Valsta 2007;Sivrikaya et al 2007; Kaipainen et al 2004; Seidl et al 2007), Canada (Hennigar et al 2008; Thompson et al 2009), Oceania (Campbell and Jennings 2004) and Asia (Ravendranath 1995; Han and Youn 2009). Forests not only have economic value through the production of commercial timber, but they also have other values to society including acting as carbon sinks, supporting biodiversity, and providing water protection (Pukkala 2002). Forest management subsidies are required from national budgets (funded by the tax payer) to increase the public benefit of forests by restricting the area that is clear cut and preventing other damaging silvicultural activities from being practiced. On the other hand, in the absence of artificial thinning, intensive self-thinning can occur (Nakajima et al., 2011d), resulting in significant CO_2 emissions. Therefore, both thinning and harvesting are necessary not only for commercial timber production, but also in order to reduce CO_2 emissions and gain carbon credits. In addition, tree growth gradually decreases with age (Nakajima et al., 2010; 2011d; Pienaar and Turnbull 1978), so older stands will eventually cease to increase their carbon stock. It, therefore, makes economic sense to undertake clear felling before such stagnation occurs in older stands and carbon credits are no longer available.

Because Japanese forest management profits have been in decline as a result of lower timber prices (Forestry Agency 2007), almost all Japanese forest owners depend on government subsidies to maintain their forests (Komaki 2006; Nakajima et al 2007b). Previous studies have shown that the area of silvicultural practice including planting, weeding, pruning, pre-commercial thinning and thinning, is strongly correlated with the amount of national subsidy that is provided (Hiroshima and Nakajima 2006). Therefore, the planted forests of Japan that are funded by national subsidies should be in a condition suitable for the public to benefit from them. Generally, it is not possible to rely on natural regeneration in planted forests in Japan. The silvicultural practices used to ensure regeneration in Japan have been described in previous studies (Nakajima et al., 2011b; Sakura 1999; Ohtsuka, 1993) and are outlined in table 1.

Silvicultural practices	Stand age (year)
Land preparation and planting	0
Weeding	1–10*
Pruning	15
Precommecial thinning	20, 25

* Weeding is undertaken every year in stands aged 1 to 10 years

Table 1. Silvicultural practices undertaken at the study site

In addition, Japanese citizens think that acting as carbon sinks will be one of the most important functions of forests in the future (Forestry Agency 2007). Based on public opinion, it would be an valuable for forest managers to include the carbon benefits in their forestry profit predictions.

In order to include carbon benefits in forest management, a number of previous studies have proposed what is known as the 'social rule' (Im et al 2007; Foley et al 2009; Hennigar et al 2008). Because the rotation period is important for forest management decision making and strongly affected by regional forest resources, some studies have focused on estimating how the optimum rotation period is affected by different carbon offset systems. Carbon offsetting may be advantageous for forest management based on optimizing the rotation period (Raymer et al 2009), but it can be disadvantageous because of the effects of natural disturbance, which can release carbon (Galik and Jackson 2009). However, few studies have investigated the effects of existing carbon offsetting programs (including forest carbon sinks) in the context of global warming policy frameworks.

Under the global policy framework (resulting from the Kyoto Protocol) the size of the carbon sink in a forest is calculated for forests that have experienced afforestation, reforestation and deforestation (ARD forests) since 1990, as described by Article 3.3; and in terms of forests where silvicultural practices have been conducted since 1990 (FM forests) under Article 3.4.

The Kyoto Protocol requires signatories to reduce their CO_2 emissions and other greenhouse gases by their quantified reduction commitments below 1990 levels during the first commitment period (FCP), 2008–2012. Now that the end of the FCP is fast approaching, each country is preparing to report on emissions and the removal of carbon by forests in accordance with the Good Practice Guidance for Land Use, Land-Use Change, and Forestry (GPG-LULUCF) (Amano 2008a; Houghton et al 1997; IPCC 2000; IPCC 2007). In the protocol, Japan is committed to reducing CO_2 equivalent emissions to 6% below its 1990 level (Amano 2008b; Amano and Tsukada 2006). At the same time, the protocol allows net changes in greenhouse gas emissions to be included. For example, removal by sinks resulting from human-induced land-use changes and forestry (LUCF) activities can be added to or subtracted from their reduction commitments as appropriate.

Under the Kyoto mechanism, carbon emission trading can be undertaken. Carbon dioxide (CO_2) credits have already been traded in some markets, such as the carbon market in the United Kingdom since April 2002. The carbon price is expected to affect forestry profits and has the potential to cause considerable changes to harvesting ages. Predicting how changes in the cutting age affect carbon prices encourages the consideration of forestry measures in these terms. In order to quantify carbon storage, previous studies have proposed various methods for estimating carbon credits, including the stock changing, average storing, and ton-year methods (Richards and Stokes 1994; Schroeder 1992; Moura-Costa and Wilson 2000).

To accelerate efforts to combat global warming in accordance with the Kyoto Protocol, based on the stock changing method, Japan's Ministry of the Environment has established a forest carbon credit system. The system, which is based on the Japan Verified Emissions Reduction (J-VER) system, was launched in November 2008 and will help in calculations of forest CO_2 absorption. This is the first system of its kind. The absorption will be calculated in credits, which can then be sold to CO_2-emitting companies already registered in the J-VER system. The Ministry hopes that the credits will be traded on the carbon market in the future and funds reinvested in the expansion of the current area where silviculture is practiced; this can then be counted as CO_2 absorption under the Kyoto protocol.

Carbon accounting is based on accounting systems developed as part of the FCP under the Kyoto Protocol. Three project types are particularly important in the J-VER system: thinning promotion and management; sustainable forest management; and plantation management. Areas thinned after 2007 will be the target of the efforts under the Japanese system. Sustainable forest management activities will focus on areas that were harvested and replanted after 1990. Plantation projects will focus on replanting, and all forests eligible for credits under the credit system need to have a forest management system compliant with current Forest Law.

No previous study has clarified the effect of this new carbon offset accounting system on the actual forest area formally identified in the J-VER system. For medium- to long-term forest management strategies, it is important to clarify the effect of the J-VER system on forestry strategies. Therefore, this study aimed to investigate the effects of the carbon offsetting system on the carbon stock and timber production relative to the carbon price. Because harvesting activities need to be included in long-term forest management, we examined the sustainable forest management project under the J-VER system.

2. Materials and methods

2.1 Study area

This research was conducted in the University of Tokyo Forest, located in the cities of Kamogawa and Kimitsu, Chiba Prefecture, Japan (Fig. 1).

This forest lies 50 to 370 m above sea level and is characterized by undulating terrain with steep slopes and primarily brown forest soils. It is located in a warm temperate zone, with an average annual temperature of 14°C and an average annual precipitation of 2182 mm. The total forest area is 2216 ha; 824 ha (37%) contain sugi (*Cryptomeria japonica*) and hinoki (*Chamaecyparis obtusa*) stands, 949 ha (43%) are natural hardwood forest, and 387 ha (17%) are natural conifer forest. The remaining 57 ha (3%) are occupied by a demonstration forest. Many permanent research plots have been established in sugi stands within the forest since

1916, and tree height, height to crown base, and diameter at breast height (DBH) have been recorded approximately every 5 years since that time. A national subsidy system for the thinning of all planted tree species is commonly applied, but mainly to forest plantations less than 35 years old. The grant rates of the subsidy systems cover approximately 70 % of the cost of thinning. Inventory data relating to the private forests, such as stand age, area, tree species, slope, address of forest owners and site index, were available and were also linked to each stand included in the geographic information system (GIS). Using the inventory data, age distribution at this study site was derived and is shown in Figure 2. The site index map in this study area was also established using the airborne LiDAR measurements (Hirata et al., 2009; Hiroshima and Nakajima 2009). Only sugi (*Cryptomeria japonica*), the best-known planted tree species in Japan, was considered.

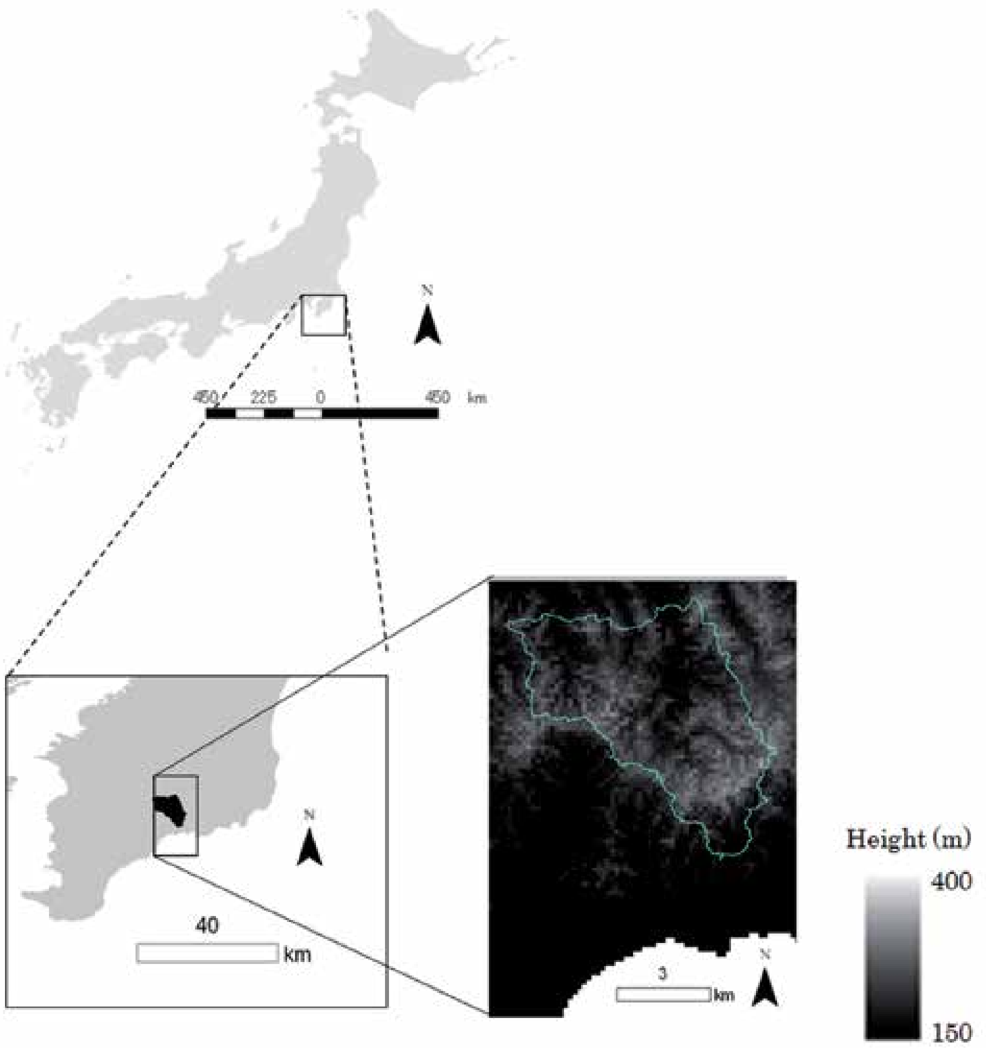

Fig. 1. Location of the University Forest in Chiba, showing an elevation of the study site. The blue line shows the forest boundary line of the University Forest in Chiba.

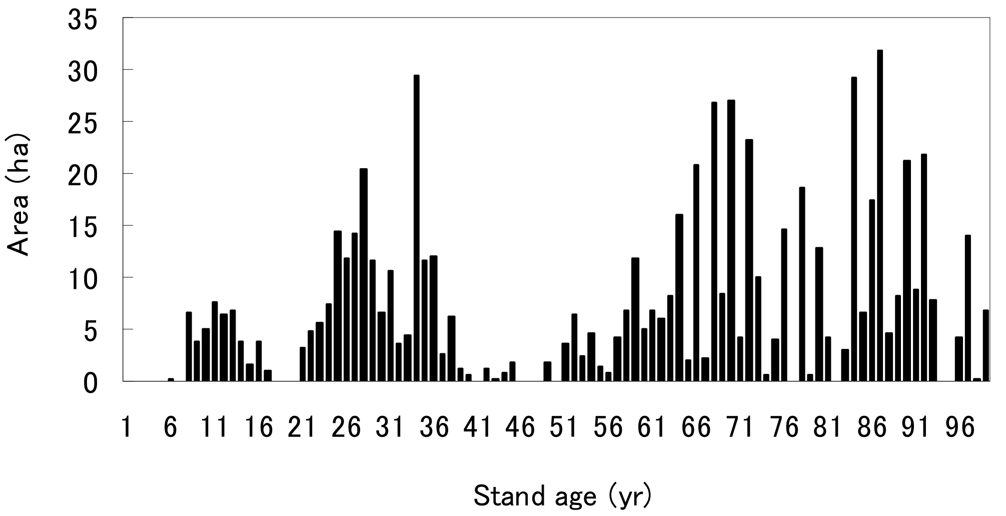

Fig. 2. The age distribution of forested areas in the study site

Approximately 58 % of planted forests in Japan are privately owned (Forestry Agency, 2007), and the forest policy subsidy system is known to have a great influence on the management practiced within them (Hiroshima and Nakajima, 2006). Furthermore, due to the socio-economic situation in Japan, there has been little financial incentive to practice sound forest management and profits have been very low as a result of decreasing timber prices. This has resulted in increased areas of unmanaged and unthinned forests, many of which have been left untended for more than 10 years (Nakajima et al., 2007).

Hence, there is an urgent need to improve the profitability of Japanese forestry. Due to the general lack of thinning, self-thinning has been increasing, accompanied by reductions in the carbon stock and adverse effects on forest ecosystem functioning. These developments are in direct conflict with a need to increase thinned areas of forest, relative to 1990 levels (Japanese Forestry Agency, 2007), under Kyoto Protocol commitments (Houghton et al., 1997; UNFCC, 1998; Robert et al., 2000; UNFCC, 2002; IPCC, 2003; Jansen and Di, 2003). Thus, there is an urgent need to expand the areas that are subject to planned thinning, and to reduce the cost of such operations by increasing their scale through forest owner cooperation. Therefore, silvicultural practices are now supported by a subsidy system (Nakajima et al., 2007), under which forest owners are required to report the conditions of their stands and the silvicultural treatments they have applied. The central government and local Prefectural government subsidize the thinning of planted forests containing trees younger than 35 years of any species, meeting approximately 70 % of the thinning cost. The subsidies for thinning are available in forests that have been subsidized in the preceding five years.

This area was one of the forest projects formally identified in Japan's Verified Emission Reduction system (J-VER), which is a Japanese carbon offset system. It is important, therefore, to establish a sustainable forest management system that takes into consideration timber production and the amount of carbon stock held in the area.

2.2 Analysis tool

The data source and analysis tool used in this study for estimating carbon absorbed by the forest were developed in accordance with the J-VER guidelines (Environmental Ministry

2009), which are based on the carbon accounting system developed for the Kyoto Protocol. J-VER guidelines suggest the use of the Local Yield Table Construction System (Nakajima et al. 2009a; Nakajima et al. 2010), which is a timber growth and carbon stock simulator. This growth model is applicable to the main tree species, including sugi (*Cryptomeria japonica*), hinoki (*Chamaecyparis obtusa*), karamatsu (*Larix leptolepis*) and todomatsu (*Abies sachalinensis*), which are planted throughout Japan. By combining LYCS with a wood conversion algorithm and a harvesting cost model (Nakajima et al. 2009a; 2009c), we can predict not only carbon stock but also harvested timber volume and forestry income. The stand age and tree species included in the forest inventory data can be used as input data for the LYCS. The harvest and silvicultural practice records of the study site, including details of incomes, costs, and labor, were used to estimate forestry profits for harvesting and silviculture. The unit price of subsidies depends on the standard silviculture system and historical records of the amount of labor required to carry out various silvicultural practices including silviculture treatments (planting, weeding, pruning, pre-commercial thinning) and harvesting (thinning, clear-cutting) were also available from the University forest in Chiba.

2.3 Data analysis

In the present study, we investigate through simulation modeling the effects of the J-VER system on timber production, carbon stock holdings. Two carbon price scenarios were assumed: Scenario 1 was no J-VER system applied to stands; Scenario 2 was the J-VER system fixing the carbon price to 1000 yen/ton-CO_2 considering previous research (Nakajima et al. 2011c), applied to stands. The international pledge made under the Kyoto Protocol commitments (Houghton et al., 1997; UNFCC, 1998; UNFCC, 2002), requires a 6 % reduction of CO_2 emissions from the 1990 level, of which 3.8 % may be attributed to carbon absorption by means of ‘forest management’ (Hiroshima 2004; Forestry Agency 2007). Increasing the area of ‘forest management’ as described under article 3.4 in the Kyoto Protocol, requires pre-commercial or commercial thinning (Nakajima et al., 2007a). Therefore, to fulfill Japan’s international pledge under the Kyoto Protocol in a global context (Hiroshima and Nakajima et al., 2006), it has been proposed that a new J-VER system (i.e. Scenario 2) can be applied. This will promote thinning and restrict large-scale clear cutting by supporting long-rotation silviculture (Forest Agency 2007).

Based on the assumptions of the two scenarios, the harvesting area, amount of harvested timber, subsidy, forestry profits, carbon stock and quantity of labor were calculated by using an existing stand growth model (Nakajima et al. 2010), a wood conversion algorithm (Nakajima et al. 2009c) and a forestry cost model (Nakajima et al. 2009a). With data describing the stand condition (stand age, site index and tree species), the thinning plan (thinning ratios, number of thinnings and the thinning age) and the timber price as model inputs, the future stand volume, timber volume and forestry profits can be generated as model output (Nakajima et al., 2009a, 2009c, 2010).

The accuracy of the basic model for predicting future stands has been exhaustively checked by comparing estimated tree growth with observed tree growth data in permanent plots (Ohmura et al. 2004) gathered over more than 30 years (Shiraishi 1986; Nakajima et al. 2010). By inputting the stand condition into these models, the future forestry profits could be estimated as a function of the harvesting plan strategies and the carbon price. However, because it is not easy to predict inflation and timber price fluctuations precisely, we assume in the model that the socio-economic situation driving these variables is constant. We therefore assume that timber price remains constant throughout the prediction period and is as described by a previous study (Nakajima et al. 2009a). We believe this assumption is justified since a survey by the forest association, and government reports (Forestry Agency 2007) indicate that the current annual average timber price has been stable over recent years. The final age at cutting was chosen to maximize the present net value of forestry profits, estimated from those valid at the most recent final cutting. Although the thinning plan is included in the input data as mentioned above, it can be changed according to a particular stand density control strategy. The optimum thinning plan was decided upon by selecting the one which maximized the net present value. We varied the thinning ratios by 5 % increments from 20 % to 40 % in line with the existing standard silviculture systems (Forestry Agency 2007). We also varied the number of thinnings between zero and three, and the thinning age by increments of 5 years between the initial stand age and the final age at cutting. By inputting these various thinning plans into the LYCS, we simulated forestry profits under all harvesting strategies. We then selected the cutting plan that maximized the present net value of forestry profits.

The forestry profits could then be estimated from the forestry income and the carbon credit. Sakata (2005) examined the effects of the carbon market on forestry profits in the USA. At the study site selected by Sakata (2005), both saw logs and pulp wood were considered to contribute to any profits. On the other hand, production of pulp wood at the current study site is not commercially viable because the cost of harvesting is so high. Therefore, the study described herein examined the effects of the carbon market on forestry profits when producing saw logs alone and not pulp wood.

The carbon stocks were also estimated by substituting stand volumes derived from LYCS into the following formula (Environmental Ministry 2009):

$$C = V \cdot D \cdot BEF \cdot (1 + R) \cdot CF \tag{1}$$

where C is the carbon stock (t-C), V is the stand volume (m^3), D is the wood density (t-dm/m^3), BEF is the biomass expansion factor, R is the ratio of below ground biomass to above ground biomass and CF is the carbon content (t-C/t-dm).

The biomass expansion factor for trees younger than 20 years was 1.57; the biomass expansion factor for trees older than 20 years was 1.23; the ratio of below ground biomass to above ground biomass was 0.25; the wood density (tonnes/m^3) was 0.314; and the carbon content（t-C/t-dm）was 0.5 (Environmental Ministry 2009; Fukuda et al. 2003). By multiplying 3.67(44/12=molecule of CO_2/molecule of C) by the amount of the carbon stock present, the amount of CO_2 can be calculated. The carbon credit can be calculated by multiplying the CO_2 increase per year by the carbon price (yen/ton).

Many previous studies (van Kooten et al. 1995; Nakajima et al., 2011c) used increases in timber volume as a base from which to calculate carbon credits. The gain in carbon credits has been calculated on the basis of timber growth, and the release of carbon credits occurred when timber was harvested. In the J-VER system, however, the accounting is based on the total volume of the tree stock (we refer to this method as J-VER accounting). Therefore, when estimating carbon credits under the J-VER system, there is no need to undertake lifecycle assessments. We conducted a sensitivity analysis, in order to clarify the effects on the net present value (NPV) of changes in various parameters, including the initial stand age (0, 20 or 40 years), the site index and the carbon price (CP) and discount rate within the J-VER system.

The traditional final cutting age in order to maintain the maximum mean growth rate in Japanese planted forests is approximately 50 years (The Tokyo University Forests, 2006). Using this age as a reference, we set the initial stand ages in our models to be 0, 20 and 40 years. The discount rate was then estimated relative to a value considered to be reasonable to society; in this case 3.0 % was considered reasonable as this represents the average long-term yield of Japanese government bonds (Tokyo Stock Exchange 2007). Using the discount rate (3.0 %) as a reference, we set the discount rate to 0, 20 and 40 % in our models. Using the yield table presented by Nakajima et al. (2010) as a reference, we set the site indexes 1, 2 and 3 to represent good, intermediate and poor site quality, respectively. We examined various combinations of the different parameters to estimate the NPV of timber production, carbon credits and total NPV. In addition, wind hazard probability is an important parameter; wind it the main natural disturbance in Japanese mountain forests and it increases with increasing stand age and height (Nakajima et al., 2009b; Tsuyuki et al., 2011). The probability of wind disturbance, thus, also increases with time. However, tremendous wind disturbance records were not observed in the study site, so we did not include this parameter when calculating NPV for the forest area studied.

Based on the methodologies for calculating NPV mentioned above, the predictions at the forest level could then be estimated by summarizing the predicted values at the stand level. Because the period of validation over which these previous studies were conducted was longer than the prediction period of 25 years adopted in the present study, estimates of future timber production and forestry profits (Nakajima et al. 2009a; 2009c) could be calculated based on predictions of future tree growth at the level of stands. If the predicted values derived from existing models at the stand level are accurate, it follows that the predicted value at the forest level, which is the sum of values at the stand level, would be also accurate. For descriptive purposes, the prediction period was set to 25 years, which is the period specified for natural resource predictions by the Japanese Ministry of Education, Culture, Sports, Science and Technology (Science Council 2008).

By inputting the stand condition derived from forest inventory data into our models, future forestry profits could be estimated as a function of the harvesting plan strategies and the carbon price. As mentioned above, the discount rate was then estimated relative to a value considered to be reasonable to society; in this case 3.0 % was considered reasonable as this represents the average long-term yield of Japanese government bonds (Tokyo Stock Exchange 2007). The total harvesting area and the quantity of harvested timber were calculated by summarizing their respective values based on the harvesting plans calculated for each of the two scenarios under the carbon price of 0 and 1000 yen/CO_2-ton. The subsidies were estimated by summarizing the silviculture and thinning subsidies derived from government subsidy unit prices. In this study, the term "thinning subsidies" refers to subsidies associated with commercial thinning. In other words, the harvesting is not conducted as part of the silvicultural practices that include pre-commercial thinning. The total forestry profits could then be estimated from the forestry income and the subsidy. The carbon stocks were also estimated by substituting stand volumes derived from LYCS into the following formula (1):

In addition, labor requirements were calculated by multiplying the amount of labor required per hectare for each silvicultural practice, by the area over which that silviculture would be practiced, based on the estimated harvesting plans and the age distribution of trees in the study site.

3. Results and discussion

Results of the sensitivity analysis, based on the initial stand age (0, 20 or 40 years), and taking into account carbon price (CP), discount rate and site index within the J-VER system, are presented in Figure 3.

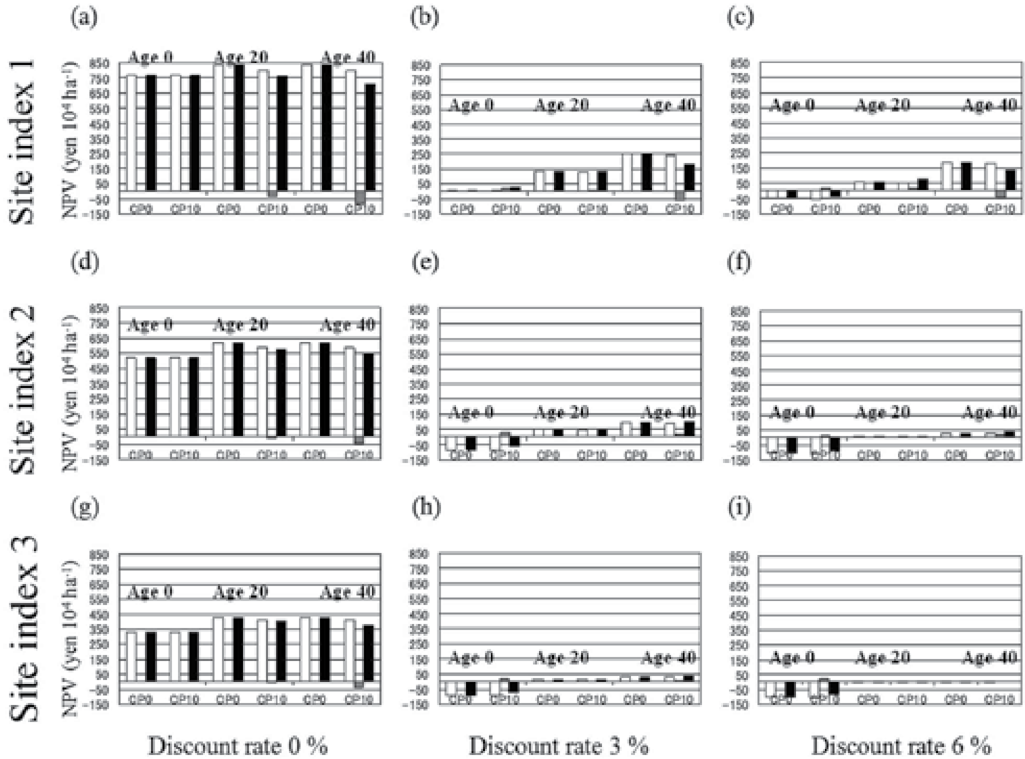

Fig. 3. Sensitivity analysis separated on the basis of initial stand age (0, 20 and 40 years), taking into account the site index, carbon price (CP) and discount rate within the J-VER system.

The white, grey and black bars show, respectively, the NPV of timber production, the carbon credit and the total NPV.

The profits change depending on the site quality, initial stand age, the carbon credit and the discount rate. As shown in figure 3, the higher the initial stand age, the lower the effect of carbon credit on the total profit. In addition, the higher the discount rate, the lower the profit. It is particularly noteworthy that the profit when the initial stand age is 20 years under the J-VER system shown in figure 3b is almost 0. This means that the carbon credit of 1000 yen for a stand with a site index of 1 and an initial stand age of 20 years could be sufficient to compensate landowners and make carbon storage economically attractive.

Several previous studies that have examined the effect of carbon price and taxes on forest management have accounted for carbon stock and release on the basis of timber volume (we call this method 'timber-based accounting' Nakajima et al., 2011c; van Kooten et al. 1995). The J-VER system had a greater impact on forestry profits than the timber-based accounting system (Nakajima et al., 2011c). Generally, the economic effect on the NPV calculated by the

J-VER accounting system was more sensitive than that calculated in previous studies using timber-based accounting (Nakajima et al., 2011c). We consider that the main reason for this result is that the estimated number of carbon credits under the J-VER accounting system is greater than estimates using the timber-based method (Nakajima et al., 2011c; van Kooten et al. 1995). This difference affected the profits derived from different forests depending on the age distribution under the carbon offsetting system. Figure 3 shows the positive or negative effect of stand age and carbon price in the targeted forest area on forestry profits. A strong positive effect was found for younger stands and a negative effect was found for older stands under the J-VER system. For example, in figure 3, the total effect of a carbon price of 1000 yen on the forestry profits for a stand with an initial age of 0 years (e.g. Fig. 3b, e, h) was positive, but with an initial stand age of 40 years (e.g. Fig. 3a, b, c, d, g) the owner would make a loss. Therefore it might be more important to consider stand age distribution, allocation of the harvesting area and carbon price fluctuation when planning forest management under the J-VER accounting system. Under the J-VER system, the total carbon storage included leaves, branches and roots, which were all counted as carbon sinks. Therefore, the lost of carbon credit by emission derived from clear cutting was greater than that calculated using the timber-based accounting system. In general, the age of existing Japanese planted forest stands is increasing (Forestry Agency 2007). Therefore, we suggest that the J-VER system may have a negative effect on forest profitability throughout Japan.

In particular, such negative impacts are likely to be greater in stands with high site quality. Therefore, harvesting, and particularly clear cutting, of stands on high quality sites will decline (Fig.3a-c). At the forest level, for the whole area examined in this study, the harvested area was calculated by summing the stand level harvesting area. Thus, the harvested area at the forest level also decreased under the J-VER system (Fig. 4).

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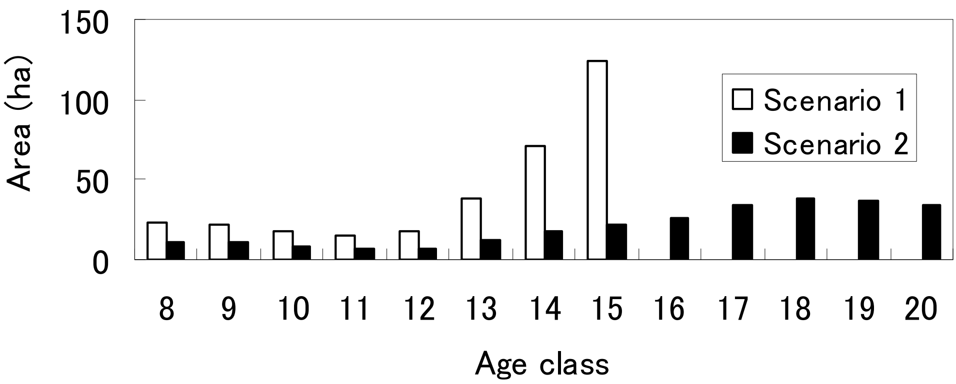
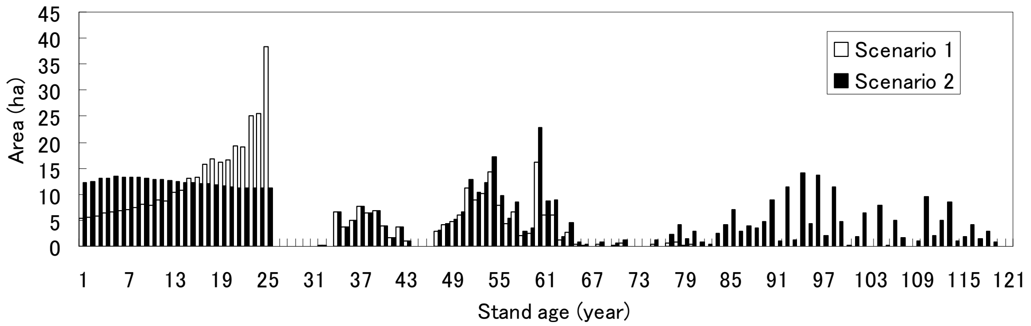
Figure 4 shows: (a) the age distribution of the final cutting area under different scenarios and (b) age class graphs for the scenarios at the end of the 25-year simulations. The former shows that the average stand age, under Scenarios 1 and 2, at the time of clear-cutting was 65 years and 80 years, respectively. The age classes at clear-cutting ranged from 8–15 years under Scenario 1, and from 8–20 years under Scenario 2. Because the target tree species was the most commonly planted species for timber production in Japan (Forestry Agency, 2007), this tendency for a reduction in the harvested area at the forest level studied could be applied to the regional level.

The increase in the potential harvesting area is derived from the increasing area of mature forest as the age distribution of stands in the study site changes over time (Fig. 5).

Under Scenario 1, profits from stands in an age class greater than 4 (36 years old) could be derived from harvest income alone, while under Scenario 2 profits could be derived from harvesting income and carbon sequestration. A comparison of the two scenarios clearly reveals a larger clear-cutting area under Scenario 1 than under Scenario 2 in the initial stage under the prediction period, the difference ranging between 1 ha and 27 ha. In 2021, the magnitude of the difference in clear-cutting areas decreased by up to 5.3 % of its maximum value. In contrast, the thinning area under Scenario 2 is clearly larger than under Scenario 1, with the difference ranging between 10 and 17 ha. These results show that the harvesting practices under the scenarios 1 and 2 were mainly clear cutting and thinning, respectively.

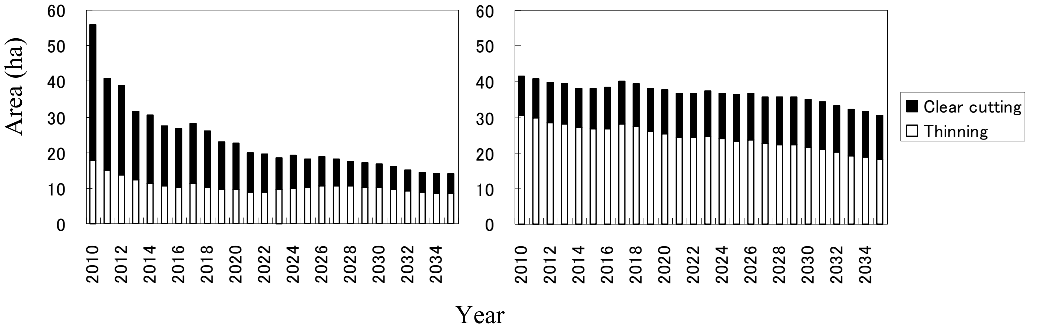


(a)



(b)

Fig. 4. (a) The age distribution of final cutting area under different scenarios and (b) age class graphs of scenarios at the end of the 25-year simulations. White and black blocks show the final cutting area under Scenarios 1 and 2, respectively.



(a) (b)

Fig. 5. The clear-cutting and thinning harvesting areas under (a) Scenario 1 and (b) Scenario 2.

White and black blocks show the thinning and clear-cutting harvesting areas, respectively.

3.1 Timber production

Figure 6 shows the differences in volumes of harvested timber under the two scenarios. Under Scenario 1, the harvest of clear-cut timber at the initial stage of the prediction period was larger than that of thinned timber, with a percentage clear-cut to thinned timber ranging from 87 % and 13 % in 2010 to 64% and 36 % in 2033.

After 2011, the volume of harvested timber decreased by up to 15.5 % of its maximum value due to a decrease of harvesting area (Fig. 5a) for clear-cutting.

Under Scenario 2 the clear–cut timber harvest was little larger than that of thinned timber with the percentages of the clear-cut to thinned timber ranging between 47% and 53 % in 2010 to 29% and 71 % in 2035.

The harvested timber volume decreased by up to 91.2 % of its maximum value between 2010 and 2035 due to a reduction in the harvested area (Fig. 5b). Although the total volume of harvested timber under Scenario 1 was larger than that under Scenario 2 up to 2014, in 2015 the pattern was reversed.

A comparison of the two scenarios clearly shows that the harvested volume of clear-cut timber in the initial stage of the prediction period was larger under Scenario 1 than Scenario 2, with differences ranging between 7.7 and 0.3 × 10^3 m^3. After 2010, the difference between volumes of clear-cut timber decreased by up to 2.5 % of its maximum value. In contrast, the volume of thinned timber harvested under Scenario 2 was clearly larger than under Scenario 1, with differences ranging between 1.2 and 1.6 × 10^3 m^3. These results show that production was predominantly of clear-cut timber especially under Scenario 1. Comparing Figs 5 and 6 shows that the ratio of clear-cut timber to total harvested timber is higher than the ratios of their respective harvested areas indicating that the volume of harvested timber per unit of harvested area was larger for clear-cut timber than thinned timber. Under the J-VER system (scenario 2), the amount of timber derived from clear cutting, which generally yields timber of larger dimensions than that derived from thinning, would be less than that under the non J-VER system (see figure 6). In particular, in the short-term, the total timber yield would be reduced under the J-VER system.

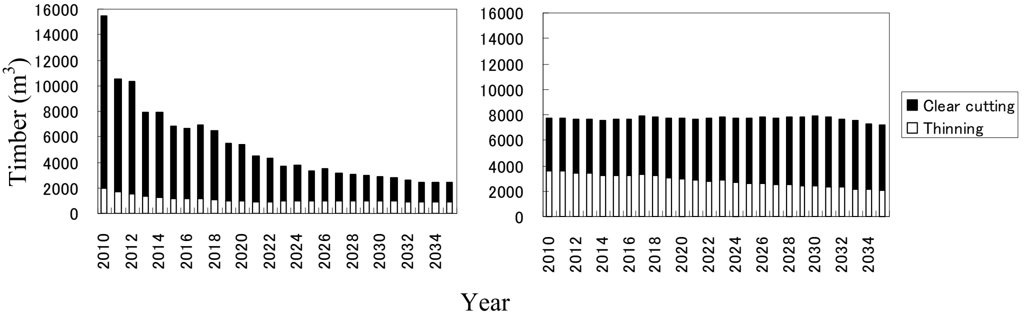


Fig. 6. The clear-cutting and thinning harvested timber volume under (a) Scenario 1, and (b) Scenario 2.

White and black blocks show the thinning and clear-cutting harvested timber volume, respectively.

3.2 Carbon stock

Figure 7 shows the response of the carbon stock to the different scenarios. Under Scenario 1 the maximum and minimum carbon stocks were 49948 tonnes in 2010 and 27639 in 2023. The carbon stock decreased by up to 55.3 % of its maximum due to the reduction in area harvested by clear-cutting (Fig. 5a).

Under Scenario 2 the maximum and minimum carbon stocks were 78037 tonnes in 2010 and 48342 in 2035. Between 2010 and 2035 carbon stock increased by up to 61.9 % of its minimum due to forest growth (Fig. 7b). The total carbon stock was smaller under Scenario 1 than under Scenario 2 throughout the prediction period.

Generally, the carbon stock under Scenario 2 was relatively more stable than that under Scenario 1. A comparison of the two scenarios clearly shows the carbon stock under Scenario 1 to be smaller than under Scenario 2 with differences ranging between 0 and 658.3 Kt suggesting that differences in carbon stock between the two scenarios were mainly due to clear-cutting. According to the carbon accounting system under the Kyoto Protocol, all carbon stock held as standing timber is counted as being released into the atmosphere by clear-cutting (Hiroshima and Nakajima 2006). Therefore, the larger clear-cutting area (Fig. 5) under Scenario 1 decreased the carbon stock dramatically.

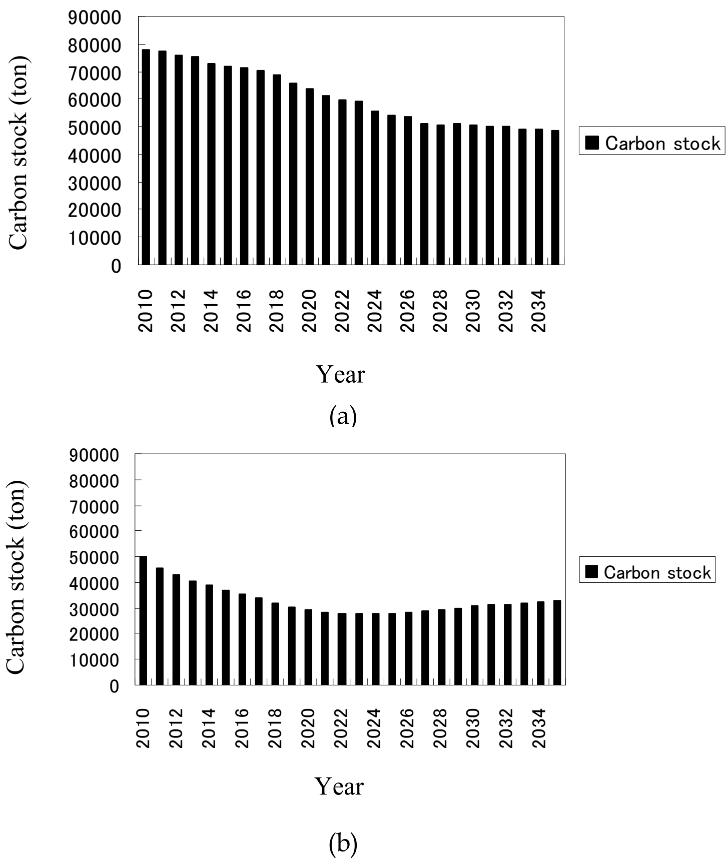


Fig. 7. The carbon stock under (a) Scenario 1, and (b) Scenario 2.

3.3 Subsidy

Figure 8 shows how subsidies vary depending on the scenario. Under Scenario 1, the maximum and minimum subsidies were 32.9 million yen (M¥) in 2017 and 6.8 M¥ in2010; the maximum and minimum silviculture subsidies were 30.0 M¥ in 2017 and 2.4 M¥ in 2010; and the maximum and minimum thinning subsidies were 4.4 M¥ in 2010 and 2.1 M¥ in 2035. Under Scenario 1 the silviculture subsidy was generally larger than the thinning subsidy, with the percentages ranging from 35 % and 65 % in 2010 to 92 % and 8 % in 2016.

After 2017, the subsidies decreased by up to 38.2 % of their maximum value due to a decrease in the harvesting area (Fig. 5a) for clear-cutting. The subsidy in 2035 was 184.6 % of the subsidy in 2010. Under Scenario 2 the maximum and minimum subsidies were 24.9 M¥ in 2017 and 1.0 M¥ in 2010; the maximum and minimum silviculture subsidies were 19.1 M¥ in 2034 and 2.4 M¥ in 2010; and the maximum and minimum thinning subsidies were 7.6 M¥ in 2010 and 4.6 M¥ in 2035. Under Scenario 2 the thinning subsidy in the initial stage of the prediction period was larger than the silviculture subsidy with percentages of silviculture and thinning subsidies ranging from 24 % and 76 % in 2010 to 80 % and 20 % in 2035.

Subsidies increased by up to 248.9 % of their minimum value over the period of simulated predictions due to an increase in the total harvesting area (Fig. 5b). The total subsidy under Scenario 1 is larger than that under Scenario 2 between 2012 and 2021.

A comparison of the two scenarios shows the silviculture subsidy in Scenario 1 of the initial stage under the prediction period to be clearly larger than that of Scenario 2, with differences ranging between 0 and 14.9 M¥. After 2012, the difference of silviculture subsidy decreased by up to 0.2 % of the maximum difference, while the thinning subsidy was clearly larger under Scenario 2 than Scenario 1, with differences ranging between 2.5 and 4.2 M¥.

3.4 Forestry profits

Figure 9 shows the forestry profits under the two scenarios. Under Scenario 1 the maximum and minimum forestry profits were 44.2 M¥ in 2010 and 1.8 M¥ in 2029. After 2011, the forestry profits decreased by up to 4.0 % of their maximum values due to a decrease of harvesting area (Fig. 5a) for clear-cutting.

Under Scenario 2 the maximum and minimum forestry profits were 19.6M¥ in 2011 and 12.2 M¥ in 2035. Between 2010 and 2035 forestry profits decreased by up to 62.4 % of their minimum values due to the increased harvesting area (Fig. 5b). Although the total forestry profits under Scenario 1 are larger than under Scenario 2 up to 2012, the pattern was reversed in 2013.

A comparison of the two scenarios shows the forestry profits under Scenario 1 before 2013 to be larger than under Scenario 2, with differences ranging between 3.4 M¥ and 24.9 M¥.

3.5 Labor requirements

Figure 10 shows the labor requirements under the different scenarios. Under Scenario 1 the maximum and minimum labor requirements were 4647 workers in 2012 and 1830 workers in 2011; the maximum and minimum number of required silviculture workers were 2977 in 2011 and 87 in 2011; the maximum and minimum number of workers for stand thinning were 799 in 2010 and 342 in 2035; and the maximum and minimum number of forest workers for clear-cutting were 1627 in 2010 and 186 in 2034. Under Scenario 1 the labor requirements for clear-cutting and silviculture were generally larger than those for thinning, with the ratio of the proportion of total labor required for clear–cutting to the proportion of the total labor required for thinning ranging from 58 % and 37 % in 2011 to 18 % and 13 % in 2019.

After 2012, the labor requirements decreased by up to 30.5 % of their maximum value. The overall decrease was due to a decrease in the harvesting area (Fig. 5a) for clear-cutting.

Under Scenario 2 the maximum and minimum labor requirements were 3272 personnel in 2030 and 2014 in 2011; the maximum and minimum numbers of workers required in silviculture were 1663 in 2032 and 87 in 2011; the maximum and minimum numbers of people involved in thinning were 1450 in 2010 and 829 in 2035; and the maximum and minimum numbers of clear-cutting forest workers were 661 in 2031 and 498 in 2010. Under Scenario 2 the labor required for clear-cutting and silviculture was generally larger than was required for thinning, with percentages of clear-cutting labor to thinning labor ranging from 25 % and 71 % in 2011 to 20 % and 27 % in 2034. Labor requirements increased by up to 162.5 % of the minimum value between 2011 and 2030, the increase being due to the increase in harvesting area (Fig. 5b).

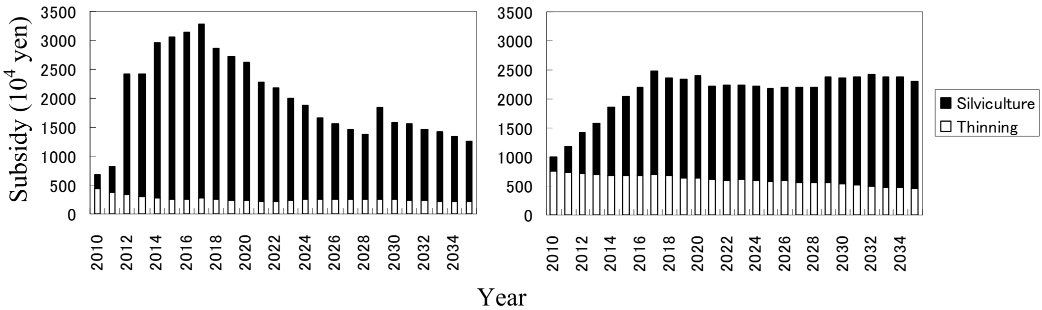
Labor requirements increased by up to 162.5 % of the minimum value between 2011 and 2030, the increase being due to the increase in harvesting area (Fig. 5b).

A comparison of the two scenarios clearly shows that silviculture requires more workers under Scenario 1 than under Scenario 2 in the initial stage of the prediction period with differences ranging between 0 and 2039 personnel. After 2013, the difference in labor requirements for silviculture decreased by up to 1.0 % of the maximum value. In contrast, the labor required for thinning was greater under Scenario 2 than under Scenario 1, with differences ranging between 486 and 857 personnel. These results suggest that the differences in labor requirements under Scenarios 1 and 2 were mainly associated with silviculture practices and thinning, respectively.

Because the estimated subsidies, forestry profits, carbon stocks, and labor requirements are affected by fluctuations in the stand age distribution and the stand condition over time, the observed pattern of increase was not monotonic.

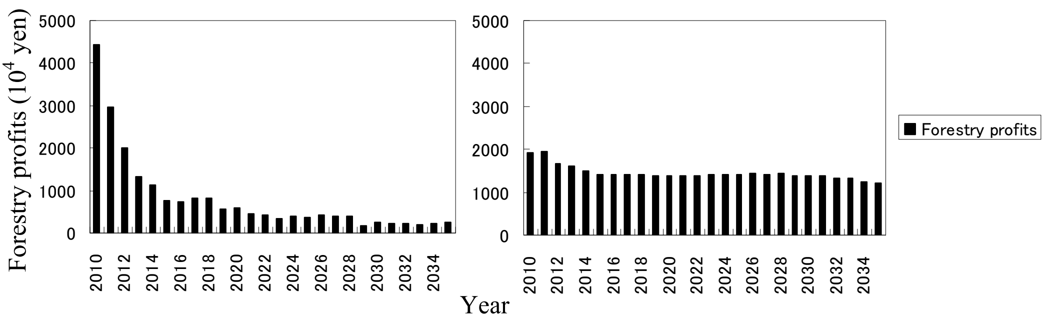
Our approach enables the effects of different carbon price scenarios on forestry to be calculated. Although timber production is the basic function of forests, their role in storing carbon stock also holds a high position in the public mind, especially during the first commitment period of the Kyoto Protocol. Figures 6 and 9 enable us to consider the influence of forest management under different carbon price on both of these factors. In addition, the simulation results for subsidies and labor requirements can be considered as important practical issues for forest management. Subsidies (Fig. 8) and labor requirements (Fig. 10) under the two scenarios were thus mainly allocated to clear-cutting and thinning (Fig. 5) under Scenarios 1 and 2, respectively. These results suggest that if the clear-cutting area were to decrease (Fig. 5a), the required subsidy (Fig. 8a) and labor (Fig. 10a) would not decrease immediately, because weeding continues to be required for 5 years after planting in the clear-cutting area.

Previous studies have analyzed useful variables and estimated parameters for several econometric models including the probit model (Dennis 1990; Pattanayak et al. 2003) and the logistic regression model (Royer 1987; Zhang and Pearse 1997), which can be used to predict the effects of forestry policies and subsidy systems. Other previous studies (e.g. Lewis and Plantinga 2007; Kurttila et al. 2006; Bolkesjø and Baardsen, 2002) have created models to estimate the effects of different amounts of subsidy. The models used herein mainly made use of established statistical techniques.



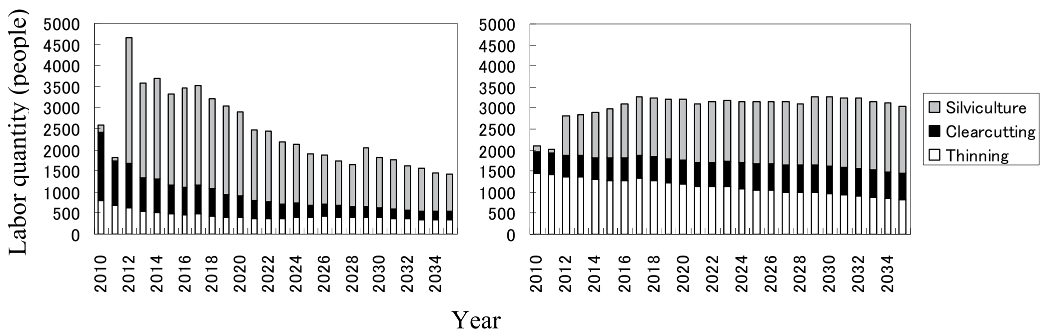
(a) (b)

Fig. 8. The silviculture and thinning subsidy under (a) Scenario 1, and (b) Scenario 2. White and black blocks show the thinning and silviculture subsidy, respectively.



(a) (b)

9. The forestry profits under (a) Scenario 1, and (b) Scenario 2.



(a) (b)

Fig. 10. The labor requirements for silviculture, clear-cutting and thinning under (a) Scenario 1, and (b) Scenario 2.

White, black and gray blocks show the thinning, clear-cutting and silviculture labor requirements, respectively.

If policy makers wish to apply these models to other geographical areas, different values for the statistical parameters may be required. We made use of a number of simulations developed and applied to Japan at the national level (Nakajima et al., 2010), so the current models are applicable throughout Japan without the need for new estimates of the parameters. Compared with other studies using similar statistical modeling approaches (Dennis 1990; Pattanayak et al. 2003; Royer 1987; Zhang and Pearse 1997; Lewis and Plantinga 2007; Kurttila et al. 2006; Bolkesjø and Baardsen, 2002), our work appears to be more broadly applicable. Although there may be dramatic changes in carbon and timber prices in the future, our approach should enable us to predict the effect of carbon price scenarios on forest resources and timber production in Japanese forest plantations.

For instance, in the present study, under Scenario 1 it is feasible to increase timber production during the early period of our predicted output (Fig. 6). However, Scenario 2 is a better option if the forests' function of holding carbon stock is the more pressing and stronger requirement (Fig. 9). The most suitable scenario could be selected by considering practical issues based on labor requirements and subsidies (Figs 8 and 10).

As explained in the introduction, Scenario 2 focuses on expanding the thinning area and restricting the clear-cutting area and so supports long-rotation silviculture as a means of increasing the carbon stock as required under the Kyoto Protocol. A comparison of the simulation results of Scenarios 1 and 2 shows that maintaining the carbon stock is more feasible under Scenario 2 (Fig. 7). Because a larger amount of subsidy is available for silviculture (Fig. 8a) following regenerations in the larger clear-cutting area (Fig. 5a). However, if the production of a large amount of timber is not an immediate requirement, Scenario 2 can be the better alternative with a lower subsidy budget. Notwithstanding this, in terms of the efficient use of the timber resource, such a choice might be irrational under some circumstances because of the possibility that some profitable stands might then be forced to avoid clear-cutting in order to produce larger timber.

These simulations can help policy makers and forestry practitioners propose policy changes that would not only enhance timber production, but also fulfill carbon stock obligations pledged under the Kyoto Protocol. Because there was no real and practical system for trading carbon credits at that time, Calish *et al*. (1978) did not consider the accumulation of carbon credits to be a management objective. The current study clarifies the effect of the Japanese carbon credit trading system on future forest resources. Sakata (2005), similarly, examined the effects of the carbon market on forestry profits in the USA. At the study site selected by Sakata (2005), unlike our site, pulp wood was a second commercially viable product, along with timber. Therefore, the current study shows the effect of the carbon market on forestry profits associated with timber but not pulp wood production.

Planted forests in the present study was conducted are highly productive of timber, especially from the main tree species (*Cryptomeria japonica*). Because this species is very broadly distributed (Fukuda et al. 2003), the simulations described here, which are based on real data, could also be applied to planted forests in other regions. In other words, *Cryptomeria japonica*, which is the target tree species in this study, is the most common tree species in Japanese planted forests (Forestry agency, 2007), so the work is applicable to other parts of Japan. In addition, the growth prediction system used in this study has been applied to the main tree species that grow throughout Japan (Nakajima et al., 2010; 2011a), so this methodology could also be applied to other areas of Japan. Sakata (2005) estimated the effect of the carbon market on the forestry profits based on standard silvicultural practices and costs over a large area including the southeastern United States. Although we considered a standard silvicultural system and costs (Nakajima *et al*., 2011a) in the present study, we made use of real age distribution and site index data for the study site, which is representative of much of the Japanese planted forest area (The Tokyo University Forests, 2006). We, thus, consider our results to be generally applicable across Japan.

Basically, the cycle for forest management depends on the management objective. Although in the present study, we assumed certain values in order to predict the effect of the real Japanese carbon trading system on timber production and carbon stock, certain socio-economic conditions that are represented by model parameters, could change. However, because the discount rate is the interest rate used to determine the present value of future cash flows (Eatwell et al. 1987; Winton JR. 1951), it is defined relative to a value that society considers to be reasonable. Although a previous study (van Kooten et al. 1995) has stated that, in general, the higher the discount rate, the shorter the rotation period, it is difficult to predict accurately not only the future timber price but also the discount rate as it might be affected by changing socio-economic conditions. Thus, it would be better to improve forest management plans by inputting into the simulation model parameter values that reflect the current socio-economic conditions, and changes in those socio-economic conditions, including discount rates and timber prices that might prevail in the future. Forest management plans could then be simulated by considering, not only socio-economic conditions, but also forest resource productivity and the age distribution of stands derived from forest inventory data.

In the present paper we have described an approach that is designed to increase information concerning objective economic and environmental outcomes of forest management such as timber volume (Fig. 6), forestry profits (Fig. 9) and carbon stock (Fig.7), budgets, operability and subsidies (Fig. 8), labor requirements (Fig. 10). Thus, policy makers could use the information from the simulations designed to understand the influence of different carbon price scenarios on local forestry, to select appropriate plans that would meet their management goals. Other simulation results could be used to decide what information should be taken into consideration when deciding whether or not the benefits of a particular management action would justify the costs of its implementation.

Although there are always uncertainties concerning the future state of socio-economic conditions, the present simulation results can at least provide information about any future tendency of estimated values to change over the prediction period in response to the carbon price scenario currently being implemented under the present socio-economic conditions. However, because estimates are prone to errors derived from a dramatic change in the socio-economic conditions that pertain to forestry, such as timber price, carbon price and discount rate, it is important that the actual forest area continues to be monitored in order to check the accuracy of simulations designed to predict future state of forestry. Although our assumptions concerning socio-economic conditions and forest resources were necessarily relatively simple for the preliminary simulation conducted for the present study, as were the patterns of the different subsidy system scenarios, any uncertainty derived from the future changes in socio-economic conditions should be monitored during the management of regional forest resources.

The next challenge is to test the uncertainty of the simulation by monitoring the study area, and to apply the simulation to other forest regions. Depending on the degree of uncertainty and the wider applicability of the simulation, it may be possible to analyze the feasibility of different management strategies and the efficiencies of different subsidy systems according to different regional forest resources, variations in local socio–economic conditions, and diverse forest management goals.

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Section 3

Forest Health

6

Cambial Cell Production and Structure of Xylem and Phloem as an Indicator of Tree Vitality: A Review

Jožica Gričar
Department of Yield and Silviculture, Slovenian Forestry Institute
Slovenia

1. Introduction

One third of Europe's land surface is covered by forests, with important economic and social value. They constitute the most natural ecosystems of the continent. Natural biotic and abiotic disturbances affect their structure and composition. Sustainable forest management and environmental policies rely on the sound scientific resource provided by long-term, large scale and intensive monitoring of forests. Long-living trees and ecosystems are suitable for studying the impact of human factors as opposed to the effects of natural system variability. Forest monitoring helps to improve our knowledge of the state of forests and to quantify changes that are taking place within forests and related ecosystems. Information about forest ecosystem functions and processes is, however, also necessary to gain an understanding of the causes underlying such changes and, subsequently, to model the future effects of natural and anthropogenic stress factors on our forests and understand the adaptation potential of forests to the effects of environmental change and air pollution (UN-ECE, 2008).

The vitality of trees is one of the most important indicators of forest condition and can be characterized by different parameters, such as assessment of the crown condition, tree growth etc. (UN-ECE, 2008). However, the latest studies show that cambium activity and increments of its products – secondary phloem and secondary xylem (wood) – reflect the vitality of trees (Gričar et al., 2009). In the different vitality of silver firs (*Abies alba* Mill.), it has been demonstrated that data on phloem increment structure, the relationship between the phloem and wood increment and the number of cambial cells in the dormant state are very useful in the assessment of the health condition of trees.

This review focuses in particular on presenting the potential of structure and width of xylem, phloem and cambium in the case silver fir and pedunculate oak, as indicators of the vitality status of trees. Forest monitoring and indicators of tree vitality status will be briefly summarized. Growth ring patterns have proved to be an appropriate tool for quantifying the response of a forest stand to changing environmental factors, so wood formation processes that determine the structure of wood and its quality will be described in more detail. Finally, tree vitality has a major influence on wood quality. Two examples, silver fir and pedunculate oak, will therefore be demonstrated.

2. Forest monitoring

Only healthy and vital forests can serve multiple ecological, social and productive roles, as understood by the modern world. To be able to acquire a reliable assessment of the state and changes in forest ecosystems and at least partly to explain and understand the most important processes occurring in them, forest monitoring programmes have been implemented worldwide (e.g., International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests – ICP Forests, Acid Deposition Monitoring Network in East Asia – EANET, Forest Health Monitoring – FHM). Monitoring is essential in order to obtain information about the condition of natural resources, their development over time and space, and to study their relationships with biotic/abiotic factors (Ferretti, 2009). Environmental and nature management, namely, cannot operate effectively without reliable information on changes in the environment and on the causes of those changes. There is therefore considerable concern in the scientific community about the ability of monitoring programmes to provide the desired information (Legg & Nagy, 2006; Vos et al., 2000). Some researchers believe that many operational monitoring programs are not very effective or useful for decision-making (Vos et al., 2000). The main reason for this is poor confidence in the quality of the data, with the most typical questions raised about the statistical basis of sampling design, the reliability and comparability of data and data management (e.g., Ferretti, 2009; Legg & Nagy, 2006; Vos et al., 2000). The results of inadequate monitoring are misleading in terms of their information quality and are dangerous because they create the illusion that something useful has been done (Legg & Nagy, 2006).

Indicators of tree health and vitality need to be accurate and reliable, but also cheap and easy to use (Martín-García et al., 2009). The quality of monitoring is thus defined by its ability to provide data that allow estimates of the status of the target resource with the defined precision level, permit the detection of change with the defined power, and are comparable through space and time. Despite considerable work on data quality control, parts of the monitoring process are still poorly covered by quality assurance and have revealed weaknesses in design and implementation. Steps towards a more comprehensive quality assurance approach have currently been undertaken (Ferretti, 2009).

3. ICP Forests

In Europe, the International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) was established in 1985. The system combines an inventory approach with intensive monitoring. It provides quality assured and representative data on forest ecosystem health and vitality and helps to detect responses of forest ecosystems to the changing environment. Air pollution effects are the particular focus of the programme. ICP Forests uses two complementary monitoring approaches on the European level. Representative monitoring (Level I) is based on around 6,000 permanent observation plots and provides an annual overview of forest condition on the European level. Intensive monitoring (Level II) on around 500 sites provides insight into factors affecting the condition of forest ecosystems and into the effects and interactions of different stress factors. These plots are located in forests that represent the most important forest ecosystems of the continent. The programme provides an early warning system for the impact of environmental stress factors on forest ecosystem health and vitality. Although forest species have responded to environmental changes throughout their evolutionary history, a primary concern in relation to wild species and their ecosystems is the rapid rate of human induced changes (UN-ECE, 2008).

4. Vitality of forest and trees

Though the vitality of trees is one of the most important indicators of forest condition, forest health cannot be assessed solely on the basis of tree condition, since forest consists of more than trees (Innes, 1993). Tree vigour is best restricted as a term to the growth of trees in relation to a hypothetical optimum, whereas tree health is defined in the pathological sense as the incidence of biotic and abiotic factors affecting the tree within a forest. Tree condition is less specific, referring to the overall appearance of trees within the forest. The health of a tree can be evaluated by such indicators as crown condition, growth rate and external signs of disease-causing agents (Kolb et al., 1994).

Tree vitality cannot be measured directly, only through several indicators, such as assessment of the crown condition, growth of bud, stem (radial or height) and root systems, measurements of cambial electrical resistance or the size and shape of needles etc. (Dobbertin, 2005). Shigo (1986) defines vigour as the capacity of a tree to resist strain. It determines the potential strength against any threats to survival. It is genetically derived and cannot therefore be changed. Tree vitality, on the other hand, is the dynamic ability of a tree to grow under the conditions present. It is important to assess the influence of external stress, since resistance to stress is an important criterion in the vitality concept (Dobbertin, 2005). Larcher (2003) defines stress as a significant deviation from the condition optimal for life. Vitality becomes weaker as stress persists. At a certain point, the capacity of a tree to overcome further stress or to survive diminishes, i.e., vitality decreases. Irreversible damage or tree death can occur. The hypothetical optimal tree vitality is not known; only the minimum vitality (i.e., tree death) can be identified (Dobbertin, 2005).

The consequences of tree death, in terms of effects on other ecosystem components and processes, depend on many variables, including the species, mortality agent, position, spatial pattern (dispersed or aggregated) and numbers that have died. Tree death is an important indicator of ecosystem health and can assist recognition of stresses caused by pollutants, such as acid rain and ozone. However, the value of tree death as an indicator of anthropogenic disturbance depends on a thorough understanding of the patterns of tree death under natural conditions. At the present time, adequate understanding of this is woefully lacking. Tree death also demonstrates some principles of ecological processes: the importance of defining the spatial and temporal context of a study, the importance of stochastic processes, and the fact that most ecological processes are driven by multiple mechanisms and that the relative importance of these mechanisms changes over time and space, and the importance of the natural histories of species and ecosystems. Tree death illustrates that many valid and useful perspectives on a single, presumably simple process exist. Furthermore, it makes clear that we need to give more consideration to the biology of organisms and ecosystems in developing, evaluating and applying theoretical constructs (Franklin et al., 1987).

It is not possible to estimate forest health condition from concepts developed at the individual organism level and simply to apply them on a landscape level (Kolb et al., 1994). In other words, extension of this concept to a complex system, such as a forest, is based on making an analogy between the functioning of an organism and an ecosystem. A dead or dying tree is not healthy. The health of a stand must take into account many more dimensions than the health of a tree. The health of a stand relates to the management objectives for that stand (utilitarian perspective) and to the long-term functioning of the organisms and trophic networks that constitute the stand (ecosystem perspective). Tree mortality in a stand would not indicate an unhealthy condition as long as the rate of mortality was not greater than the capacity for replacement. Stand objectives such as wildlife habitat, soil and water protection and preservation of biodiversity do not require that all trees be healthy. A dead tree is not healthy but it may be part of a healthy stand. The health of a forest ecosystem or landscape is similarly more complex than the health of a stand (Kolb et al., 1994).

5. Indicators of tree vitality status

Biochemical indicators on the plant cell level, such as phytohormones or enzymes, may best reflect the reaction of trees to various environmental stresses (Larcher, 2003). Unfortunately, many such indicators cannot readily be extracted in the field or are very expensive. Several indicators, such as assessment of the crown condition, growth of bud, stem (radial or height) and root system, measurements of cambial electrical resistance or size and shape of needles etc. may instead be used (Dobbertin, 2005).

Crown condition is a major indicator of forest health in Europe. The condition of forest trees in Europe is monitored over large areas by a survey of tree crown transparency and discoloration, which is a fast reacting indicator of numerous natural and anthropogenic factors affecting tree vitality. Crown transparency and discoloration is a valuable indicator of the condition of forest trees. It reflects, among other factors, weather conditions and the occurrence of insects and fungal diseases. Such information is extremely relevant for monitoring the reactions of forest ecosystems to climate change and for ensuring sustainable forest management in the future (UN-ECE, 2008).

Crown condition assessments are commonly used in monitoring programs, since they are quick, easy and cheap to do. However, interpretation of these data may be complicated by the occurrence of strong fructification years in some tree species, when the foliage is reduced (Beck, 2009). In addition, tree crown transparency and discoloration used to be visually estimated by observers from the ground, raising the questions about the subjectivity of human assessment, data quality and comparability across the countries (Mizoue & Dobbertin, 2003). These issues were tried to be solved by combining field and control team assessments, using data from cross-calibration courses to estimate correction factors, using reference photographs or standard sets of two-dimensional silhouettes representing various degrees of foliar density (Frampton et al., 2001; Ghosh & Innes, 1995; Innes et al., 1993; Solberg & Strand, 1999). Nevertheless, since these improvements were sometimes not enough, the researchers started to replace visual ground assessment by digital photo (Martín-García et al., 2009; Mizoue, 2002) or by remote sensing techniques (Coops et al., 2004; Stone et al., 2003)

Moreover, it is not possible simply to conclude tree vitality by crown condition. Growth rates can be considerably reduced while foliage is still inconspicuous (Beck, 2009). In particular, the relation between crown condition and xylem increment in trees has not been satisfactorily explained. In principle, physiological investigations of these relationships through case studies may be useful for improving our understanding.

As summarised by Kozlowsky & Pallardy (1997), the requirements for tree growth are carbon dioxide, water, and minerals for raw materials, light as an energy resource, oxygen and favourable temperature for growth processes. The capacity of photosynthetic processes (i.e., foliar biomass) and competition for resources are constraining factors for tree growth. Tree growth processes can be ranked by order of importance as foliage growth, root growth, bud growth, storage tissue growth, stem growth, growth of defence compounds and reproductive growth (Waring, 1987; Waring et al., 1980). Under stress, photosynthesis is reduced and carbon allocation is altered. Stem growth may be reduced early on, since it is not directly vital to the tree. Comparison with a suitable reference is important for any potential vitality indicator. Depending on the aim of the study, the references used can be the growth of trees presumed to be without stress. The general disadvantage is that no absolute growth reference is available. Some stresses, such as competition, root rot or mistletoe occurrence, affect the tree over extended time periods, whereas other stresses, such as drought or insect defoliation, cause immediate reactions. Annual or inter-annual stem growth assessment is therefore needed in long-term monitoring plots. Tree growth can serve as a vitality indicator if a reference growth or growth trends are available (Dobbertin, 2005). It is noteworthy that not every stress is necessarily negative for trees but can instead induce increased resistance to stress (Kozlowsky & Pallardy, 1997; Larcher, 2003). A short-term stress reaction may therefore not coincide with a long-term change in tree vitality. Growth changes must thus be interpreted on a long-term perspective (Dobbertin, 2005).

6. Growth ring patterns as indicators of tree health

Beck (2009) emphasized dendroecological analysis of tree and stand growth patterns as an appropriate tool for quantifying the response of a forest stand to changing environmental factors. Tree growth parameters, which reflect changing growth conditions year by year, are very important. Such parameters of tree vigour presented as a time series retrospectively enable an insight into the growth history of the stand. Namely, tree-ring analysis can provide information on trees and stand development in the past. The growth of trees and site history of a stand can be reconstructed using tree-ring time series, which contain lots of information on environmental conditions and their impact on the growth of trees. Wood formation is the final result of the complete metabolic balance. It is the share of the balance of matter produced by the foliage, respiration and the higher priorities of allocation to other tree organs (roots, fruits etc.), which has not been consumed elsewhere. This remaining share refers directly to the state of the reserve pool. The amount of new wood formation can therefore be understood as a suitable tree health indicator. In contrast to visually assessed crown condition, tree-ring widths are measured (qualitative) data. Subjective estimation is thus eliminated (Beck, 2009).

Tree growth rates are affected by both pollutants and climatic stress. In view of this complexity, a comprehensive dendroecological analysis of tree and stand growth patterns is considered to be an appropriate tool for quantifying the response of a forest stand to changing environmental factors. The inclusion of dendroecological methods in monitoring programs provides many advantages and new findings. The elaboration and analysis of tree ring networks (chronologies well scattered with respect to space and altitude) are currently seen as successful fields of ecological research. Ongoing depositions and increasing climatic stress urgently require the quantification of the growth response of forests as an indicator of tree and stand vigour (Beck, 2009).

Decreasing growth curves are among the most obvious growth-related characteristics of dying trees, which is not only a species-specific but also a site-specific feature. A combination of growth levels or relative growth and growth trends has been shown to increase the reliability of mortality predictions. Abrupt declines in growth or strongly negative growth trends may indicate a rapid physiological adaptation to changed environmental conditions (Bigler et al., 2004). Reduced xylem increments, as one of the first indicators of decreased tree vitality, are very useful for reconstruction of past tree vitality and evaluation of mortality risk (Bigler & Bugmann, 2004; Bigler et al., 2004). These assessments of individual tree vitality and accurate mortality predictions may be used in forest management to identify and selectively cut low-vitality trees, so as to release the remaining healthy trees (Bigler et al., 2004).

7. The role of cambium in a tree

The growth of trees, leading to an increase in the size and mass of an organism, occurs only in specific areas, so called meristems, and involves cell division, expansion and differentiation. Cell division is an essential part of growth, resulting in an increase in the number of cells. In the expansion phase, cells increase in size. The cytoplasm grows and the vacuole fills with water, which exerts pressure on the cell wall and causes it to expand. In the next step, cells differentiate, or specialize, into various cell types. A tree is composed of various cell types that perform different functions required in a multicellular organism (Berg, 2008).

Meristems consist of actively dividing undifferentiated cells, which retain the capacity for growth through their entire lifespan. Two kinds of meristematic growth occur in trees; primary or extensional and secondary or lateral (Berg, 2008). Primary growth occurs as a result of the activity of apical meristems, which are located at the tips of stems and roots and lead to an increase in a tree's length and development of various tissues: epidermis, cortex, conducting veins, pith and leaves (Fig. 1) (Mauseth, 1988). In woody plants, even in the first year of growth the primary tissues of stems and roots are replaced by secondary tissues formed by the secondary lateral meristems: vascular cambium (in short cambium) and cork cambium (phellogen). The activity of secondary meristem is expressed as radial growth, which allows the increase of the volume of the conducting system and the formation of mechanical and protective tissues (Plomion et al., 2001; Taiz & Zeiger, 2002). Cambium produces secondary vascular tissues, which conduct water and nutrients and provide support. Cork cambium produces protective tissue (periderm), which protects the stem and root from water loss, pathogens, and herbivorous insects (Larson, 1994; Panshin & de Zeeuw, 1980).

The cambium, as an uninterrupted, thin layer ring, lies between the secondary xylem on the inner side and secondary phloem on the outer side, the two tissues it produces (Larson, 1994; Mauseth, 1988). It has been called the "least understood plant meristem", because of the associated technical difficulties when working with trees (Groover, 2005).

The cambium consists of a layer of cells that divide actively, have small radial dimensions and have no intercellular space (Savidge, 2001). It differs from other meristems by two types of highly vacuolised cells: short, rather isodiametric ray cells, from which radial rays are formed, and elongated fusiform or spindle-shaped cells, which form axial elements (Fig. 2). New cambial cells are formed by anticlinal divisions, which ensures an increase in girth of the cambium. The cambium has a decisive role in radial growth and the development of trees, since new vascular tissues of xylem and phloem are formed through periclinal or additive divisions that occur in the tangential plane, by which the diameter of the tree increases. The production of secondary conductive tissues represents approximately 90% of all mitoses (Lachaud et al., 1999; Larson, 1994).

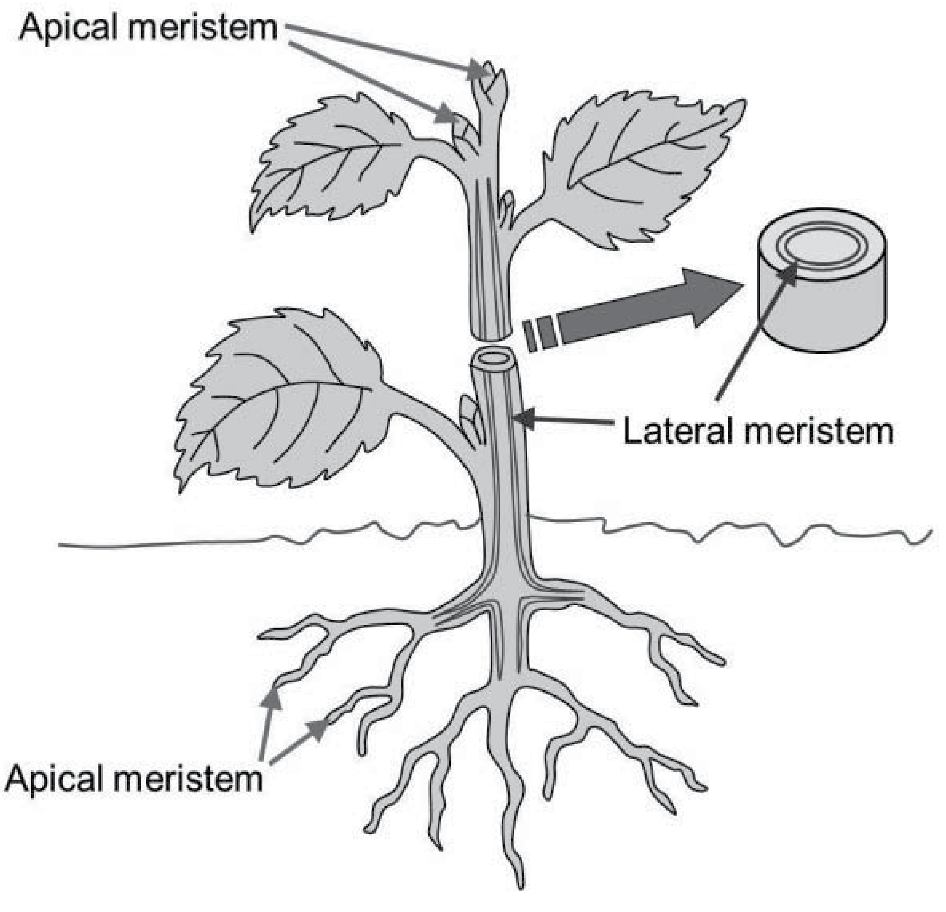

Fig. 1. Location of apical and lateral meristems in a plant.

A characteristic of tree species in the temperate climatic zone is a seasonal alternation of cambial activity and dormant (resting) periods, which is generally related to alternations of cold and hot or rainy and dry seasons (Larcher, 2003). Cambial activity usually starts in spring with cell division and ends in late summer with the completed development of the latest newly formed cells (Fig. 2). At the beginning of cambial activity, the number of cambial cells increases and they start to divide, which is followed by differentiation of derivatives into the adult elements of xylem or phloem. In the process of differentiation, which includes post-cambial cell growth, deposition of the secondary cell wall and – in wood tracheids, fibers and vessels – also lignification and programmed cell death, the cells specialize in order to perform their functions (Fig. 3) (Plomion et al., 2001). The vascular system in trees is very complex, composed of various types of cells, which are differently orientated (Chaffey, 2002).

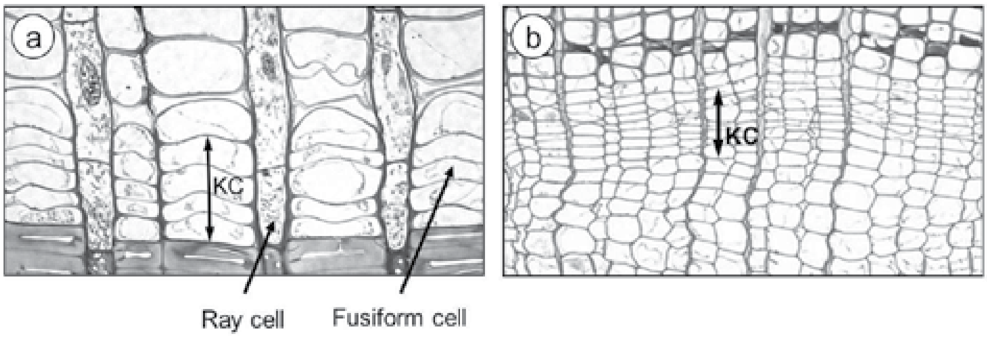

Fig. 2. (a) Dormant and (b) active cambium. KC – cambial cells

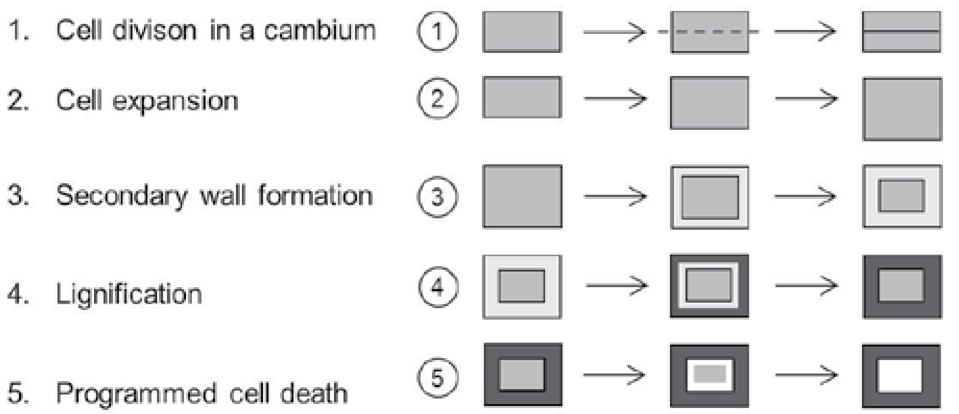

Fig. 3. Schematic illustration of formation of tracheid from cambial cell in conifers.

8. Wood and phloem formation

Xylo- and phloemogenesis are periodic processes driven by a variety of internal and external factors, the influence of which changes during the growing season. Xylem and phloem increments are not predetermined, but are plastic end-products of interactions between the genotype and the environment (Savidge, 2001). The environment determines the physical conditions and the energy for xylo- and phloemogenesis. The external factors affect the onset, end and rate of individual growth processes, which determine the morphology of cells (Wodzicki, 2001). Xylo- and phloemogenesis lead to specialization of cells in terms of their chemical composition, morphological characteristics and function. Cell divisions in the cambium and post-cambial growth determine the width of the annual xylem and phloem increment, and the deposition of the secondary cell wall (and lignification) determines the accumulation of biomass in the walls of the xylem and phloem cells (annual biomass increment) (Fig. 3) (Plomion et al., 2001).
The number of dormant cambial cells depends on several factors; such as tree species, tree age, part of the tree, and tree vigour and vitality. The cambium's cell production, under

normal growth conditions, is more intensive on the xylem than on the phloem side. However, under physiologically very demanding conditions, the phloem increment can exceed the xylem one, which may not appear at all in exceptional cases (Larson, 1994; Panshin & de Zeeuw, 1980).

Of all the secondary tissues, xylem and its formation is by far the most investigated, particularly due to its great economic and ecological importance. The width and structure of xylem growth rings is a source of information about past and present factors affecting the development processes in an individual tree (Fritts, 1976; Wimmer, 2002). The width of the xylem increment is closely related to its anatomical structure, which defines the physical and mechanical properties of wood and, consequently, its end-use (Fig. 4).

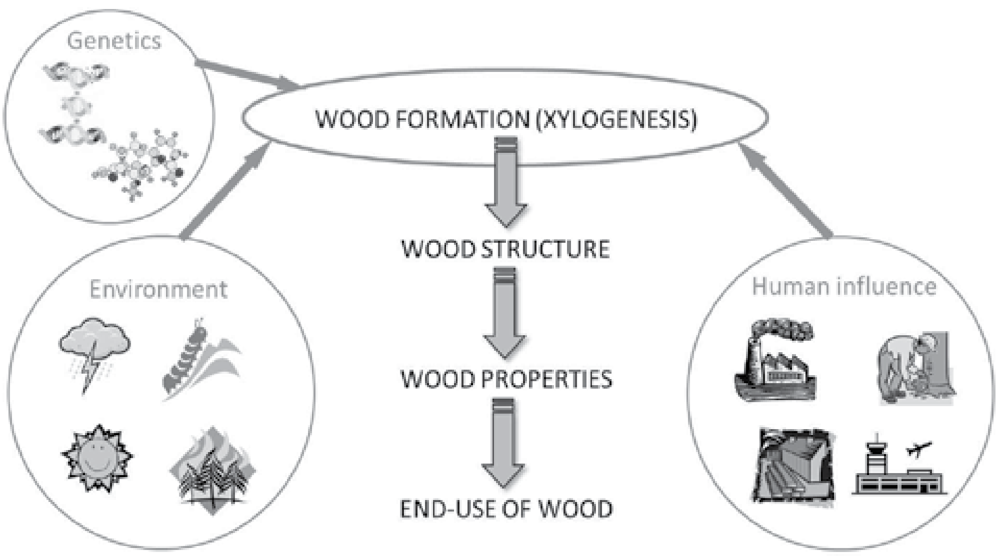

Fig. 4. Xylogenesis is affected by a variety of internal and external factors, which influence wood structure and properties and, consequently, the end-use of wood.

Studies of the seasonal dynamics of phloem growth rings are fewer, which can be partly explained by a lower interest in the commercial use of bark in comparison to the use of timber. In addition, the phloem increment is exposed to relatively fast secondary changes of the tissue, e.g., collapse, sclerification and inflation of axial parenchyma, so only the structure of one or two of the most recent phloem growth rings can be seen clearly. Older non-conducting tissue eventually collapses in a radial direction, deforms and later often also falls off and is thus not suitable for dendrochronological and dendroecological studies (Gričar, 2009).

Nevertheless, the seasonal dynamics of phloem formation is very important in studies of trees' radial growth because cambium is a bi-facial meristem, so studies of cambial activity and wood formation reveal only part of the information on cambial cell productivity during the growth season. Moreover, the processes of wood and phloem formation differ in terms of time and space, and internal and external influences affect the mechanisms of their

formation differently. Phloem increment is more stable and less subjected to fluctuations of environmental conditions. Comprehensive studies are therefore vital for investigating the influence of specific climatic factors on the radial growth of trees.
The impact of changing environment will modify the seasonality and rate of growth, which can have an important effect on tree performance and survival and also on wood structure and properties and, consequently, on the end-use of wood.

9. Wood density in relation to ring widths

Wood formation determines the morphology of cells, the structure of the xylem growth ring and thus the wood properties. Xylem rings are composed of early wood and late wood. Early wood cells are formed at the beginning of the growing season and are characterized by a large radial dimension and thin cell walls. The development of late wood cells with small radial dimensions and thick cell walls occurs in summer, resulting in its higher density. In sapwood, the ratio between the density of early wood and late wood is 1: 2.3 in fir and 1: 4.0 in pine (Gorišek, 2009). In ring-porous deciduous tree species this ratio is about 1: 2.5 and in diffuse-porous trees much smaller, e.g., 1: 1.5 in beech (Gorišek, 2009).
Wood is heterogeneous material composed of various types of cells that perform different functions. Consequently, the density of wood is related to the morphological characteristics of the cells. Growth of trees from temperate climate regions is seasonal resulting in the formation of growth rings. At the beginning of the growing season the dominant function appears to be conduction, while in the second part of the season is support. Early wood cells have therefore thinner cell walls and bigger cavities than late wood ones. Hence, the greater is the proportion of late wood the greater are the density and strength. However, wood density and its strength are influenced by the ring width (Fig. 5) (Dinwoodie, 1981). This relationship is relatively complex; in ring-porous tree species, such as oak and ash, increasing ring width results in an increase in the percentage of late wood, which contains most of the fibres and, consequently, the density will increase (Fig. 6). In diffuse-porous tree

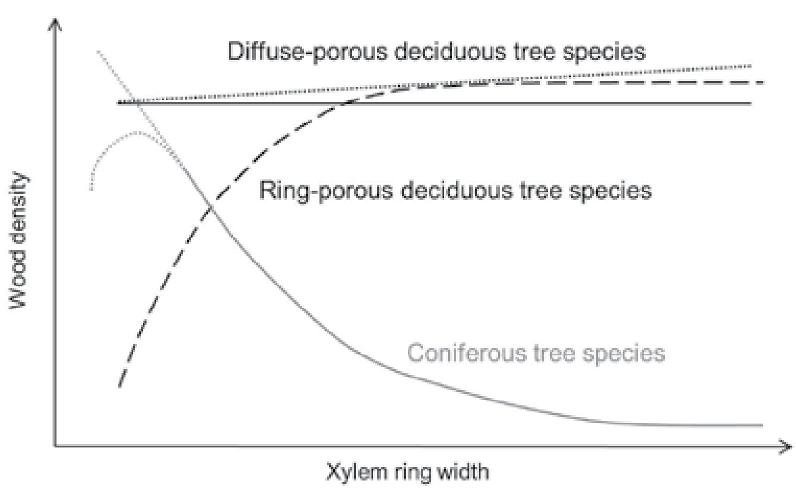

Fig. 5. Impact of xylem ring width on the density of wood in ring-porous and diffuse-porous deciduous trees and conifers.

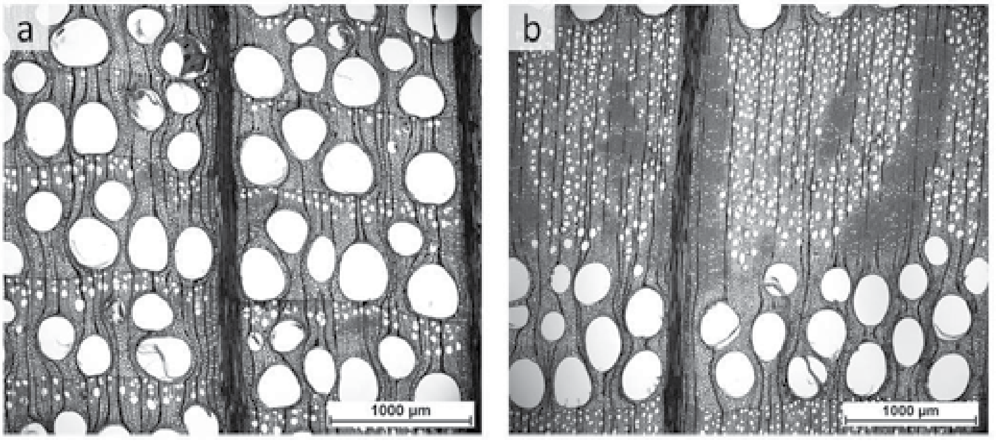

Fig. 6. Wood density is closely related to ring width in ring-porous oak; non-vital (a) and vital oak (b).

species (beech and maple, for example), in which the wood anatomical structure in the xylem ring is relatively homogenous, increasing ring width has almost no effect on wood density. In softwoods, however, increasing ring width results in an increased percentage of low-density early wood and, consequently, a decrease in density (Fig. 7). Exceptionally, softwoods from very cold areas may have narrow rings with low density, because late wood formation is restricted by the short summer period (Dinwoodie, 1981).

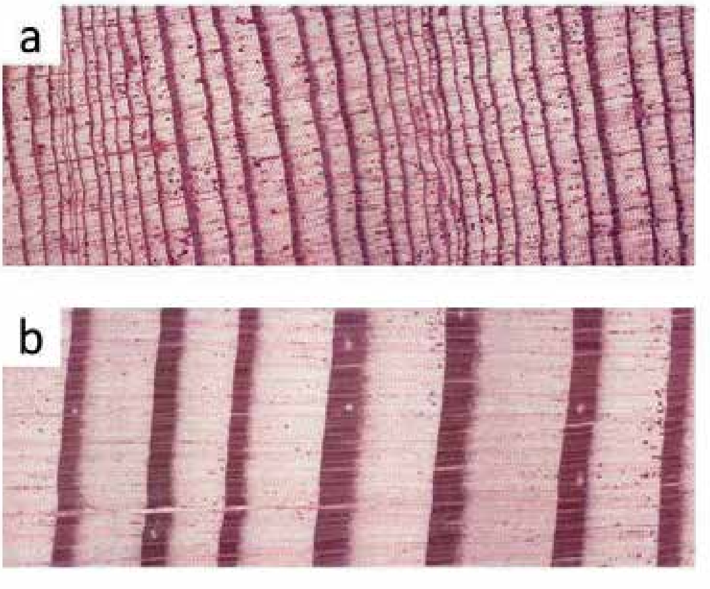

Fig. 7. Widths of xylem rings in European larch, with different proportions of early wood and late wood.

Of the wood properties that affect quality, basic density is one of the most important because it determines its utilization in sawmills, manufacturing factories, cellulose plants and as planks. Several factors influence the variability in basic density: site, climate, geographic location, species, age and silviculture. However, tree vitality also has a major effect on wood quality and properties; the relationship among all this parameters therefore deserves deeper investigation.

10. Relationship among the number of cells in xylem, phloem and dormant cambium in silver fir (*Abies alba* Mill.) trees of different vitality

Silver fir *(Abies alba)* decline has appeared in many European countries, including Slovenia, since about 1500. The exact cause of silver fir decline is still not satisfactorily explained; however, it has been interpreted as a complex disease due to the interaction of several unfavourable factors, such as drought, frost, pollution, competition among trees, soil acidification, inappropriate silvicultural treatments, insects, pathogens etc (e.g., Bauch, 1986; Dobbertin, 2005; Fink, 1986; Schweingruber, 1986; Torelli et al., 1986).

Decline is characterized by reduced cambial production, especially towards the xylem, shorter cambial activity and crown damage, including needle loss and yellowing foliage (Fig. 8, 9) (e.g. Bauch, 1986; Fink, 1986; Innes, 1993; Schmitt et al., 2003; Schweingruber, 1986; Torelli et al., 1999). Reduced wood formation often occurs prior to visual symptoms of crown decline (Torelli et al., 1986, 1999), which highlights the usefulness of assessment of a tree’s current mortality risk based on growth patterns and a derived statistical mortality model that clearly identifies trees at high risk of dying (Bigler et al., 2004).

Fig. 8. Crowns of differently vital silver firs.

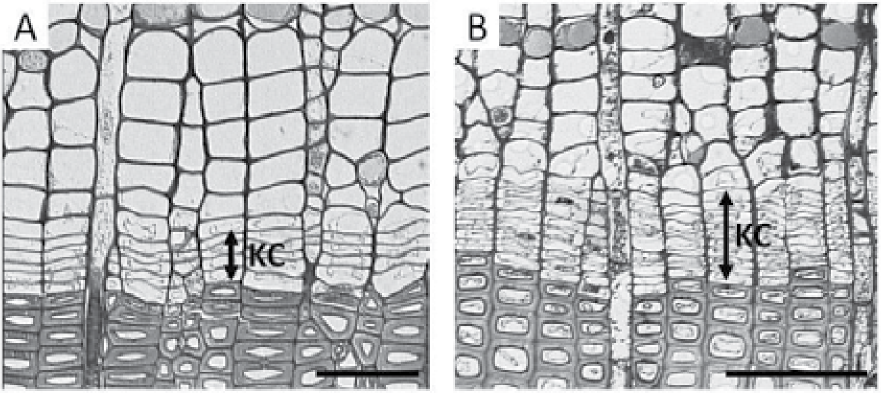

Fig. 9. Narrow cambium in declining silver fir, consisting of four to five cell layers (a) and a wide cambium in healthy silver fir consisting of about ten cell layers. KC – cambial cells, Scale bars = 50 µm (A), 100 µm (B)

Although some papers have been published concerning the anatomical structure and dynamics of secondary phloem formation (e.g., Gričar & Čufar, 2008), the relationship among the number of cells in phloem, xylem and dormant cambium is still poorly understood. In a paper published in 2009, we investigated the anatomical structure of phloem and xylem growth rings, as well as dormant cambium in relation to vitality in 81 adult silver fir trees (*Abies alba* Mill.) (Gričar et al., 2009). Specifically, we investigated the number of cells produced in the current phloem growth ring, xylem growth ring and their ratio, the number of cells in the dormant cambium and the structure of the phloem growth ring, which included characterization of early phloem, late phloem and the presence, absence and continuity of tangential bands of axial parenchyma.

The silver fir (*Abies alba* Mill.) trees were located in an *Abieti-fagetum dinaricum* mixed forest at Ravnik, Slovenia (approx. 45°52′N, 14°16′E, elevation 500-700 m). The studied trees were dominant or co-dominant, with an age of 150–180 years and DBH greater than 50 cm. The trees belonged to a population of 269 mature trees monitored from 1987 to 2007. The health condition of the trees was assessed by determining the crown status index based on progressive needle loss and cambial electrical resistance (CER) (Torelli et al., 1999). Trees were assigned to 3 categories: A – trees with a full crown and productive cambium; B – trees with intermediate characteristics and C – trees with a sparse crown and suppressed cambium (Fig. 8, 9, 10). For the study, we used microscopic slides of 81 trees of different vitality. Sample blocks (0.5 x 0.5 x 1 cm) contained inner phloem, cambium and outer xylem taken from living trees at 1.3 m above ground during the dormant seasons of 1999 to 2003. We used observations of transverse sections using light microscopy.

Microscopic examination of cross-sections revealed that the trees could be classified into three groups on the basis of the ratios between the number of cells in the xylem and phloem growth rings (Table 1). Group 1 (43% of the trees) contained trees with up to four times more cells in the xylem ring than in phloem ones. The trees in group 2 (30% of the trees) had a ratio between xylem and phloem ring from 4.0 to 10.0, and group 3 (27% of the trees) consisted of trees with a ratio between xylem and phloem ring greater than 10.0 (Gričar et al., 2009).

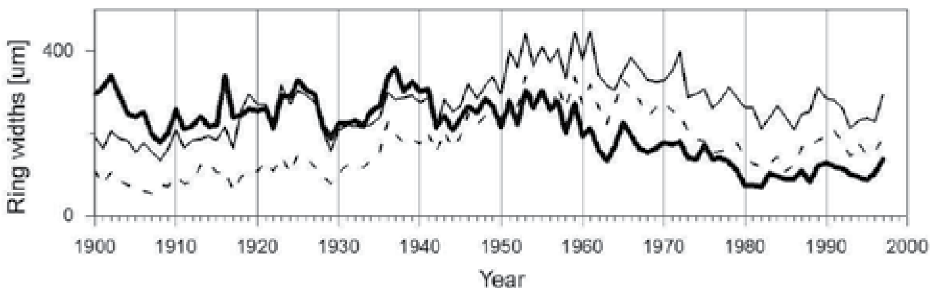

Fig. 10. Xylem ring widths of silver firs of different vitality. Category A (thin line), B (dashed line) and C (thick line) (Archives of the Chair of Wood Technology, Department of Wood Science and Technology, Biotechnical Faculty, University of Ljubljana) (Gričar, 2006).

Group	Ratio XR:PR	No. of trees (%)	No. of cell layers		Structure of PR
			XR	PR	
1	<4.0 : 1	34 (43%)	3-26	3-7	AP missing or discontinuous, EP 1-5 or > 5 cells wide, LP 1-3 cells wide or absent
2	(4.0-10.0) :1	24 (30%)	25-80	5-9	AP discontinuous or continuous, EP 2-4 cells wide, LP present
3	>10.0 :1	23 (27%)	60-144	6-12	One or two bands of AP

Table 1. Characteristics of tissues in three groups of trees with different ratios between xylem and phloem growth ring widths in terms of number of cells (XR:PR). XR – xylem growth ring, PR – phloem growth ring, AP – axial parenchyma, EP – early phloem, LP – late phloem (Gričar et al., 2009)

We confirmed that the structure and width of the phloem are closely related in silver fir (Fig. 11). Early phloem is in general 2-5 layers of cells wide and is less dependent on tree vitality whereas late phloem is subject to higher alterations in the width and type of cells. The occurrence and amount of axial parenchyma varies in accordance with the width of the phloem ring: a) it can be absent or scarce when rings are very narrow; b) present as one, more or less continuous, tangential band between early phloem and late phloem, as observed in the majority of phloem rings; or c) also forming an additional, second, discontinuous tangential band in the late phloem of very wide rings. The cambium of vital trees normally produces more xylem than phloem cells. The ratio between xylem and phloem declines with decreased vitality of trees. Only in extreme cases can the phloem ring be wider than the xylem one. The numbers of cells in phloem, xylem and dormant cambium are correlated in silver fir. Information on the width and structure of phloem rings, as well as on the relation between xylem, phloem and dormant cambium could provide additional criteria for determining tree vitality (Gričar et al., 2009).

Inspection of the current condition of the investigated trees revealed that more than half of the trees (62%) with a ratio between phloem and xylem increments lower than 4:1, with a very narrow xylem (about 20 cell layers) and phloem only 3-5 cell layers wide, died in the years following the sampling of tissues for our analyses. Our results suggest that the ratio between xylem and phloem, as well as the widths of xylem, phloem and dormant cambium, are related and indicate the health condition of a tree. They could therefore be used for assessment of the vitality of silver firs. This information could be beneficial in forest management practice, for planning the cutting of non-vital trees with poor survival prognosis and for identifying and promoting healthy and productive ones (Gričar et al., 2009).

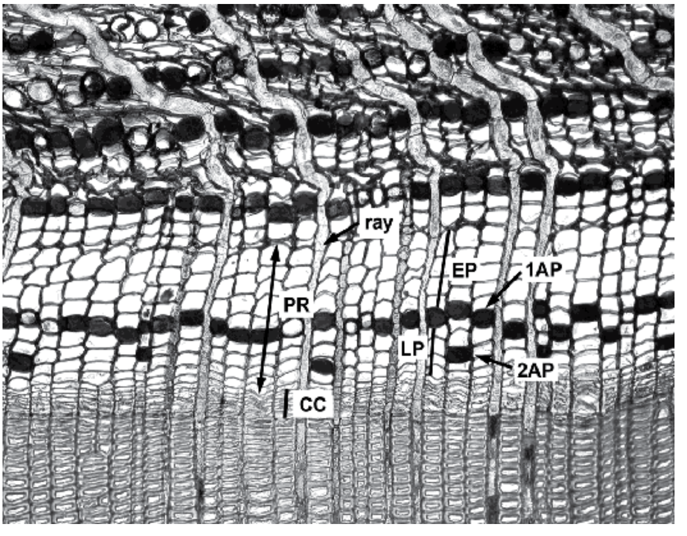

Fig. 11. Cross-section of a phloem growth ring in silver fir. PR – phloem ring, EP – early phloem sieve cells, LP – late phloem sieve cells, 1AP - first band of axial parenchyma, 2AP - second band of axial parenchyma in late phloem, CC - cambium

Since the study in silver firs gave fairly encouraging results, we tested whether similar relations can also be found in ring-porous species, such as pedunculate oak (*Quercus robur*).

11. Cambial productivity and widths of xylem and phloem increments in pedunculate oak (*Quercus robur* L.) trees of different vitality

In Slovenia, oaks (*Quercus robur* L. and *Quercus sessiliflora* Salisb.) are economically and ecologically very important wood species and represent about 7% of the entire wood stock (Zavod za gozdove Slovenije, 2010). In the case of pedunculate oak, the lowland forest area has been shrinking, due to human settlement in the past, intensive and unplanned silvicultural and agricultural exploitation of the land and conflicts of interest, so only a few lowland oak forest stands have managed to survive (Kadunc, 2010; Žibert, 2006). Similarly as in many European countries, a trend of decreasing vitality of pedunculate oak has been observed in most sites in recent decades. In 2007, pedunculate and sessile oak had the highest share of damaged and dead trees; i.e., 35.2% of the analysed tree species. The highest defoliation of pedunculate and sessile oak was observed in 2005. The condition of these species is characterised by some recuperation in 2006 and another increase in 2007 (UN-ECE, 2008).

One of the main reasons for decreasing vitality of pedunculate oak in Slovenia is ascribed to a lowering of the ground water level due to changes in climatic conditions and unsuitable artificial melioration of land for agricultural purposes (Kadunc, 2010). Namely, numerous drainage ditches were excavated in 19th century. The most obvious response of oaks to the changing environmental (hydrological) conditions is seen in the reduced wood increment, which is closely related to the structure of wood and its quality (Fig. 12).

Oak wood is considered to be very aesthetic due to its specific anatomical structure (texture) and colour. Since it also has good mechanical and durability properties, it is used for high value sawn wood products. A major factor in the utilization of wood is the degree of variation of wood properties at different scales. Variations are the result of site-to site differences in wood, population-level differences within a site and within a single tree (Gasson, 1987; Leal et al., 2007, 2008; Lei et al., 1996; Panshin & de Zeeuw, 1980; Zhang, 1997).

In addition to major economic consequences in these areas, the ecological issues associated with the decreasing vitality of pedunculate oak stands cannot be neglected. From a physiological point of view, wood tissues in trees perform several functions simultaneously, of which the two most important are to provide mechanical support and water transport. Different cell types, their morphological characteristics and their proportion in the xylem growth ring affect the survival and efficiency of the living tree. Vessel diameter, area and percentage conductive area strongly influence the amount of water that can be transported in the living tree, and so the larger the proportion of the ring occupied by conductive elements, the less tissue is available for supporting, strengthening and storage. Any changes in the proportion among different cell types therefore very likely modify the hydraulic and mechanical properties of wood (Tyree & Zimmermann, 2002).

The relationship among number of cells in phloem, xylem and dormant cambium in oak is still poorly understood. We have hypothesised that the structure and width of the phloem increments, the ratio between the phloem and xylem increments and the width of the dormant cambium would reflect the health condition of the tree. More vital trees are expected to have much wider xylem than phloem increments, whereas in declining trees, the ratio between xylem and phloem will decrease. For that purpose, we investigated the width of the phloem growth rings, late phloem, xylem growth rings, late xylem, as well as the number of cells in the dormant cambium, in 80 adult pedunculate oaks (*Quercus robur* L.) of different vitality. The health condition of the oaks was defined according to the crown condition and the width of the xylem increment.

Oak trees of various vitality were sampled at a *Pseudostellario-Carpinetum* mixed forest in Krakovo, Slovenia (45°54′N, 15 25′E, elevation 150 m). Krakovo is the largest lowland oak forest in Slovenia, which is flooded by the Krka River. It is dominated by *Quercus robur*, *Carpinus betulus* and *Alnus glutinosa* tree species. Sampled trees were dominant or co-dominant with diameter 50-60 cm, height 25-30 m and age above 80 years. In December 2009, we took micro-cores (2.4 x 2.4 x 20 mm) containing inner phloem, cambium and outer xylem taken from living trees at 1.3 m above the ground. The material extracted from the trees was immediately fixed, dehydrated in a graded series of ethanol and embedded in paraffin (Gričar, 2006). Observations and analysis were made on transverse sections using light microscopy.

The anatomical structure and widths of the phloem and xylem increments are closely related. In ring-porous oak, increasing ring width results in an increase in the percentage of late wood and late phloem, respectively (Fig. 12). In both cases, the widths of early phloem and early xylem were relatively stable as the widths of the phloem and xylem rings changed, whereas late phloem and late xylem were quite variable and increased with ring width (phloem R2 = 0.597; xylem R2 = 0.955) (Fig. 13). Other researchers have also found that the late wood portion in oak tends to increase with increased ring width, whereas the width of early wood is more or less constant (Phelps & Workman, 1994; Rao et al., 1997; Zhang, 1997). The reason is in their completely different anatomical structure and, consequently, their densities, which are much higher in late wood (ca 800 kg/m^3) than in early wood (ca 560 kg/m^3) (Guilley et al., 1999). Namely, the diameter of late wood vessels is much smaller and the proportion of fibers is higher. The total ring density of oak is influenced by variation in the late wood structure and by changes in the proportion of late wood to early wood.

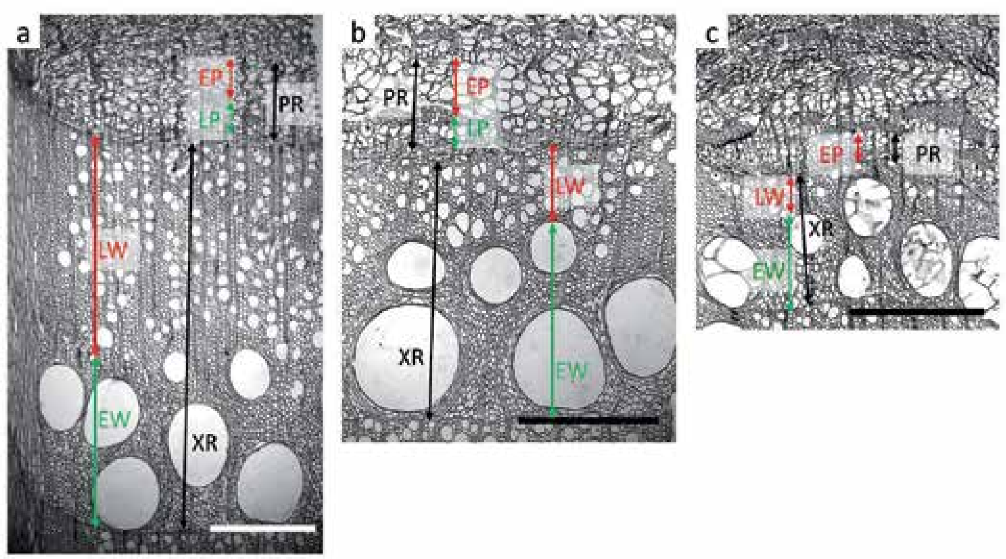

Fig. 12. Ring width and wood structure in ring-porous oaks of different vitality are closely related. XR – xylem increment, EW – early wood, LW – late wood, PR – phloem increment, EP – early phloem, LP – late phloem, Scale bars = 500 µm

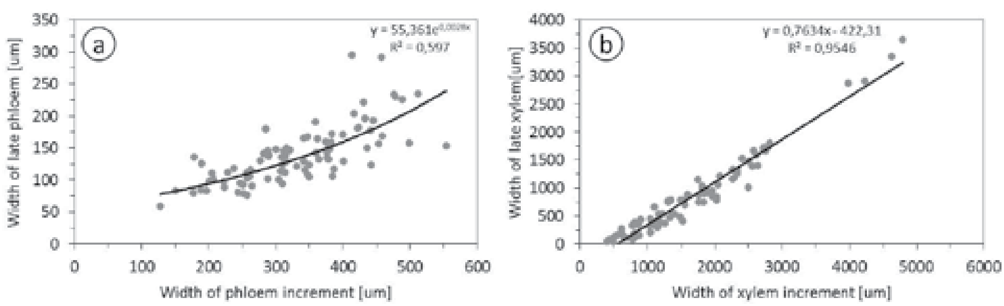

Fig. 13. Relationship between: (a) widths of late phloem and phloem increments and (b) widths of late xylem and xylem increments.

The cambium of vital trees normally produces more xylem than phloem cells. In trees with diminished vitality, xylem production is reduced and, consequently, the ratio between the xylem and phloem increment becomes progressively smaller. Only in extreme cases can the phloem increment be wider than the xylem one (Larson, 1994; Panshin & de Zeeuw, 1980). Of 86 sampled trees, the ratio between phloem and xylem in 40% of them was from 9% to 20%, in 50% of sampled oaks the ratio between phloem and xylem was from 21% to 40%, and only in 4 trees was the ratio higher than 50%. The xylem increments in this case were narrow; i.e., below 1000 μm. We found a high negative correlation between the ratio of phloem to xylem and the width of xylem increment (R2 = 0.724), whereas no such relation was found with the width of the phloem increment (Fig. 14). However, unlike in the case of silver fir, the phloem increment in pedunculate oak was smaller than the xylem one in all sampled trees.

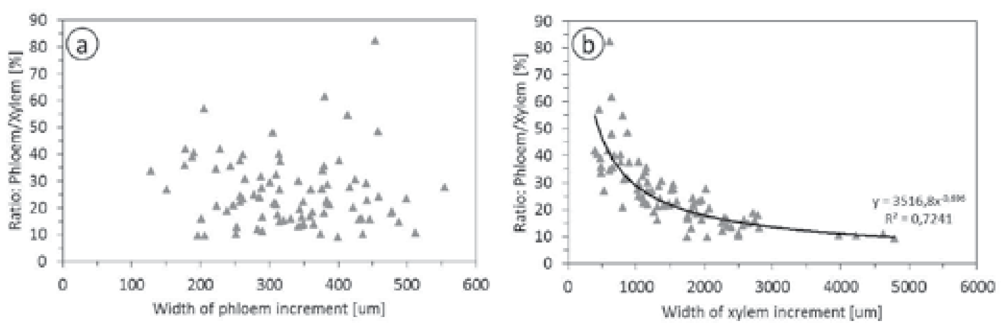

Fig. 14. Ratio between phloem and xylem increments in relation to phloem (a) and xylem (b) ring widths.

The widths of phloem and xylem and the number of cells in the dormant cambium were shown to be positively correlated. The variability in the widths of increments was higher in the xylem (400–4660 μm) than in the phloem (155-518 μm). In 25% of sampled oaks, the xylem increment was 400-1000 μm wide, in 44% 1000-2000 μm wide, in 24% 2000-3000 μm wide and in 7% 3000-4660 μm wide. In the case of phloem, 37% of oaks had an increment from 160-300 μm, 35% of them from 300-400 μm and 28% from 400-518 μm. We found a positive relationship between the width of phloem increments and the number of cells in the dormant cambium (R2 = 0.320), between the width of xylem increments and the number of cells in the dormant cambium (R2 = 0.561) and between the width of phloem and xylem increments (R2 = 0.351) (Fig. 15). The highest correlation was thus between the xylem increment and the number of cells in the dormant cambium.

We can summarize that:

- The widths of phloem and xylem increments and the number of cells in the dormant cambium are correlated in pedunculate oak.
- The widths of early phloem and early xylem are less dependent on tree vitality; whereas late phloem and late xylem are subject to higher alterations in width.
- The cambium of oak produces more xylem than phloem cells.
- The ratio between phloem and xylem increment declines with decreased vitality of trees.

The study on ring-porous pedunculate oak therefore confirms the findings obtained from coniferous silver fir trees. Information on the width and structure of xylem and phloem increments, as well as the number of cells in the dormant cambium, could indeed provide additional criteria for determining tree vitality.

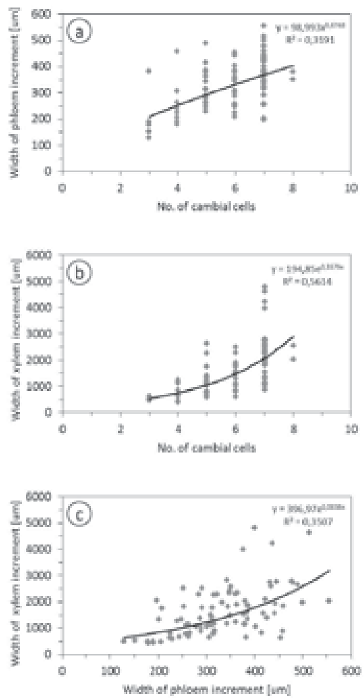

Fig. 15. Relationship between: (a) width of the phloem increment and number of cells in the dormant cambium, (b) width of the xylem increment and number of cells in the dormant cambium and (c) width of the phloem increment and width of the xylem increment in oak.

12. Conclusions

The anticipated environmental change is one of the main factors threatening the health condition of the economically most important tree species; economically essential forest stands are therefore potentially endangered. Knowledge of a species' growth characteristics and the effect that climatic variables and silvicultural management decisions have on tree growth is obviously a key issue for assessing and preserving the sustainability of forests (UN-ECE, 2008). Radial growth patterns have been shown to be valuable indicators of tree health condition. The width and structure of xylem growth rings are a source of information about past and present factors affecting development processes in an individual tree. Tree vitality also has a major effect on wood quality and properties. Information on the width and structure of xylem and phloem increments, as well as the number of cells in the dormant cambium could provide additional criteria for determining tree vitality. Indicators of tree health and vitality need to be accurate and reliable, but also cheap and easy to use. The proposed indicators comply with these desirable characteristics. Since vital forest resources are the basis of sustainable forest management and the production of quality timber, early identification of trees with an increased risk of dying could help assess and manage the health condition of forest stands in the future.

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7

Evaluating Abiotic Factors Related to Forest Diseases: Tool for Sustainable Forest Management

Ludmila La Manna
Centro de Investigación y Extensión Forestal Andino Patagónico
Universidad Nacional de la Patagonia San Juan Bosco, CONICET
Argentina

1. Introduction

The influence of abiotic factors in the development of a disease is recognized in plant pathology. An abiotic factor may be the direct cause of a disease or may determinate the importance of an infectious disease or may be a key factor in forest decline diseases. Numerous studies have related forest diseases with abiotic factors around the world and for different forest species (Baccalla et al., 1998; Bernier & Lewis, 1999; Demchick & Sharpe, 2000; Dezzeo et al., 1997; Hennon et al., 1990; Horsley et al., 2000; Maciaszek, 1996).

Statistical techniques coupled with geographical information systems have fostered the development of predictive host habitat distribution models. The habitat-association approach can be used to generate risk maps, an important tool for developing forest management criteria (Fernández & Solla, 2006; Meentemeyer et al., 2004; Van Staden et al., 2004; Venette & Cohen, 2006). Many techniques with varying complexity were developed: rule based habitat models (Schadt et al., 2002a), niche modeling (Meentmayer et al., 2008, Rotemberry et al. 2006), neutral landscape models (With, 1997; With & King, 1997), etc.

This chapter aimed to describe some usefully methods for evaluating abiotic factors in relation to forest diseases at landscape scale and for developing risk models as tool for forest management. The methods described in this chapter were used for modeling *Phytophthora* disease risk in *Austrocedrus chilensis* [(D. Don) Pic. Serm. & Bizzarri] forests of Patagonia (La Manna et al., 2008b, 2012).

2. Collecting information

The predictive ability of a risk model is strongly associated with the quality and the level of detail of the habitat information on which the model is based. Developments and sophistication in remote sensing and geographical information systems have resulted in the potential for great increases in both the quality and quantity of habitat-level information that can be obtained and analyzed. These improved techniques also assist the study of forest pathology (Lundquist & Hamelin, 2005).

For developing risk models two issues are needed: a distributional map of the forest species and its health condition, in order to limit the study to the area of interest, and site thematic layers which were considered a priori as relevant for the disease occurrence.

The distribution map of the tree species and health status can be accomplished through techniques of varying complexity and cost. Currently, there are a variety of satellite images of different spatial, spectral and temporal resolution that can be applied to the study of forest ecosystems (Coppin et al., 2004; Iverson et al., 1989). The accuracy of the map will depend on the sensor's ability to discriminate the focal species from others, and its health status, based on measuring changes in electromagnetic energy (Karszenbaum, 1998).

Some of the sensors used for forest studies include Landsat Thematic Mapper (TM), Enhanced Thematic Mapper (ETM) and Multispectral Scanner (MSS), SPOT HRV, the Advanced Very High Resolution Radiometer (AVHRR), Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER)**,** QuickBird and Ikonos, at varying degrees of success **(**Chuvieco & Congalton, 1989**;** Franklin, 1994; Hyyppä et al., 2000; Lefsky et al., 2001; Martin et al., 1998; Peña & Altmann, 2009; Zhu & Evans, 1994). Sometimes, an intensive checking and corrections on the basis of field information are needed, and an iterative approach between image processing and field check must be applied (La Manna et al., 2008a). Aerial photographs are greatly useful for mapping and monitoring forests (Hennon et al., 1990; Holmström et al., 2001; Tuominen & Pekkarinen, 2005), however in many countries they are too expensive to acquire. It is also important to define if visual damage is enough for diagnosing the disease.

The site variables that should be included a priori in a risk model depend on the forest disease. Variables should be pre-selected based on current knowledge of the disease. For example, for mapping the risk of sudden oak death caused by *Phytophthora ramorum*, temperature and moisture variables were considered taking in account the pathogen persistence (Meentemeyer et al., 2004). For some species and areas of study, the wind was a relevant factor (Gardiner & Quine, 2000). For other forest diseases, the nutrition and soil characteristics were determinant factors (Bernier & Lewis, 1999; Demchik & Sharpe, 2000; Dezzeo et al., 1997; Horsley et al., 2000; Thomas & Büttner, 1998). *Austrocedrus chilensis* disease was associated with wet soils (La Manna & Rajchenberg, 2004a,b), agreeing with other diseases caused by *Phytophthora* species (Jönsson et al., 2005; Jung and Blaschke, 2004; Jung et al., 2000; Rhoades et al., 2003). Basing on this previous knowledge, climatic, topographic and edaphic thematic layers were considered for building the disease risk model. The environmental variables included in that case were mean annual precipitation, elevation, slope, aspect, distance to streams, and soil pH NaF (as indicator of allophane presence in volcanic soils) (La Manna et al., 2012).

On the other hand, the availability of information is also necessary taken into account. The quality and accuracy of the thematic layers will be the key for developing an useful risk model and for determining its scale. In this sense, there is great disparity in the information available according to the country or the region of study (Matteucci, 2007). However, the access to free information has greatly increased in recent years. For example, Google Earth (www.earth.google.com) may be a good tool for characterizing geomorphologies and drainage systems. Digital elevation models are also freely available. The Global Digital Elevation Model (GDEM) from ASTER has 30m resolution and covers the 99% of the earth surface. The Shuttle Radar Topography Mission (SRTM) obtains altitude data by radar interferometry and covers the 80% of the earth surface. This sensor has 90m resolution, and it also has a 30m resolution band with a lower coverage. Both elevation digital models present advantages and disadvantages (Hayakawa et al., 2008).

Global climate data can be freely obtained from the global grid of precipitation (www.worldclim.org), with 1km spatial resolution (Hijmans et al., 2005). Sometimes, local weather stations or local climate models may provide more useful data since they have a greater level of detail. The greater resolution data may significantly improve risk assessment (Krist et al., 2008).

3. Developing the database

Developing risk models require both, forest health condition and abiotic factors, to be combined in a geographic information system (GIS). In this chapter, tools from Arcview 3.3 and ERDAS software are described, but newer software for editing GIS have similar tools. On the other hand, researchers around the world are developing free GIS, which now or in the future will probably have the same tools.

The database must include information from training sites, i.e. geo-located forests patches whose health condition and abiotic factors are known. Training sites can be selected from field checking (La Manna et al., 2012) or from the map of species distribution and health condition (La Manna et al., 2008b). The patches should have an homogeneous health conditions; and training sites should include diseased and healthy patches or just diseased ones, depending on model requirements. The selection of training sites requires a proper sampling method, covering the range of host and abiotic conditions in order to minimize bias. A stratified-random sampling or a random sampling should be applied, and the extension Table Select deluxe tools v.1.0 of Arc View software can be useful for selection.

The abiotic factors should be mapped in all the study area. Environmental features of training sites are needed to build the database; but the environmental features along all the area of distribution of the forest species are needed to build the risk map. Figure 1 schematizes the process for building database.

Once the environmental layers are complete, the mean values of each site attributes layer can be extracted by the Zonal attributes tool of ERDAS software for each training site. This tool enables to extract the zonal statistics (mean, standard deviation, minimum and maximum) from a vector coverage and save them as polygon attributes.

4. Building risk models

There are different modeling techniques for developing risk model based on abiotic factors, with predictive performance varying according to the focus of the study (Brotons et al., 2004; Manel et al., 2001; Pearson et al., 2006, 2007; Phillips et al. 2006). Data requirements vary between the techniques. While some models require data of presence and absence of the disease (i.e., diseased and healthy training sites), others need only presence data. The former models are appropriate if absence of the disease is due to environmental restrictions, while the latter approach is appropriate when factors other than environmental variables (e.g. history of spread) explain most of the absences.

In some cases, absence data are doubtful; for example for forest diseases that are manifested earlier in the lower stem and latter in the crown, delaying detection by remote sensing. In these cases, health condition of training sites should be obtained from the field (La Manna et al., 2012), since failure to detect absences results in false negatives, which change mathematical functions describing habitats.

Among the available modeling techniques, three are described in this chapter on the basis of their requirements on disease presence or disease presence/absence data: Mahalanobis distance (requires only presence data), Maxent (requires only presence data and generates

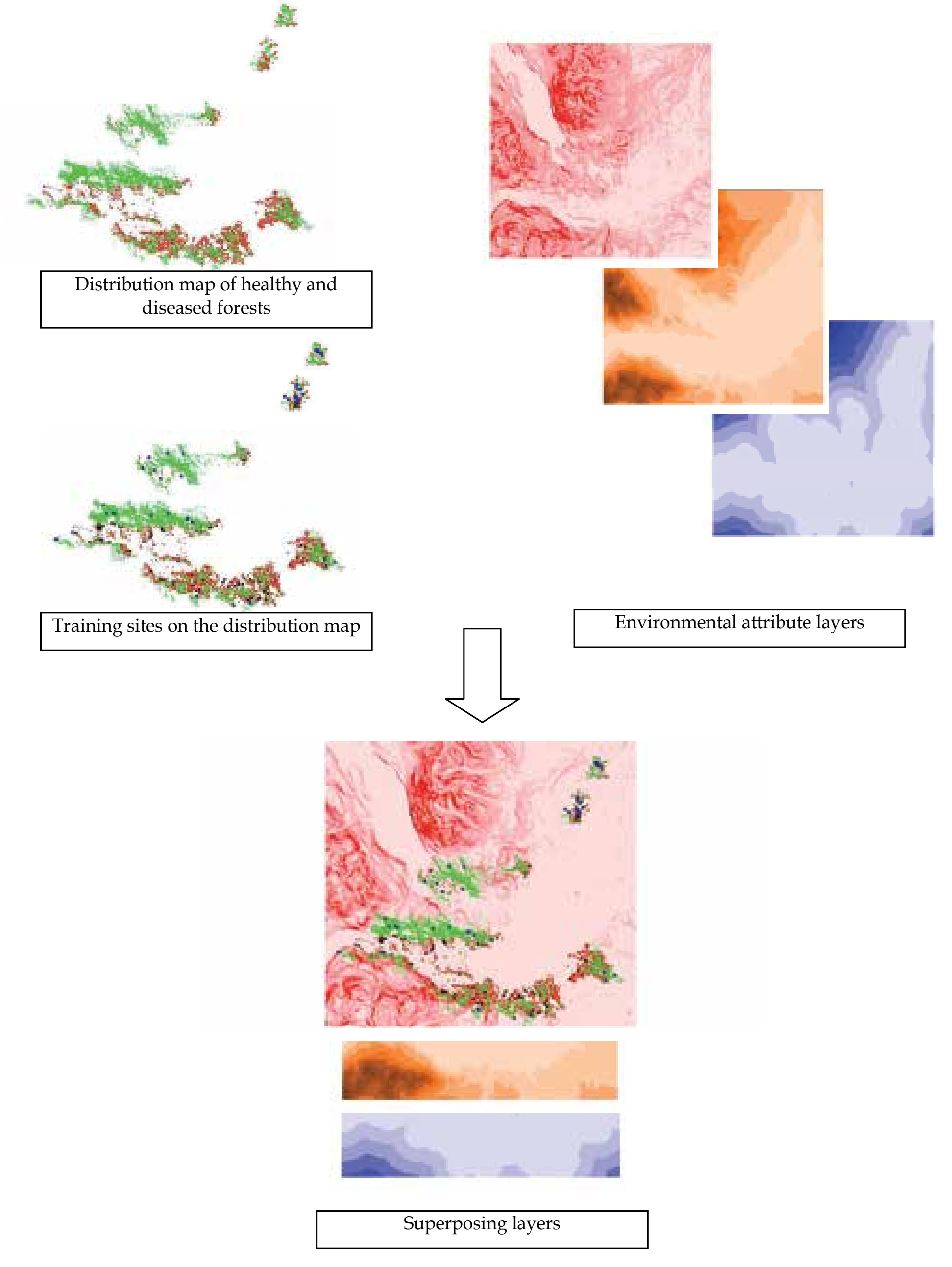


Fig. 1. Scheme for database building

pseudo-absences) and Logistic regression (based on presence/absence data). These methods are inherently flexible, being applicable to a wide range of ecological questions, taxonomic units, and sampling protocols and they produced useful predictions in other studies (DeVries, 2005; Elith et al., 2006; Hellgren et al., 2007; La Manna et al., 2008b, 2012; Marsden & Fielding, 1999; Pearson et al., 2006;Schadt et al., 2002b).

4.1 Mahalanobis distance model

4.1.1 Brief description of the mathematical model

Mahalanobis distance, which requires only presence records, projects the potential distribution of the disease into a geographical space without giving weight to observed absence information (Pearson et al., 2006). Mahalanobis distance was introduced by Mahalanobis (1936) and it is the standardized difference between the values of a set of environmental variables describing a site (rasterized cell or pixel in a GIS) and the mean values for those same variables calculated from points at which the disease was detected (Browning et al., 2005; Rotenberry et al., 2006). Mahalanobis distances are based on both the mean and variance of the predictor variables, plus the covariance matrix of all the variables. Mahalanobis distance is calculated as:

$$D^2=(x-m)^T C^{-1} (x-m) \qquad (1)$$

where:
D^2 =Mahalanobis distance
x=Vector of data
m=Vector of mean values of independent variables
C^{-1}= Inverse Covariance matrix of independent variables
T=Indicates vector should be transposed
The greater the similarity of environment conditions in a point with mean environmental conditions in all training points, the smaller the Mahalanobis distance and the higher the disease risk at that point. Mahalanobis distance has been used in studies employing a GIS to quantify habitat suitability for wildlife and plant species (DeVries, 2005; Johnson & Gillingham, 2005; Hellgren et al., 2007).

4.1.2 Applying Mahalanabis distance in a GIS

Since Mahalanobis distance considers points (and not patches), the polygon layer with the diseased training sites, selected in the field or from the map, must be converted to a point layer. This conversion is done founding the point at the center of each patch, by "Convert shape to centroid" option from Xtool ArcView extension. The vector of mean values for each site variable and the variance/covariance matrix for site variables is generated from this point layer (Figure 2).
The Mahalanobis distance for each cell of the study area is calculated based on this matrix with Mahalanobis distances extension for ArcView (Jenness, 2003). This extension may be freely downloaded from: http://www.jennessent.com/arcview/mahalanobis.htm. For an easier interpretation of results, the Mahalanobis distance statistic can be converted to probability values rescaling to range from 0 to 1 according to χ^2 distribution (Rotenberry et al., 2006).

4.2 Maximum entropy species distribution modelling (Maxent)

4.2.1 Brief description of the mathematical model

Maxent, as Mahalanobis distance, is a model requiring presence data, but it generates "pseudo-absences" using background data as substitute for true absences (Phillips and Dudík, 2008). Thus, Maxent formalizes the principle that the estimated distribution must agree with everything that is known (or inferred from the environmental conditions at the occurrence localities) but should avoid placing any unfounded constraints. The approach is

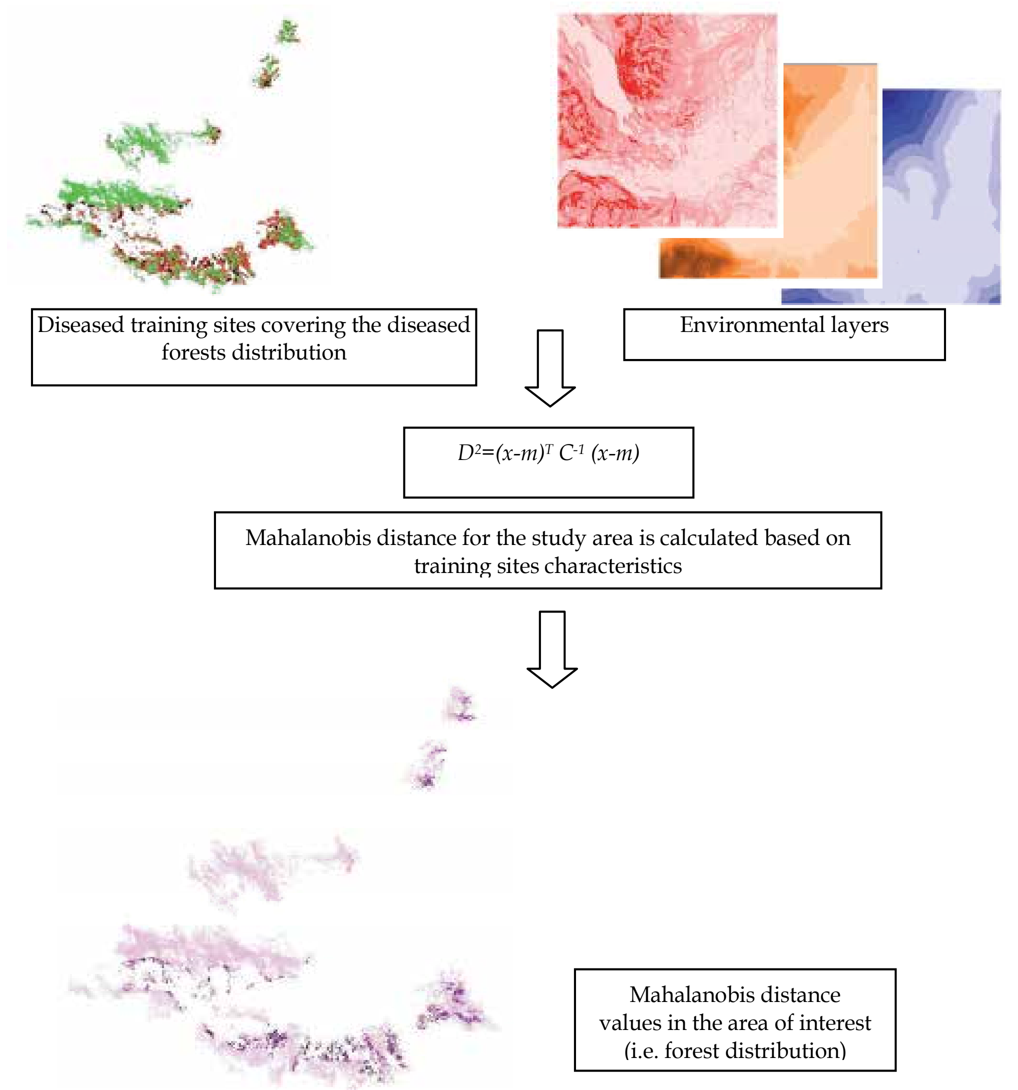


Fig. 2. Schematic representation of Mahalanobis distance procedure

to find the probability distribution of maximum entropy (i.e, closest to uniform, or most spread out), subject to constraints imposed by the information available regarding the observed distribution of the disease and environmental conditions across the study area. The Maxent distribution belongs to the family of Gibb's distributions and maximizes a penalized log likelihood of the presence sites. The mathematical definition of Maxent and the detailed algorithms are described by Phillips et al. (2006), Phillips & Dudík (2008) and Elith et al. (2011).

Maxent has been applied to modeling species distributions and disease risk with good performance (La Manna et al., 2012; Pearson et al., 2007; Phillips & Dudík, 2008; Phillips et al., 2006).

4.2.2 Applying Maxent in a GIS

Maxent can be freely downloaded and used from: http://www.cs.princeton.edu/~schapire/maxent/ and it is regularly updated to include new capabilities. A friendly tutorial explaining how to use this software is provided in the web page, including a Spanish translation.

To perform a run, a file containing presence localities (i.e. diseased training sites), and a directory containing environmental variables need to be supplied. The implementation of Maxent requires the conversion of the files to proper formats. The file with the list of diseased training sites must be in csv format, including their identification name, longitude and latitude. The environmental layers must be saved as ascii raster grids (i.e. .asc format) and the grids must all have the same geographic bounds and cell size. Environmental grids can be saved as ascii file by "Export data source" tool of ArcView. Maxent must be run following the detailed information included in the tutorial (Phillips et al., 2005).

Maxent supports three output formats for model values: the Maxent exponential model itself (raw), cumulative and logistic. The logistic output format, with values between 0 and 1, is easier interpreted and it improves model calibration, so that large differences in output values correspond better to large differences in suitability (Phillips & Dudík, 2008).

4.3 Logistic regression model

4.3.1 Brief description of the mathematical model

The logistic regression is a generalized linear model used for binomial regression, and requires presence/absence data. What distinguishes a logistic regression model from the linear regression model is that the dependent variable is binary or dichotomous (Hosmer & Lemeshow, 1989). The binary dependent variable is disease occurrence (i.e., diseased training site; y=1) and disease absence (i.e., healthy training site; y=0). In contrast to others described models, Logistic regression projects the potential distribution of the disease onto a geographical space whereby information regarding unsuitable conditions resulting from environmental constraints is inherent within the absence data (Pearson et al., 2006).

Logistic regression predicts the probability of occurrence of an event by fitting data to a logistic curve, presenting the following formula:

$$\text{Logit}(P) = \beta 0 + \beta 1 \times V1 + \beta 2 \times V2 + \ldots + \beta n \times Vn \tag{2}$$

where P is the probability of disease occurrence

$\beta 0$ is the Y-intercept

$\beta 1 \ldots \beta n$ are the coefficients assigned to each of the independent variables ($V1 \ldots Vn$)

Probability values are calculated based on the equation below, where e is the natural exponent:

$$P = e^{\text{logit}(P)} / 1 + e^{\text{logit}(P)} \tag{3}$$

A comprehensive description of logistic regression and its applications is presented by Hosmer & Lemeshow (1989). Figure 3 shows a graphical example of a logistic regression model based on presence/absence data of a disease and a soil feature as independent variable.

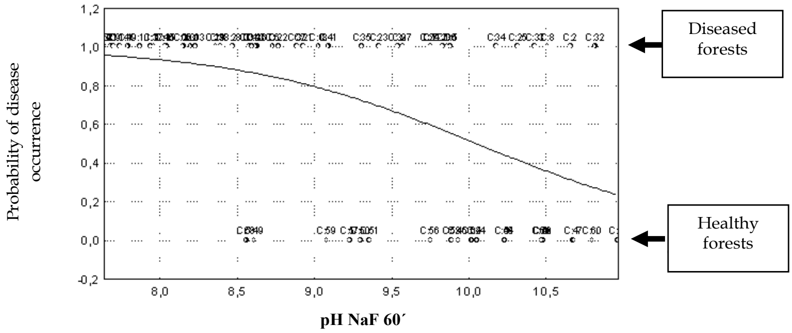


Fig. 3. Observed and estimated (by logistic regression model) probability of *Austrocedrus chilensis* disease according to soil pH NaF values.

4.3.2 Applying logistic regression in a GIS

From the database combining health condition and abiotic factors from training sites, the logistic regression model can be performed using common statistical software, as SPSS, SAS, Infostat, or free software packages. For example, Infostat is a friendly and economic statistical software and it offers a version that can be freely downloaded from: http://www.infostat.com.ar.

The output of logistic regression analysis shows the coefficients assigned to each of the environmental variables (V1… Vn), and the probabilities values for each cell of the study area can be obtained in the GIS. Calculations can be done with "Calculate maps" tool from Grid Analyst extension of ArcView, considering site layers in grid format (Figure 4). Thus, a grid with probabilities of disease occurrence is generated according to the logistic model.

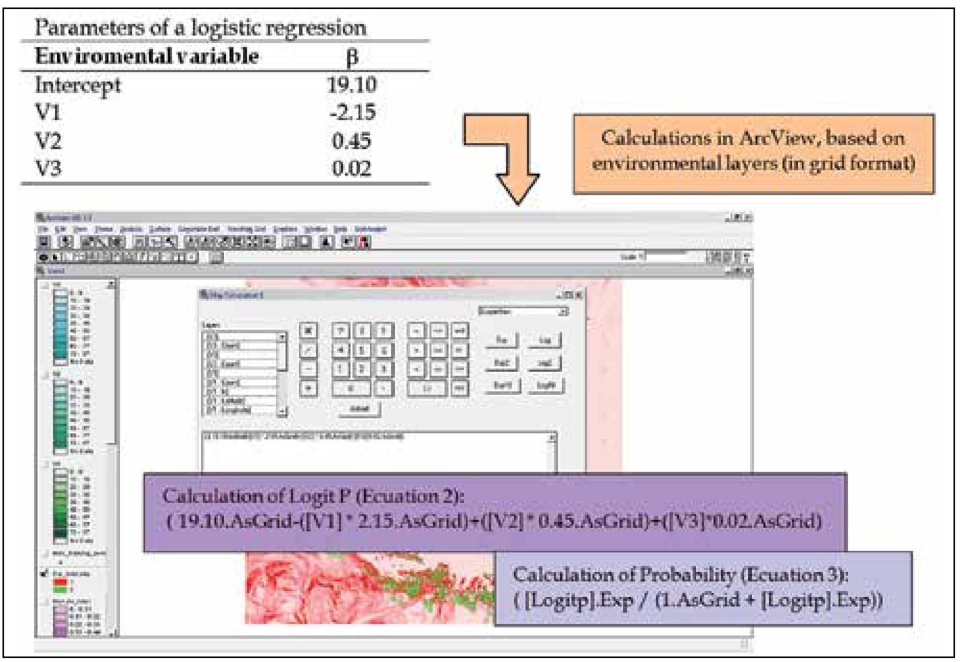


Fig. 4. Example of logistic regression model applied in a GIS.

4.4 Evaluating abiotic factors selecting the most important variables

An advantage of Maxent and the logistic regression models respect to Mahalanobis distance, is that the former allow easily discriminating the abiotic factors most related to the disease and choosing the better combination of variables. As mentioned above, environmental variables included a priori in the models depend on the knowledge about the disease. However, not all the variables considered a priori could be equally important for quantifying the disease risk at the landscape scale.

Maxent allows detecting which variables matter most, calculating the percent contribution to the model for each environmental variable (Phillips et al., 2005). As alternative estimates of variable´s weight, a jackknife test can also be run by Maxent. Figure 5 shows an example of jackknife test, where the environmental variable "agua-move" appears to have the most useful information by itself (blue bar). The environmental variable that decreases the gain the most when it is omitted is also agua_move (light blue bar), which therefore appears to have the most information that is not present in the other variables.

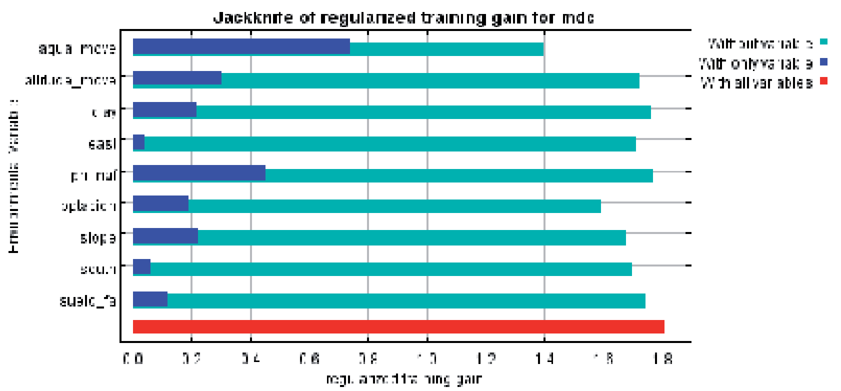


Fig. 5. Example of jackknife test of variable importance according to Maxent software.

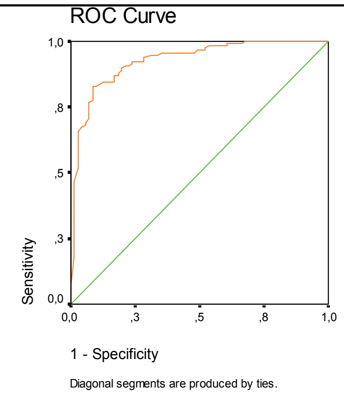
In the case of logistic regression the better combination of variables can be chosen according to the best subsets selection technique (Hosmer and Lemeshow, 1989), the lowest Akaike information criterion (AIC) (Burnham & Anderson, 1998), the greatest sensitivity (i.e., proportion of correctly predicted disease occurrences) or the stepwise method (Steyerberg et al., 1999).

4.5 Assessment of model performance

The predictive performance of modeling algorithms may be very different (Brotons et al., 2004; Manel et al., 2001; Pearson et al., 2006, 2007; Phillips et al., 2006). Differences could be related to the intrinsic properties of mathematical functions inherent to each model and to the various assumptions made by each algorithm when extrapolating environmental variables beyond the range of the data used to define the model (Pearson et al., 2006). Further, the set of data for running the models differs according to consider presence or presence/absence data.

Receiver operating characteristic (ROC) curves and Kappa statistic are index widely used for assessing performance of models. ROC curve procedure is a useful way to evaluate the performance of classification schemes in which there is one variable with two categories by which subjects are classified. The area under the ROC curve (AUC) is the probability of a randomly chosen presence site being ranked above a randomly chosen absence site. This procedure relates relative proportions of correctly and incorrectly classified predictions over a wide and continuous range of threshold levels (Pearce & Ferrier, 2000). The main advantage of this analysis is that AUC provides a single measure of model performance, independent of any particular choice of threshold. AUC can be calculated with common statistical software. ROC plot showed in Figure 6 is obtained by plotting all sensitivity values (true positive fraction) on the y axis against their equivalent (1—specificity) values (false positive fraction) on the x axis. Specificity of a model refers to the proportion of correctly predicted absences.

ROC analysis has been applied to a variety of ecological models (Brotons et al., 2004; Hernández et al., 2006; La Manna et al. 2008b, Pearson et al., 2006; Phillips et al., 2006). Values between 0.7 and 0.9 indicate a reasonable discrimination ability considered potentially useful, and rates higher that 0.9 indicate very good discrimination (Swets, 1988). If absence data are not available, AUC may also be calculated with presence data and pseudo-absences chosen uniformly at random from the study area (Phillips et al., 2006). However, counting with both true absence and presence sites is better for evaluating model performance (Fielding & Bell, 1997).



Test Result Variable(s): REGRE

Area	Std. Error(a)	Asymptotic Sig.(b)	Asymptotic 95% Confidence Interval	
			Lower Bound	Upper Bound
,927	,020	,000	,888	,966

Fig. 6. Example of ROC curve obtained for a regression model by SPSS software.

Kappa statistic is another index widely used (Loiselle et al., 2003; Hérnández et al., 2006; Pearson et al., 2006), that can be calculated with common statistical software. The Cohen's Kappa and Classification Table Metrics 2.1a, an ArcView 3x extension, may also be useful and can be freely downloaded from: http://www.jennessent.com/arcview/kappa_stats.htm. Cohen's kappa is calculated at thresholds increments, e.g. increments of 0.05, from 0 to 1, and the maximum Kappa for each model is considered. Kappa values approaching 0.6 represent a good model (Fielding & Bell, 1997).

The models should be run on the full set of training data, to provide best estimates of the disease's potential distribution (Philips et al., 2006). However, in order to assess and to compare the model performance, models should be run with just a portion of the training sites and the rest of data should be used for the assessment. For each model, some (e.g. ten) random partitions of data are done maintaining the remaining 25% of training sites for performance assessment. Then, AUC and Kappa values are calculated for each random set

of assessment data and for each model, and they are compared between models by non-parametric analysis (Philips et al., 2006).
The performance of the three models described in this chapter (i.e. Mahalanobis distance, Maxent and Logistic Regression) was compared for modeling a forest disease in Patagonia (La Manna et al., 2012). Results showed that all the models were consistent in their prediction; however, Maxent and Logistic regression presented a better performance, with greater values of AUC and Kappa statistics; and logistic regression allowed the best discrimination of high risk sites. Studies that compared presence-absence versus presence-only modeling methods, suggest that if absence data are available, methods using this information should be preferably used in most situations (Brotons et al., 2004). However, Maxent is considered as one of the best performing models (Elith et al. 2006; Hernández et al., 2006; Pearson et al., 2006; Phillips et al., 2006), and Mahalanobis distance also provided good results in conservation studies (DeVries, 2005; Johnson & Gillingham, 2005; Hellgren et al., 2007).
The performance of the risk models may greatly vary in each case and forest disease. Building and comparing models based on different algorithms allow finding the best.

4.6 Mapping the risk. Selecting thresholds

The three risk models presented in this chapter have as result grids with probabilities values of disease occurrence, varying between 0 and 1. However, for proposing management criteria is important to define what probability represents a high risk of disease. 0.4?, 0.5?, 0.7?... In order to convert quantitative measures of disease risk (i.e., probability) to qualitative values (i.e., low, moderate or high risk) threshold values must be selected.
A possible criterion is to define thresholds by maximizing agreement between observed and modeled distributions for the sampled dataset. Sensitivity (the proportion of true positive predictions vs. the number of actual positive sites) and specificity (the proportion of true negative predictions vs. the number of actual negative sites) are calculated at different thresholds according to AUC coordinates. The threshold at which these two values are closest can be adopted. This approach balances the cost arising from an incorrect prediction against the benefit gained from a correct prediction (Manel et al., 2001), and is one of the recommended criteria for selecting thresholds (Liu et al., 2005).
The lowest predicted value associated with any one of the observed presence records can also considered as a threshold (i.e, lowest presence threshold) (Pearson et al., 2007). This approach can be interpreted ecologically as identifying pixels predicted as being at least as suitable as those where the disease presence has been recorded. The threshold identifies the maximum predicted area possible whilst maintaining zero omission error in the training data set.
Using the two thresholds, three risk categories can be defined: low (with p values lower than the lowest presence threshold); moderate (p values between the lowest and the sensitivity-specificity approach thresholds); and high risk (p values greater than the sensitivity-specificity approach threshold) (La Manna et al. 2012).
Risk maps of disease occurrence can be generated for each model by reclassifying the model outputs, using Grid analyst extension of ArcView software.

5. Conclusions

Forest diseases are key determinants of forest health, and information about disease presence and potential distribution are important to any management decision. Risk maps are more likely to be used if they addresses the same scale at which management decisions are made. Stand scale management is increasingly being supplemented or replaced by landscape-scale management (Lundquist, 2005). Forest diseases risk assessment provides important information to the forest services that makes critical decisions on the best allocation of often-scarce resources.

Risk models for pine wilt disease (*Bursaphelenchus xylophilus*) in Spain allowed planning control actions and preventing to plant susceptible species in the high risk areas (Fernández & Solla, 2006). Risk models for sudden oak death in California provide an effective management tool for identifying emergent infections before they become established (Meentemeyer et al., 2004). Risk models for economically important South African plantation pathogens allowed to asses the impact of climate change on the local forestry industry (Van Staden et al., 2004). Risk maps for *A. chilensis* disease in a valley of Patagonia allowed to detect healthy forests at risk only inside protected areas. These results allowed to suggest management actions for cattle and logging in disease-prone sites. This risk map also provided useful information for preventing restock in areas where the risk is greatest (La Manna et al., 2012).

Risk models discussed in this chapter allowed the evaluation of abiotic factors related to the disease. This kind of models provides important information, which can be improved if knowledge about the biology and spreading of a causal biotic agent is available. It is important to know whether the forest pathogen under study is endemic or exotic. If it is exotic, the susceptibility must be assessed, based on the biological availability of a host and the potential for introduction and establishment of the disease within a predefined time frame. For this evaluation, the connectivity between patches may be key (Ellis et al., 2010). On the other hand, if it is endemic, the disease is already established throughout a region, and then a susceptibility assessment is not required because the potential or source for actualized harm is assumed to be equal everywhere (Krist et al., 2006).

For both endemic and exotic diseases, mortality occurrence may vary greatly depending on site and stand conditions, and models like those shown in this chapter are a good tool for assessing risk. Variables included in the models should be carefully pre-selected according to the previous knowledge about the disease. These models (i.e., Mahalanobis distance, Maxent and Logistic Regression) also admit variables like distance to roads, or distance to foci of infection, that could be important for spreading of infectious diseases.

6. Acknowledgment

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8

A Common-Pool Resource Approach to Forest Health: The Case of the Southern Pine Beetle*

John Schelhas[1] and Joseph Molnar[2]
[1]Southern Research Station, USDA Forest Service
G.W. Carver Agricultural Experiment Station, Tuskegee University, Tuskegee
[2]Department of Agricultural Economics and Rural Sociology
Alabama Agricultural Experiment Station, Auburn University
USA

1. Introduction

The southern pine beetle, *Dendroctonus frontalis*, is a major threat to pine forest health in the South, and is expected to play an increasingly important role in the future of the South's pine forests (Ward and Mistretta 2002). Once a forest stand is infected with southern pine beetle (SPB), elimination and isolation of the infested and immediately surrounding trees is required to control the outbreak. If insect-infested trees are not swiftly removed, infestations can spread to healthy forests. The most effective approach to managing SPB is through preventive measures that maintain forests in vigorous, healthy conditions, including thinning and prescribed burning. At a landscape level, preventive measures reduce the overall incidence of SPB and thereby the spillover of SPB to adjacent landholdings. Yet many forest landowners do not undertake the management actions that can limit SPB outbreaks. The tragedy of the commons in forest health takes place when individual private owners do not acknowledge their communal responsibilities thus risking catastrophic losses due to poor management and/or absentee tenure.

The South's forests are largely in private ownership (89% of the South's timberland, with nonindustrial private forest (NIPF) land ownerships representing about 95% of the private forest landowners and 63% of the private forest land region (Birch 1996, Wicker 2002). Population growth and suburban and exurban expansion in the South have divided many forest landholdings into increasingly smaller-sized parcels. Surveys of forest landowners in the South find that 90% of the NIPF owners hold less than 100 acres, and that owners are diverse in occupation, income, residence, forest land ownership objectives, use of professional forest management assistance, and forest management strategies (Birch 1996, 1997; Bliss and Martin 1989).

The diversity of ownership objectives and management styles on NIPF lands results in widely different awareness and responses to forest pest problems (Ward and Mistretta 2002). Pine beetle outbreaks are cyclic, sporadic, and potentially highly devastating (Meeker

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et al. 1995). Extensive outbreaks not only inflict setbacks on individual owners who suffer losses from forced sale of high-value saw timber for low-value pulp, but also collective damages on all forest owners.

The maintenance of healthy pine forests and the various benefits associated with them in the South depends on effective management and control of the Southern Pine Beetle. To a significant extent, SPB management is a social problem because the most practical way to control SPB requires collective action by individual landowners across the pine forest landscapes in the South. Most social research on programs for forest landowners in the U.S. has tended to view them as individuals, and be oriented toward transferring new knowledge, technical assistance, financial assistance and even cultural content to autonomous forest landowners (Best and Wayburn 2001; Schelhas et al. 2004). Accordingly, we have oriented much of our analysis on forest landowners and SPB to understanding why individual landowners do or do not engage in practices known to be effective in the prevention of SPB (Molnar et al. 2003).

However, we also recognize that, from a social science viewpoint, the characteristics of the SPB issue–the need for action at the landscape level, when landscapes are in multiple ownerships--is a problem of the commons (Ostrom 1990). Natural resource management in the commons has been subject to a great deal of study over the past few decades, although little or none of this research has addressed questions of forest health. However we believe that the general principles of the management of common-pool resources can provide some important insights for SPB management. In this paper we explore the usefulness of examining the management of SPB from the perspective of common-pool resource management. As Hardin (1968) notes, an implicit and almost universal assumption of discussions of resource management problems is that a technical solution must exist and the task is to find it. A technical solution may be defined as one that requires a change only in the techniques of the material sciences, demanding little or nothing in the way of change in human values or ideas of morality.

2. A brief review of theory of common-pool resources and forests

Three types of resources can be identified based on different combinations of two characteristics: (1) *subtractability or rivalness*, or the degree to which use by one person diminishes the potential for use by another, and (2) *excludabilty*, the cost of excluding potential beneficiaries from the resource (McKean 2003). **Private** resources are subtractable in consumption and others can be excluded relatively easily. **Public** resources are available to all (exclusion is not possible or is extremely costly) but not subtractable. Examples include public radio stations, scientific knowledge, and world peace. Individuals may enjoy the benefits of these without contributing to their production (free ride), but if everyone does this a less than ideal amount of the good will be provided (Dietz 2001, Ostrom and Walker 1997). **Common-pool** resources are subtractable but exclusion is difficult (Dolsak and Ostrom 2003).

Although it has been common in the past to discuss common property resources, recent work has emphasized the importance of distinguishing types of resources (based on their inherent attributes, from types of ownership (Dietz et al. 2001). Property may be held in four ways: (1) *private*, in which individuals or corporations have the rights to exclude others from using a resource and to regulate a resource; (2) *public or state*, in which the government has rights to a resource, and makes decisions about access as well as the nature and level of exploitation; (3) *common property*, in which the resource is held by an identifiable group of interdependent users with the rights to exclude others, and (4) *open access*, in which there are no well-defined property rights, the resource is unregulated, and it is free and open to everyone (Feeny et al. 1990). Research on the commons suggests that the fit between property type and resource type has an important bearing on effective resource management (Dietz et al. 2001, Stern et al. 2002).

Geores (2003) points out that forests are complex, large scale resources that can be defined and assigned property rights in various ways: (1) Forest are appreciated as renewable natural resources, valued for the use of their products and for their roles in maintaining watersheds, soil fertility, and air quality, as well as for their importance as cultural resources, both religious and aesthetic. (2) Forests are resources that contain resources, being made up of biosystems of varying complexity and used for many different social and economic functions as a part of complex social systems. (3) Forests resources are dynamic and defined on multiple scales. Forest and forest resource definitions differ in scale, but are not necessarily mutually exclusive.

Southern forests illustrate this in the way that the wider public values them for wildlife, watershed, biodiversity, and climatic benefits (each requiring management at different scales). In contrast, trees and forests are used and valued by individual landowners for timber. Even when considering only a single resource, such as timber production or wildlife by individual owners, owners of individual parcels may want to encourage or guarantee that owners of adjacent parcels have compatible and complementary interests in their parcels. Neighbors want their neighbors to maintain wildlife habitat and keep vegetative cover intact. They also want adjacent land owners to allow wildlife transit and to refrain from introducing or encouraging certain problem species (McKean 2000).

Gibson and Becker (2000), recognizing that forests generally constitute multiple resources, note that strong individual property rights alone do no guarantee a forest's health since individuals can have short term incentives to convert or degrade forests that conflict with long term forest sustainability. Because they are common-pool and public resources, many forest resources cannot be effectively managed on the scale at which they are owned or in the decision-making time frames of some private owners. As a result, individual forest owners have an interest in what happens on lands adjacent to theirs. Southern pine beetle is a classic example of the stake neighbors have in the way their neighbors attend to forest health.

One of the problems facing common-pool resources is the appropriation problem. If resource units have high value and institutional constraints do not restrict use, individuals face a strong temptation to overexploit and thereby degrade the resource. For example if a forest is open to access by all with no social institutions to limit use, is it likely that timber would be removed at such a rate that the forest would degrade and future timber harvests would be reduced (Hardin 1968). Extensive study of the appropriation problem by social scientists has found that the tragedy of the commons is not inevitable; resource users can organize to implement social mechanisms to restrict use to sustainable levels (Richard and Stein 2003). Other problems of common-pool resources, such as provision and maintenance problems, have received less study but are still important (Ostrom 1999).

Forest health is essentially a provision and maintenance problem. In many ways, it is a public good, in that people can free ride on other people's efforts to enhance forest health at a landscape or regional level. But McKean (2003) notes that public goods that are subject to crowding, wear, and depletion are not pure public goods, and have many characteristics of

common-pool resources. Furthermore, Ostrom (1999) notes that in the case of negative public goods (e.g. forest pests), individual owners or appropriators tend not to be motivated to pay for or take the collective actions that are required to reduce the negative public good, resulting in a negative provision of that good (e.g. poor forest health). Provision problems in common-pool resources are very similar to pure public good problems (Ostrom et al. 1994).
Having shown that forests and forest health have important attributes of common-pool resources, the next question is what common-pool resource theory and scholarship can contribute to the health and management of Southern pine forests. Ostrom and Walker (1997) examined many cases of successful common-pool resource management. They identified design principles for development of institutions that increase the efficiency of management of common-pool resources, institutions that are often developed in combination by the resource users and the state.

3. Key understandings from research on individual NIPF owners

A legacy of medieval times, Carlsson (1996) explains why Swedish common forests have survived as vital and competitive actors in the timber market. These lands are held in common under shareholder arrangements managed by the government. He offers three main explanations: the commoners' conscious attempts to reduce transaction costs, their general inventiveness in adjusting to changed circumstances, and their acclimatization to present economic conditions. Although he does not specifically address forest health issues, the notion that a commons institutions offers multiple advantages to a dispersed, nonresidential, and nontechnical population of forest owners suggests a need for new institutions and mechanisms to bind and benefit nonindustrial private forest land owners (NIPF).
Most NIPF landowners are aware of SPB, many are interested in preventing the pest, and some express a desire to accomplish control measures (Molnar et al. 2003). Those actually taking action to prevent and manage infestations are few, however.
Molnar et al. (2003) found important differences by size of forest landholding. Larger landholders are more likely to have taken steps to control infestations, but there were markedly lower levels of awareness, surveillance, and prevention activities among small holders. Larger landowners had high surveillance efforts and took more action to respond to SPB damage when it happened on their land. Larger landowners were also strongly influenced by timber prices in their efforts to control SPB.
Smallholders lacked knowledge about what to do about SPB, lacking familiarity with public agency programs and utilization of financial assistance. They used fewer information sources, and expressed less desire for information about forest management (Molnar et al. 2003).
Some values that landowners–large and small–have for their forest land may provide less than compelling motivations for SPB management. Those interested in recreation and outdoor enjoyment and indicating preservation as a primary reason for forest ownership were less aware and interested in SPB management (Molnar et al. 2003). The control of SPB and the protection of forest health, involves more than the vigilance of the individual forest owner, however.
Carlsson (1996: 12) concludes that the Swedish forest commons have survived as prosperous timber producers and providers of public goods, not only because of their conscious reduction of transaction costs but also because this reduction has been made possible by a

general fragmentation of the centralized State, playing its multiple roles. This fragmentation has provided a local 'opportunity structure' that the commons have utilized. This has been possible because the commons, their forest managers, boards and assemblies of shareholders still possess sufficient local, current knowledge to be able to adjust the commons to industrialized society. The main lesson to be learned from the Swedish common forests might be their successful integration, rather than their separation, from the logic of the negotiated economy and industrialized society. Designers of institutional mechanisms to articulate and organize the collective aspects of forest health might learn much from the Swedish experience.

4. Calculating the benefit from change in rules of forest management

Ostrom (1999:4) emphasizes that the "social behavior of adopting new practices in natural resources management as a rational decision process. Each user has to compare the net benefits continuing to use the old rules of harvesting from a resource to the benefits he or she expects to achieve with a new set of rules. Each user must ask whether his or her incentive to change is positive or negative.

If the incentive to change is positive for some users, they then need to estimate three types of costs: the up-front cost of time and effort devising and agreeing upon new rules; the short-term costs of adopting new strategies, and the long-term cost of monitoring and maintaining a self governed system over time (given the norms of community where they live). If the sum of these expected costs for each user exceeds the incentive to change, no user will invest the time and resources needed to create new institutions. And if this applies to all the users, no change will occur (Ostrom 1999:4).

In field settings, not everyone expects the same cost and benefits from a proposed change. Consequently, the collective choice rules used to change the day-to-day operational rules related to management activities affect whether an institutional change favored by some and opposed by others will occur (Ostrom 1999:4).

These comparisons can be difficult to make in practice since considerable uncertainty always exists concerning the strategies that participants will follow once rules are changed (Ostrom 1999:4). But even though this is a difficult task, it is one undertaken frequently by users after discussing the effects of a change in rules. Rules about monitoring forest lands for SPB infestation may be one example of an institutional change.

Prevention efforts require vigilant surveillance for infestations and adherence to planting and management recommendations that discourage SPB outbreaks. Once outbreaks occur, control requires prompt treatment, and a comprehensive response by all forest owners to stop the spread of SPB to neighboring lands (Egan and Jones 1993, Ervin et al. 2001). Yet many NIPF owners have weak and uneven ties to their properties, and many do not share the sense of urgency that professional foresters often have about SPB prevention and control (Williston et al. 1998).

5. Forest health as a common property resource

Land (and forest) tenure is now widely understood as bundle of rights, all or some of which may be privately owned. Under communal systems, no individual resource rights are privately owned. Under private property systems, the deed holder seemingly owns all rights.

It is increasingly clear that some rights in the bundle can never be exclusively held by individuals, and are in fact dependent on communal cooperation and respect. Forest health may be one such communally owned and managed resource that is held by all forest owners but no one singly. This common pool, open access resource, abused by one, can cause all to suffer. An ephemeral and situational commodity, forest health is often taken for granted when insects, fire, or other threats are not imminent.

The owners of the forest health right or resource are connected in concentric levels of proximity. That is, near neighbors are more frequently and intensively affected by mutual actions and responsibilities. Distant parties are less frequently benefited or harmed by an individual landowner's vigilance and response to forest health problems. Institutions such as forest fire districts sometimes connect land owners in defense of fire threats, but fire threats are not commonly limited to pest prevention.

These indirect and fleeting communal connections among NIPF owners are at the core of the problems facing public agencies charged with promoting forest health. For the most part, locally resident forest land owners often have little basis for interpersonal association. Even among landowners who reside in the same county as their forest land, the increasing separation of residence from ownership diminishes the prospect for face-to-face interaction with neighboring forest land owners.

McKean and Ostrom (1995) find it noteworthy that the definition of private property rights has to do with the rights, not the nature of the entity that holds them. The privateness of private property rights does not require that individual persons hold them; they may also be vested in groups of individuals. Unfortunately, the rights to forest health are not alienable or separable; such rights are evanescent or intangible. Yet when unevenly exercised, forest fires or large-scale timber losses from insect damage are the result.

Scholars who have designed taxonomies to point out the difference between open access arrangements and common property have sometimes distinguished four very general "types" of property: public, private, common, and open access. McKean and Ostrom (1995) object to this classification because it creates the erroneous impression that common property is not private property and thus does not share in the desirable attributes of private property, although forest health property rights are indeed commonly held. They feel that common property is in fact shared private property and should be considered alongside business partnerships, joint-stock corporations and cooperatives. Yet, the shared resource of forest health is often not widely recognized as a common property resource

Oakerson (1986) has suggested a model to analyze and explain the main factors involved in the management of common property resources. In its simplest form, the Oakerson model is based on understanding the relationships between the physical characteristics of the resource, the decision making rules of the group or users involved, the patterns of interactions resulting from the appropriation and use of the resource, and the outcomes of this process. Blaikie and Brookfield (1987) have modified the Oakerson model to explain the dynamic interactions and adaptive changes when a resource is managed under a communal (or collective) regime.

Mutual regulation through the institutional equivalent of a common property regime is more desirable as resource use intensifies and approaches the productive limits of a resource system (McKean and Ostrom 1995). Further, since it is people who use resources, forest health common property becomes more desirable - not necessarily more workable but more valuable and thus more worth trying - as population density increases on a given resource base. Thus the challenge to resource agencies endeavoring to create a common property resource in forest health must find a way to communicate with NIPF owners in such a way so they become aware of the common property resource they share and have a sense of ownership in the commons.

Natural resources stakeholders have different interests, and investigation of these through discussion can help to identify how people view their current and potential roles in forest management (Higman et al.1999: 170). The challenge to resource managers is to communicate the common property resource aspects of forest health. Higman et al. (1999:170) claim that finding out how people see their own roles in forest management is an essential step toward agreeing about the objectives of forest management. One way of doing this is to focus discussion on stakeholders' rights, responsibilities and results with respect to surveillance and timely response to SPB outbreaks.

As a result from their different rights, responsibilities and returns, stakeholders also have different sorts of relationships with each other. Some may not be aware of each other, or may ignore each other; others may be in varying states of disagreement or cooperation in different issues related to forest management. Yet all share some level of common interest in forest health.

6. Characterizing a robust common property system for forest health

A robust system of social organization for NIPF owners that would promote and protect the common property aspects of forest health has yet to be devised. McKean (1992, 1996, 2000) has written on the nature of common property systems that would lead to ecological benefits for the natural world. She identifies a number of design criteria that may make common property systems robust (McKean 2000a), focusing on internal and external features of the resource management system.

Internal Features pertain to relationships among co-owners, that is, among NIPF owners. Each of McKean's design features is discussed in terms of a common property management system for forest health.

1. Co-owners of resource rights must be a self-conscious and self-governing group.

This feature is hard to envision occurring beyond a watershed or county scale. As previously discussed, nonresident, nontechnical, and dispersed landowners have no mechanism for communication or collaboration. Thus efforts to promote the common pool resource aspects of forest health must develop new mechanisms for linking heretofore-unconnected NIPF owners.

2. The group needs a mechanism for resolving internal conflict.

Current mechanisms generate little direct conflict because NIPF owners have little occasion to interact with one another. Animosity toward noncompliant landowners may be manifested under specific circumstances, but the forest health consequences of NIPF owner indifference or neglect are typically absorbed or ignored by neighboring landowners.

3. The rules need to provide for monitoring of behavior and enforcement of sanctions.

Some states have laws and regulations that sanction noncompliant NIPF for neglecting SPB infestations, yet it is not clear how often these measures are put into play nor how effective they are in influencing behavior.

4. The rules need to include arrangements to prevent abuse by guards.

It is not clear who the "guards" might be for forest health. At present, public forest managers monitor aerial photos and accumulate reports of infestations to provide assessments of SPB problems. Under a common property regime, NIPF owners themselves might play a greater role in surveillance, requiring access to private lands and other measures that might otherwise compromise individual property rights. If such access were used for private gain – e.g., off-roading, hunting, fishing, or trapping -- cooperation and the common property institution would be undermined.

5. **The rules need to be easily enforceable and ecologically conservative.**

Rules for managing forest health as common property would require a great deal of public education and would have to be nested in the current web of property law and public agency regulation. Monitoring and infestation response requirements would have to achieve a level of technical and sociopolitical consensus about the techniques of SPB control. Motivating NIPF owners to participate in such discussions would a challenge to resource management agencies not only in terms of the sheer number of actors that would have to be contacted, but also in terms of the communication and participation efforts that would be needed to enlist and sustain NIPF owner involvement and commitment.

6. **The allocation of benefits from the commons needs to be roughly proportional to the effort (time, money) invested in the commons.**

Under the Swedish system discussed earlier, common members are shareholders in corporate institutions that protect and manage production from forest lands (Carlsson 1996). A U.S. system that endeavored to enlist NIPF in monitoring and managing forest health on a per acre basis might not produce sufficient incentives for small holders. Devising institutional incentives that motivate participation and commitment from large and small holders would have to balance the costs of participation with the infrequently tangible, usually delayed, and often diffuse benefits of forest health.

External Features encompass relationships between the body of co-owners and the outside world. Four considerations relate to the issue of forest health.

7. **The co-owning community of resource users is much better off if it has independent jurisdiction or autonomy**

Soil and water conservation districts are examples of communities of resource users that have some independent jurisdiction. Such entities are, however, creatures of state and federal laws that enable them. It is clear that not all landowners participate, nor do all that participate benefit equally from these programs – particularly in terms of size of holding and ethnicity of the land owner (Schelhas 2003).

8. **The boundaries of common property regimes need to be set at an appropriate ecological** scale and need to match ecosystem boundaries.

It is not clear what the appropriate ecological scale is for forest health. Other efforts are underway to organize land owners on the scale of the watershed, thus is seem prudent to seek coincident boundaries between soil, water, and forest resource units of social organization. McKean (2000:10) points out that it is silly to introduce common property institutions where parceled individual property would make more sense, and it is vital to use common property where parcelization to individuals is not a good idea. Forest health is not a resource that is easily parcellized.

9. **It is important to select the right group to vest common property rights in order to get capacity to affect the problem.**

The unit of organization must be close enough to the problem to aggregate individual decisions and realize consequences for the resource to be managed. A common property institution should combine NIPF forest landowners in a way that connects their efforts to the cause of forest health and achieves demonstrable consequences for the resource as well as the NIPF owners.

10. On large resource systems, it is important to nest new layers of governance (federalism)

Social organization designed to coalesce NIPF owners to achieve forest health must be aligned with the other emerging forms of association that endeavor to promote and protect resources. Forest resource management must complement water and soil management efforts; there must be some level of mutual reinforcement and synergy to achieve effective environmental management. The environment is interconnected; so must the efforts to make it sustainable.

7. Social capital, social organization, and common property

Each U.S. county has some level of social capital – fire districts, irrigation districts, soil conservation districts, forest associations, extension councils, etc.--that can be drawn on to construct the common property institution in forest health. Institutional changes that expand fire protection vigilance to forest health surveillance including SPB monitoring can build on existing social arrangements to protect forest health. Flora (2000: 87) notes the importance of building human and social capital for communities that are engage in natural resources management. Social capital involves mutual trust where people know they can count on someone, which fosters reciprocity. Mutual trust is established when different institutions and individuals can both give and receive. Mutual trust and reciprocity tend to occur when people work together.

Flora (2000: 87) mentions that one way of building trust is to start with small projects that have immediate visible results that everyone can measure and contribute to. Face-to-face groups are the building blocks of social capital. The measurement of increased social capital is done by looking at the strengthened relationships and communication among unlikely segments within or outside the community and the increased availability of information and knowledge. McDonald and McLain (2003) describe the successful integration of community well-being and forest health in the Pacific Northwest. They found that a central vehicle for change was the creation of a quasi-public organization (Conservation and Development Council) that had as its first objective to improve economic and social well-being. Specifically, the Council promoted forest health and community well-being through habitat restoration programs that employed people in the area. The Council used special forest products programs to encourage businesses to pool resources for equipment and marketing, and give employees training in forest products harvest and marketing. The Council also sponsored a wood products production and marketing activities programs to help public and private owners produce and market wood products.

Council activities also played an important role in creating new alliances and changing relationships among local and non-local organizations. It increased the capacity of local groups to obtain funds and gain access to technical expertise from outside organizations. In short, it provided an institutional substrate for managing the forest health commons.

Ostrom (1999:2) defined a self-governed forest resource as one where actors, who are major users of the forest, are involved over time in making and adapting rules within collective-choice arenas regarding the inclusion or exclusion of participants, appropriation strategies, obligation of participants, monitoring or sanctioning, and conflict resolution. In most modern political economies it is rare to find any resource system that are governed entirely by participants without rules made by local, regional and national authorities also affecting key decisions. Thus in a self-governed system, participants make many, but not all, rules that affect the sustainability of the resource system and its use.

Both the natural physical boundaries of a forest as well as the legal boundaries for a particular community's forest must be clearly identified and defined (McKean and Ostrom 1995). The lack of definition and assignment of forest health property rights quite clearly represents a barrier to forestry management, on the one hand limiting the realization of prevention and control benefits and, on the other, encouraging "free rider" behavior and giving rise to the so-called tragedy of the commons—outbreaks that spread to neighboring properties and create otherwise avoidable catastrophic timber losses.

Inflexible rules are brittle, and thus fragile, and can jeopardize an otherwise well-organized common property regime (McKean and Ostrom 1995). In particular, the science behind SPB had not fully defined the rise and fall of SPB populations. Consequently, in some years natural forces driving surges in SPB infestation may overcome high levels of surveillance and response to outbreaks. The setbacks and frustrations occurring to NIPF owners stress the institutions that normally prevent and control SPB outbreaks.

Institutions for managing very large systems need to be layered, with considerable authority devolved to small components. Many different communities, some of which are in frequent contact with each other and some of which are not, may use a large forest. The need to manage a large forest as a unit would seem to contradict the need to give each of that forest's user communities some degree of independence. Nesting different user groups in a pyramidal organization appears to be one way to resolve this contradiction, allowing simultaneously for independence and coordination (Cernea 1985). The most successful models of nesting come from irrigation systems serving thousands of people at a time (McKean and Ostrom 1995). It is not clear whether such high levels of social organization are necessary or feasible to achieve forest health.

8. Conclusions

If forest health is an emerging commons, every new enclosure of the commons involves the infringement of somebody's personal liberty (Hardin 1968). Infringements made in the distant past are accepted because no contemporary complains of a loss. Newly proposed infringements to articulate monitoring and management responsibilities may be vigorously opposed by NIPF owners as violating property rights. But what do property rights mean? When landowners mutually agree to prevent and limit losses from natural threats, all forest owners become more free and perhaps more wealthy. As Hardin (1968) concludes by citing Hegel, "Freedom is the recognition of necessity"; individuals locked into the logic of the commons are free only to bring on universal ruin. Once they see the necessity of mutual coercion, they become free to pursue other goals.

Like individual parcellation, the recognition of common property gives resource owners the incentive to prevent and control insect damage, to make investments in forest health and to manage them sustainably and thus efficiently over the long term (McKean and Ostrom 1995). Forest health cannot be privately owned; it is an open-access resource. However, unlike individual parcellation, common property offers a way to continue productive use of the private aspects of a resource system while solving the monitoring and enforcement problems posed by the need to survey forest lands for insect problems.

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Section 4

Protective and Productive Functions

9

Ecological Consequences of Increased Biomass Removal for Bioenergy from Boreal Forests

Nicholas Clarke
Norwegian Forest and Landscape Institute
Norway

1. Introduction

The increased use of renewable energy sources, including forest biomass, in energy consumption is a marked characteristic in many countries' current energy policies. Use of forest biomass for energy is supported as a sustainable form of energy that contributes to social welfare, local development and forest economy. Thus, in Europe there is a sharp increase in demand for wood as a source of renewable energy as well as for production of wood products. Forest inventories show that standing stock as well as annual growth would allow an increased use of the existing forest resource.

In conventional stem-only timber harvesting (SOH), where branches and tops are left in the forests, the organic material will decay on the site and nutrients are thus returned to the biogeochemical cycle. In whole-tree harvesting (WTH), branches and tops are removed, although in practice the amount removed is about 60-80% (Helmisaari et al., 2011). As a large part of the nutrients in trees are located in the foliage and branches, removing these will reduce the supply of nutrients and organic matter to the soil. In the longer term, this might increase the risk for future nutrient imbalances and reduced forest production (Egnell & Leijon 1999; Raulund-Rasmussen et al., 2008; Worrell & Hampson, 1997), as well as changes in species composition and biodiversity (Jonsell, 2007). In some countries, such as Finland, stumps may also be harvested, although this will not be considered here.

Forests provide a number of environmental services, such as water protection, carbon sequestration and biological diversity, which need to be maintained both during and after harvesting. Removal of forest residues after harvesting could increase the risk for adverse effects on these services. Thus, there is a potential for conflict between such goals as increased use of forest resources for bioenergy and rural employment on the one hand, and protection of ecosystem services together with long-term site sustainability on the other. In order to minimise the potential for conflict, legislation, certification systems and management guidelines have been developed. However, for these to be effective, there has to be a scientific basis, and there is at present insufficient knowledge about which factors determine the contrasting effects found in field experiments on increased biomass removal (see below), or of how variation in these controlling factors affects long-term site sustainability. This review will address the current state of knowledge regarding sustainable removal of branches and tops for bioenergy from boreal forest ecosystems.

2. Effects of harvesting intensity on soil and water

Nutrient depletion is the major environmental concern regarding WTH as compared with SOH, as this is relevant not only environmentally but also economically due to the risk for reduced growth in the next rotation. As stated above, a large portion of tree nutrients are in the foliage and branches, so removing these from the forest will also remove the nutrients. If these nutrients are not replaced, either by weathering, deposition or fertilisation, reduced growth in the next rotation may result. This risk will vary greatly, depending on site nutrient status, and a nutrient-rich site may tolerate a considerable nutrient removal. However, even on a nutrient-rich site, removal of nutrients without making sure they are replaced is inconsistent with the principles of sustainable forest management. Raulund-Rasmussen et al. (2008) suggested a nutrient balance approach for predicting sites at risk. This will require considerable knowledge of the various nutrient pools and fluxes (shown schematically in Fig. 1), which are sometimes difficult to obtain, leading to a large degree of uncertainty in nutrient balance calculations. A further approach suggested by Raulund-Rasmussen et al. (2008) was to classify forest soils into robust and sensitive types with respect to the risk for nutrient depletion (Table 1). Among relevant factors are temperature, soil depth, soil type (organic/mineral), soil texture, pH and mineralogy. In predicting site sensitivity, knowledge about similar sites is another useful tool. This knowledge can in many cases be obtained from literature studies, e.g. on harvesting experiments or fertilisation experiments.

To minimise the risk of nutrient depletion, it is important to develop methods for leaving the nutrient-rich foliage on site (Helmisaari et al., 2011). In forestry practice, piles of branches and tops are often left in the forest for periods of up to one year before removal, in order for as much as possible of the foliage to fall off (Fig. 2). This allows the return of the nutrients to the site.

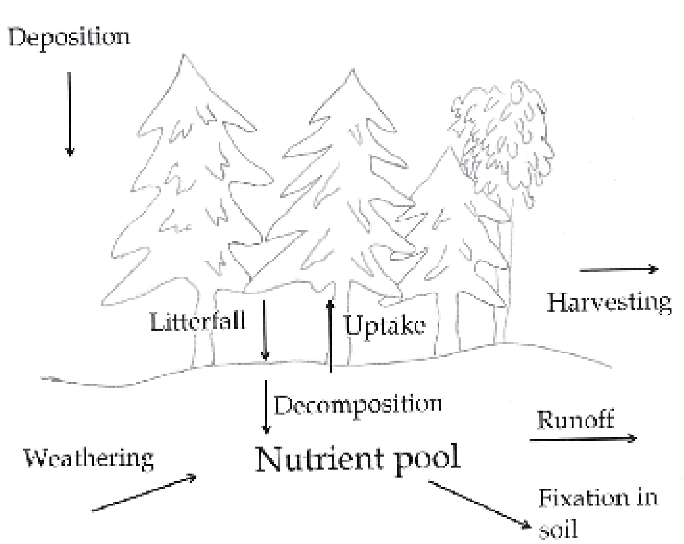

Fig. 1. Schematic overview of nutrient fluxes in the boreal forest ecosystem

Temperature	Depth	Type/texture		pH	Minerals	Sensitivity
<2°C	-	-	-	-	-	S
>2°C	<30 cm	-	-	-	-	S
>2°C	>30 cm	Organic	Fen	-	-	R/S
>2°C	>30 cm	Organic	Bog	-	-	S
>2°C	>30 cm	Mineral	Loamy	<4.8	-	S
>2°C	>30 cm	Mineral	Loamy	4.8-6	-	R
>2°C	>30 cm	Mineral	Loamy	>6	-	R
>2°C	>30 cm	Mineral	Sandy	<4.8	Quartz	S
>2°C	>30 cm	Mineral	Sandy	<4.8	Dark minerals	S
>2°C	>30 cm	Mineral	Sandy	4.8-6	Quartz	S
>2°C	>30 cm	Mineral	Sandy	4.8-6	Dark minerals	R
>2°C	>30 cm	Mineral	Sandy	>6		R

Table 1. Classification of soils into robust (R) and sensitive (S) types, based on Raulund-Rasmussen et al. (2008)

Fig. 2. Removal of a pile of branches and tops six months after harvesting, Gaupen, Norway (photo: Kjersti Holt Hanssen, Norwegian Forest and Landscape Institute)

There is some concern that piling of branches and tops might increase the risk for pest outbreaks, in contrast to direct removal of these residues after harvesting or chipping on-site. However, compared to SOH, piling, if carried out before insect colonisation, might even reduce the risk for outbreaks because larger amounts of wood (on the insides of the piles) would become less accessible to the pests (Schroeder, 2008). Piles of forest fuel from final cuttings are in some cases located close to stand edges of living trees, while whole trees from thinnings may be piled and stored in rows inside the stands; in these cases, there is a clear risk that bark beetles might attack nearby standing living trees (Schroeder 2008). However, it is not certain that the risk is great in practice. Recommendations include avoiding summer storage of large amounts of spruce with a diameter exceeding 10 cm close to mature living spruces, avoiding storage of spruce in thinned stands after warm and dry summers, and avoiding storage of both pine and spruce in defoliated forests (Schroeder 2008). National legislation regarding the amounts of coniferous wood that may be left or stored in the forest exists in many countries.

Returning wood ash to the forest has been suggested as a measure against nutrient loss, as all the major plant nutrients except nitrogen are found in wood ash. Wood ash input increases concentrations of base cations and reduces soil acidity (Arvidsson & Lundkvist, 2003; Brunner et al., 2004). Concentrations of potassium and magnesium in tree fine roots increase (Brunner et al., 2004). However, experiments with ash input on mineral soils have shown no significant increase (or decrease) in growth, probably due to nitrogen limitation (Karltun et al., 2008; Ozolinčius et al., 2007a). On peat soils, the situation is different: Swedish and Finnish experiments have shown an increase in tree growth after ash input (Karltun et al., 2008). There are concerns about increased concentrations of heavy metals after ash input (e.g. Reimann et al., 2008), especially in fungi and berries (Karltun et al., 2008). This will depend on the dose of ash added, and it is recommended to add a dose giving an amount of heavy metals no higher than the amount removed (Swedish Forest Agency, 2001). Effects on ground vegetation were limited when crushed hardened wood ash was used (Arvidsson et al., 2002). There may be a risk for damage to mosses and lichens (Ozolinčius et al., 2007b). Changes in mycorrhizal species composition have been observed (Karltun et al., 2008). The risk for negative effects appears to be low if the ash is treated before use, e.g. by hardening (Arvidsson et al., 2002) or added as granules (Callesen et al., 2007) or pellets (Rothpfeffer, 2007).

Soil organic matter (SOM) is an important reservoir for nutrients, especially nitrogen; its decomposition and mineralisation are important in nutrient cycling. In northern boreal forests, soil temperature and moisture are below optimal for decomposition, and changes in these after harvesting might be expected to increase decomposition and nutrient availability and leaching at least in the short term (Yin et al., 1989). However, increased soil moisture as a result of decreased evapotranspiration might lead to waterlogging and anaerobic conditions in the rooting zone that might inhibit decomposition (Prescott et al., 2000). In fact, the effect of harvesting on soil organic matter is variable. Decomposition rates of surface litter have been found to decrease after clear-cutting (Yanai et al., 2003), while accelerated mineralisation as a result of clear-cutting has been observed in Finland (Palviainen et al., 2004).

The effect of harvesting intensity on soil C and N has been found to vary greatly (Johnson and Curtis, 2001; Olsson et al., 1996a; Vesterdal et al., 2002). In their meta-analysis, Johnson and Curtis (2001) found different effects from different harvesting methods and tree species: SOH of coniferous species appeared to cause an increase in soil C while WTH caused a decrease. SOH of hardwoods, on the other hand, also appeared in most cases to lead to a decrease in soil C. A decrease in soil C was observed independent of harvest intensity for Norway spruce and Scots pine in Sweden (Olsson et al., 1996a). Harvest intensity may affect the decomposition of existing SOM as well as the build-up of new SOM from litter and forest residues.

At present, management recommendations for harvesting do not deal with optimisation of the carbon content of forest soils, although recommendations regarding erosion, soil compaction, drainage, and site preparation will clearly influence the carbon content. One reason for this is our incomplete understanding of the processes involved in carbon cycling in boreal forest ecosystems, and of which factors are most crucial for maximising carbon sequestration in these ecosystems.

Harvesting decreases evapotranspiration and thus increases runoff quantity. Haveraaen (1981) observed that clear-cutting might increase runoff by up to 40% in an area of eastern Norway with shallow soil. Harvesting also influenced water quality: nitrogen loss increased by up to six times (from 1.5 to 7-9 kg/ha), mostly (about 6 kg/ha) as nitrate. Corresponding increases were from 2 to 12-13 kg/ha for potassium, 18 to 24 kg/ha for sulphur in the form of sulphate, and from 16 to 35 kg/ha for chloride (Haveraaen, 1981). Removal of harvesting residues might possibly reduce runoff of these and other elements. Runoff water can become more acid after harvesting (Stupak et al., 2007).

Clear-cutting on Norway spruce-dominated drained peatlands has been shown to cause increased export of dissolved organic carbon (DOC) (Nieminen, 2004). Mineralisation of organic nitrogen followed by nitrification will increase nitrate concentrations. Because uptake is low, this nitrate will be largely leached from the system, together with base cations (Raulund-Rasmussen & Larsen, 1990). Nitrate in deposition will not be taken up to such a large extent as before harvesting, but will also be leached together with base cations. In Sweden, a clear nitrogen leaching gradient has been found on clear-cuts from the west to the east, following the deposition gradient but also influenced by higher runoff in the west (Akselsson et al., 2004). Increased export of all main forms of dissolved nitrogen (nitrate, ammonium and organic nitrogen) has been observed after harvesting (Haveraaen, 1981; Nieminen, 2004). However, small clear-cuts on a nitrogen-saturated site in Germany did not appear to increase the risk for nitrate contamination (Huber et al., 2004). P concentrations did not significantly increase, while P export increased only slightly after harvesting (Haveraaen, 1981; Nieminen, 2004). Base cation fluxes in runoff may increase after harvesting (Haveraaen, 1981; Hu, 2000), as increased decomposition of organic matter may lead to increased concentrations of base cations in runoff water. Piirainen et al. (2004) observed only slightly increased fluxes of P and base cations from below the B horizon after clear-cutting, despite increased fluxes from the O horizon.

Soil water chemistry in forest soils is affected by harvesting, with increased leaching of nutrients such as nitrogen and base cations after harvesting. For example, Hu (2000) found higher nitrate and potassium concentrations in soil water from mineral soils 2-3 years after harvesting and Piirainen et al. (2004) observed that the phosphorus flux under the organic layer increased three times and the base cation flux increased two times after SOH. This leaching is counteracted by growth, partly of ground vegetation (Fahey et al., 1991; Palviainen et al., 2005) and partly of new trees. Removal of forest residues influences soil water, as reduced concentrations of nitrate, ammonium and potassium have been observed (Staaf & Olsson, 1994). Where stumps had been removed, Staaf and Olsson found increased ammonium concentrations for two years, followed by two years of increased nitrate concentrations and acidity. These effects were only temporary: after four years there was no great difference between plots with stem-only harvesting, whole-tree harvesting, and whole-tree harvesting together with stump removal. As effects of harvesting on soil water chemistry change with time, it is important to have long-term experiments.

Significant forest resources are often located in more difficult situations, especially in mountain areas. Due to difficult access and the high cost of traditional (motor-manual) harvesting systems, these areas are currently underused. Today improved technical equipment as well as higher market prices make it possible to harvest also steeper slopes with partly- or fully-mechanized harvesting systems. Depending on the type of the technical system (wheeled or tracked harvesters, skidding/forwarding, cable systems etc.) and the design of the harvesting operation (distance and slope of the skid trails/roads) but also on soil quality and slope, various degrees of erosion and other physical damage to the soil can be observed after mechanized harvesting operations (Worrell & Hampson, 1997). Heavy erosion creates problems for soil, water and technical accessibility in the future. There is concern about the effect of increased sediment loads on water quality downstream of the harvested site: this might affect rural water treatment plants and fish reproduction (Nisbet, 2001). In addition, erosion causes loss of nutrients and organic matter from the forest ecosystem. Methods for reducing erosion risk are well-known, including for example building of culverts, bridges, and silt traps, and these methods have been incorporated in management guidelines in some countries (e.g. Forest Service, 2000; Forestry Commission, 2003).

3. Effects of harvesting intensity on biological diversity

Many organisms are dependent on logging residues as habitats or shelter, so removing this material for fuel will clearly affect these organisms' ability to survive. Species that depend on wood for their survival are termed saproxylic, and in northern Europe there exist several thousand such species, mainly fungi and insects. A further risk is that insects colonise wood bound for heating plants, and are thus trapped in wood that is burned. It is possible to make qualitative recommendations about which types of habitats or wood types that have the most threatened fauna and flora, based on information about landscape history and microhabitat associations (Jonsell, 2007). For example, in Sweden, based on studies of saproxylic beetles, it appears that coniferous wood can be harvested as forest fuel to a rather large extent, whereas deciduous tree species, and especially southern deciduous species and aspen, should be retained to a larger degree (Jonsell, 2007). In addition to saproxylic species, other organisms which feed on them, such as woodpeckers, are also likely to be affected.

Some studies have dealt with effects on ground vegetation. Vegetation retains nutrients in the ecosystem and can decrease nutrient leaching prior to stand re-establishment after clear-cutting (Palviainen et al., 2005). WTH and removal of logging residues leads to reduced amounts of woody debris in clear-cuts and changes in physical and other environmental conditions (Åström et al., 2005), including soil nutrient contents (Staaf & Olsson, 1994), microclimate (Åström et al., 2005), increased light supply and mechanical disturbance. These changes could lead to changed species composition, reduced biodiversity and reduced nutrient content in the humus layer (Olsson et al., 1996a, 1996b). Increased biomass removal may change the abundance of plant species with a key ecosystem role (Bergstedt & Milberg, 2001). Differences in ground vegetation related to felling (selective vs. clear-cutting) have been found as long as 60 -70 years after harvesting (Økland et al., 2003).

Reported effects of increased biomass removal on boreal forest vegetation differ (e.g. Åström et al., 2005; Olsson & Staaf, 1995). Fahey et al. (1991) found that grass biomass increased more rapidly after WTH compared with SOH and continued to make up a higher proportion of the biomass during the first four years after harvesting, while

Bergquist et al. (1999) found no effects of WTH on grasses. Åström et al. (2005) found that WTH reduced bryophyte cover by half (hepatics in particular were affected) and increased graminoid cover with 10% but found no significant effects on other vascular plants, whereas Olsson and Staaf (1995) reported lower cover of most vascular plants after WTH, while bryophytes were unaffected by the logging method. These contrasting results may be due to several factors, e.g. differences in environmental and climatic conditions at the study sites, sampling methods and statistical treatment (T. Økland, personal communication).

4. Effects of harvesting intensity on forest regeneration and productivity

The major concern about WTH from the point of view of the forestry industry has been that the removal of branches and foliage will lead to reduced productivity in the next rotations, as these parts contain a large share of the nutrients in the tree. This has generally (although not always) been found to be the case. In Fennoscandia, Jacobson et al. (2000) demonstrated growth decreases in the first 10 years after WTH in thinnings of Norway spruce and Scots pine stands when compared with conventional thinnings. The growth reduction could be counteracted by nitrogen fertilisation and they concluded that the reduction was due to reduced nitrogen supply. The growth reduction continued in a second ten-year period, but could also then be compensated by fertilisation (Helmisaari et al., 2011). Results from one of the sites included by Jacobson et al. (2000) and Helmisaari et al. (2011), Bergermoen in Norway, are given in Table 2. In the Table, it can be clearly seen that plots where whole-tree thinning had been carried out (Treatment 2) had lower production than plots where stem-only thinning had been carried out (Treatment 1), and that this reduced production could be counteracted using fertilisation (Treatments 3 and 5).

Revision year	1	2	3	4	5
1989	168	163	177	180	182
1994	209	196	216	221	225
1999	255	239	259	265	261
2005	312	293	308	320	313

Table 2. Total production (m^3/ha) by treatment and revision year in an experiment with stem-only vs. whole-tree thinning with and without fertilisation in a Norway spruce forest at Bergermoen, Norway. Thinning took place in 1984. The treatments are: 1) stem-only thinning (SOT), 2) whole-tree thinning (WTT), 3) WTT + NPK compensation fertilisation, 4) SOT + 150 kg N + 30 kg P/ha, and 5) WTT + 150 kg N + 30 kg P/ha. The data are available at http://www.skogforsk.no/feltforsok/Langfig.cfm?Fnr=1057 (in Norwegian)

Egnell and Valinger (2003) also found reduced growth in a Scots pine stand 24 years after WTH as well as branch and stem harvest (BSH). Comparable results have been found in the UK by Proe and Dutch (1994) in second generation Sitka spruce after clear-cutting including removal of residues. However, the effect of WTH seems to be site-dependent as well as species-dependent, as not all studies have shown an unambiguous nutrient decrease with subsequent growth reduction after whole-tree harvesting (Egnell & Leijon, 1999; Olsson et al., 1996a). Results from the North American Long-Term Soil Productivity study showed only a limited effect of WTH compared to SOH: although growth decreased, seedling survival was in fact improved five years after WTH (Fleming et al., 2006).

5. Legislation, certification and management recommendations

As mentioned above, the increased use of renewable energy sources, including forest biomass, is a marked characteristic in current energy policy. In forest policy, the use of forest biomass for energy is usually supported as a sustainable form of energy that contributes to social welfare, rural development and the forest economy. Energy legislation is used directly as a tool to promote renewable energy including forest and other biomass, whereas forest legislation rather works to ensure sustainably produced forest biomass for all uses (Stupak et al., 2007). However, increased use of forest biomass for energy might lead to conflict between different interests, all of which are politically important: on the one side, the need for a secure and renewable source of energy as well as rural employment, and on the other ecologically sound long-term timber production, biological diversity and other uses of the forest such as recreation. Trade-offs between these various interests will then be necessary, and increased knowledge is essential in order to optimise these trade-offs. Sustainability principles and criteria have therefore to be incorporated into policy frameworks and support schemes, as well as management guidelines. Many countries have produced national recommendations and guidelines for forest fuel extraction and/or wood ash recycling to encourage the extraction of forest fuels taking place in agreement with the principles of sustainable forest management. Certification is another approach: the main forest certification schemes are the Programme for the Endorsement of Forest Certification schemes (PEFC) and Forest Stewardship Council (FSC). In national PEFC and FSC standards, issues related to wood for energy are included under several criteria. Recommendations elaborated by governments or other groups of stakeholders could be used for further development of legislation, certification standards, and guidelines in relation to the sustainable use of forest biomass for energy. Recommendations vary according to subject, but on the whole, economic, ecological and social questions are treated for the whole forest fuel chain, from removal of biomass from the forest to recycling of wood ash to the forest (Stupak et al., 2007). Scientific results must be interpreted and transferred to operational criteria, indicators, recommendations and guidelines, with the final thresholds being set by politicians, certification bodies or other stakeholders (Stupak et al., 2007). This interpretation will necessarily include a large degree of uncertainty, so that continuous further development will be necessary as new knowledge is obtained.

6. Conclusions

Removal of forest residues for bioenergy after harvesting might increase the risk for adverse effects on the environmental services provided by forests, such as water protection, carbon sequestration and biological diversity. Forest legislation, certification systems, and management guidelines have been developed in order to reduce the risk for non-sustainable use of forest resources. However, not enough is known at present about which factors determine the contrasting effects found in field experiments, and more research is therefore needed, and further development of legislation, certification standards and management guidelines is likely to take place as new knowledge is obtained.

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10

Soil Compaction – Impact of Harvesters' and Forwarders' Passages on Plant Growth

Roman Gebauer, Jindřich Neruda, Radomír Ulrich and Milena Martinková
Mendel University in Brno, Brno
Czech Republic

1. Introduction

The goal of forestry management is to sustain continual development of forest ecosystems that optimally fulfil their productive and non-productive functions. In order to achieve this goal, the full productive capacity of forest stands needs to be maintained while respecting all the natural processes in the soil, including microbiological organisms, physical properties, nutrient reserves and regeneration processes of the ecosystem.

We need to approach herbs as well as woods holistically, including the root system architecture and functions. Growth of the above-ground system depends on the state of the root system functions, and vice versa. If the conditions for an activity of the root system are limited, the functioning of the above-ground system will be limited too.

During thinning activities in all age groups of forest stands and during the subsequent recovery, progressive harvesting technologies that use mobile means of mechanisation (predominantly harvesters and forwarders) are applied more and more commonly. In contrast to the motomanual technologies that were used in the past, harvesters and forwarders are considerably safer and more productive. However, the passage of heavy machinery on the soil surface causes disruption of the soil environment and mechanical damage to roots. In 1947, it was found that harvesting disrupted soil by modifying its structure and moisture characteristics (Munns, 1947). Despite more than sixty years of research, we still do not fully understand the impact of soil compaction on forest productivity. Due to the global interest in maintaining forest resources and the sustainable development of forest production, a number of conferences have been organised, including the Earth Summit in 1992, which gave rise to the Montreal Process (Burger & Kelting, 1998). At this summit, soil compaction was defined as one of the soil indicators of the forest health state.

Soil compaction is affected by both endogenous and exogenous soil factors. Horn (1988) defined the following endogenous factors as responsible for soil compaction: distribution and size of soil elements, type of clay mineral, type and amount of absorbed cations, content of organic matter, soil structure, soil stabilisation, topsoil material, bulk density of soil, pore continuity and water content. Exogenous factors include the duration, intensity and means of wood harvesting and wood loading. For instance, different machines, or even the same machines with different tyres, differ in their loading and pressure on the soil. Work by Greacen & Sands (1980) and Ole-Meiludie & Njau (1989) support the finding that the compaction rate depends on the concrete soil characteristics, pressure and vibrations of the machines. The rate of soil erosion varies depending on the loading technology and intensity of harvesting. Generally, soil is disrupted by harvest cutting more than it is by selective logging or thinning (Reisinger et al., 1988). The high number of variables leading to soil compaction makes it difficult to find a single parameter that best defines the impact of the passage of a harvester or a forwarder.

2. Harvester and forwarder machinery

Most of the machines currently in use today are heavy and wheeled. The interaction of the wheels with the soil surface in a stand during harvesting and forwarding activities puts pressure on the soil, the intensity of which depends on tyre inflation, toughness and adhesive loading of the traction mechanism. Brais (2001) identified soil compaction by the passage of forestry machines as one of the main factors in soil degradation. Soil compaction during harvesting usually changes the soil structure and moisture conditions by disruption of soil aggregates, decreased porosity, aeration and infiltration capacity, and increased soil bulk density, soil resistance, water interflow, erosion and paludification (Kozlowski, 1999; Grigal, 2000; Holshouser, 2001). Soil compaction may become even more problematic as the weight of harvesters and forwarders increases (Langmaack et al., 2002).
A harvester is a mobile, multi-operational machine that can fell timber, cut branches and chop trunks into assorted lengths in a single cycle (Fig. 1). Individual cut-outs remain in the stand in piles and heaps. The entire process is fully mechanised and automated. Harvesters are classified into four groups based on the kind of undercarriage (wheeled, tracked, walking and combined harvesters). The undercarriage of multi-operational machines has two sections linked by an articulated joint. A forwarder collects the logs made by a harvester and loads them onto a load section of a tractor and forwards them to a storage area (Fig. 2). The main loading function is carried out by a hydraulic crane that reaches 6-10 m with a rotator and a grab.

Fig. 1. Harvester John Deere 1270E with a rotating cab

Fig. 2. Forwarder John Deere 810 D at the platform balance

Forest managers have to concern the total weight of forwarders for particular applications and also the maximal load of tyres has to be observed. Prescribed values for allowable load of tyres according the German Forestry Council (KWF) are given in Table 1. The maximum allowable load of tyres should be up to 4.9 tunes with optimal load up to 4.0 tunes.

Max. weight of forwarder in tunes	Total weight of forwarder (with load) in tunes	Ratio of load on loading part	load of tyre in tunes
8	20	65%	3.2
12	26	65%	4.2
14	30	65%	4.9
16	38	65%	6.2

Table 1. Values for allowable load of forwarder tyres according the German Forestry Council (KWF).

3. Impact of the passage of harvesters and forwarders on soil

The soil compaction that occurs as a consequence of the passage of harvesters and forwarders is connected with significant changes to the soil structure and moisture conditions (Standish et al., 1988; Neruda et al., 2008). Increased bulk density of soil, decreased porosity, decreased water infiltration, increased erosion and changes in plant physiology can all arise from soil compaction. Other changes include the disruption of soil aggregates and loss of pore continuity (Kozlowski, 1999).

3.1 Soil bulk density

Higher soil bulk density is caused by lower porosity and lower water capacity, and it can inhibit root growth (Gebauer & Martinková, 2005). Soil compaction usually occurs in the 30 cm surface layer of soil, which contains the majority of the root biomass (Sands & Bowen, 1978; Kozlowski, 1999) (Fig. 3). The bulk density of soil in the upper layers (0-8 cm) increases by 41-52% after the passage of tractors (Kozlowski, 1999). In the case of a forwarding line, the bulk density of soil in the surface layers (0-10 cm) rose by 15-60% and, in the case of a crossing line, it increased by 25-88% (Lousier, 1990). The compaction decreased in deeper layers; nonetheless, it was recorded even at depths of 30 cm and more. The highest rate of compaction occurred during the first several passages of tractors (Lousier, 1990). The following passages had less effect, but could still lead to rates of compaction that might significantly affect root growth. The critical value of soil bulk density ranges from 1200 to 1400 kg m^{-3}. When this value is exceeded, root growth is reduced in most soil types (Lousier, 1990).

Fig. 3. Superficial root system of a Norway spruce tree showing the majority of the roots growing in the upper soil layer

3.2 Soil porosity

Soil compaction changes the porosity by reducing macroscopic spaces and raising the number of microscopic spaces. The change in porosity affects the balance of soil air and water in pores, which is critical for plant growth. Soil air is a gaseous compound that exists in pores that are not filled with water. Compared with atmospheric air, it includes less oxygen and more CO_2 (ranging from 0.5 – 5% or even higher) (Hillel, 1998). The higher CO_2 content in the soil arises from root respiration and the aerobic decomposition of organic matter. Grable & Siemer (1968) defined the critical value of aeration for plant growth as 10% porosity. Soils with a high content of CO_2 and a low content of oxygen are poorly aerated, and there may even be anaerobic conditions within such soil (Hillel, 1998). A concentration of CO_2 in the soil higher than 0.6 % indicates significant changes to the soil structure that can impact root growth (Güldner, 2002). Our measurements show that this critical value was significantly exceeded in almost all cases after the passage of harvesters and forwarders, and in some cases, the value was exceeded by severalfold (e.g., 1.2% and 3.4% CO_2 in a harvester track as opposed to 0.4 % and 0.5% CO_2 on the surface unaffected by harvesters) (Fig. 4).

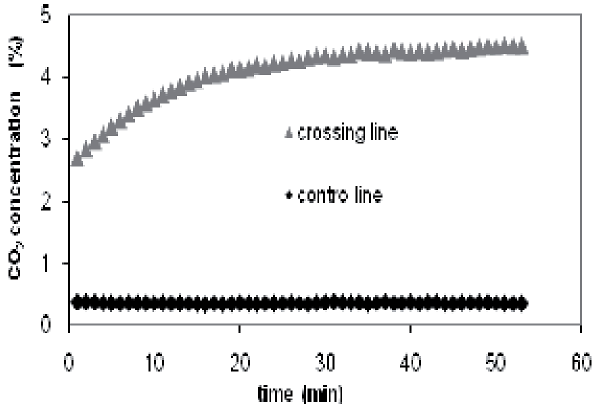

Fig. 4. Upper panel - Concentration of CO_2 in soil air in a crossing line after several harvester passages (soil moisture: 35%). Lower panel – CO_2 measurement in control line with GMP 221 Carbon dioxide probe (Vaisala, Finland)

3.3 Water infiltration and erosion

Soil compaction is often related to the creation of crust, causing decreased water infiltration and ultimately increasing water runoff (Malmer & Grip, 1990). In the places where water runoff is not possible (e.g., holes after passage, terrain depressions), there is weak drainage, which causes local inundation (Jim, 1993) (Fig. 5). Experiments have shown that harvesters and forwarders can accelerate the rate of surface erosion from 2 to 15 times, compared with unpassaged soil and 85% of the total surface erosion appears in the first year after disruption (Lousier, 1990).

We should consider the soil capability i.e. the ability of soil to cope with external forces, which can cause permanent or temporal deformation, when heavy machines are moving in the forest. The rut depth from 15 - 50 cm (according the soil humidity) brings high ecological risk (Fig. 6). The soil capability of different soil types is given in Table 2.

Fig. 5. A case of unsuitable preparation of a site with disruption of soil aggregates. If an Eco-Baltic wheeled track had been used, the lines would not have been cut to a depth of 50 cm and deeper along the way.

Fig. 6. A case of rut depth up to 25 cm, which is a point when an ecological risk may appear.

3.4 Plant physiology

3.4.1 Disorders in photosynthesis and water regime

Heavy compaction leads to a variety of physiological disorders in plants. Roots react to soil compaction by increasing demand for photosynthates (Zaerr & Lavender, 1974), which are needed to support the metabolism required to overcome the increased soil resistance to elongation growth. The physiological cost of recovering the functions of fine roots may be as high as 70% of the accessible carbon flow (Ågren et al., 1980; Vogt et al., 1996). Kozlowski (1999) found that the increased carbon flow due to soil compaction leads to an overall decrease in photosynthesis. This is a result of reduced foliage surface, which is an outcome of reduced water intake caused by changes in the soil structure and moisture conditions (Arvidsson & Jokela, 1995). Therefore, a plant might not have enough energy to reconstruct its root system, and the growth of roots as well as the above-ground parts stagnate or even die. Reduced foliage surface is a reaction to a water deficit in the leaves, which is brought about by soil compaction and may lead to the closing of pores and further loss of photosynthesis (Masle & Passioura, 1987).

degree of resistance	soil capability	rut depth, soil consistence	soil taxonomy
1	extremely low dry: 30 -50 kPa wet: 5-12 kPa	≥ 35 cm, incohesive, strongly crumble, slush	Histosols, Gleysols
2	very low dry: 50-140 kPa wet: 12-22 kPa	25-35 cm crumbly, clay, loam, very soft	Stagnosols, gleyic Stagnosols
3	reduce dry: 140 - 300 kPa wet: 18-50 kPa	15-25 cm hardly dig, loam, sandy clay, soft	Cambisols, Luvisols, Fluvisols - subtype - gleyic
4	slightly reduce dry: 300-600 kPa wet: 50-80 kPa	7-15 cm hardly dig, solid, sandy loam	dry and slightly wet Cambisols, Luvisols, Regosols, Chernozems
5	bearable dry: > 600 kPa wet: 80-120 kPa	< 7 cm solid, hard, stony	Podzols, Leptosols

Table 2. Soil capability measured as a rut depth after one passage of the special forest tractor (LKT 80) with inflation of tyres 200 kPa. Dry and wet means humidity of sandy and loam-sandy soil 4-8 % and 18-30%; sandy-loam and loam soil 8-15% and 35-45%; clay-loam and clay soil 15-25% and 45-55%, respectively.

3.4.2 Disorders in nutrient uptake

Often, extreme soil compaction leads to reduced absorption of mineral nutrients by the roots, especially nitrogen, phosphorus and potassium. Nutrient uptake is reduced as a result of the loss of minerals from soil, reduction of root access to nutrients and decreased root capacity for nutrient intake (Kang & Lal, 1981; Kozlowski & Pallardy, 1997). A reduction of nutrient uptake caused by soil compaction in the upper as well as deeper soil layers (Kozlowski, 1999) might be the reason for different reactions to the compaction among species, as some have higher nutrient demands than others.

3.4.3 Effects on mycorrhizas and plant hormones

Soil compaction also affects the structure, development and function of mycorrhizas (Entry et al., 2002) and causes changes in the levels of stress hormones in plants, mainly abscisic acid and ethylene (Kozlowski, 1999).

3.4.4 Respiration disorders

Soil compaction induces hypoxia, which is related to the reduction of aerobic micro-organism activity and an increase of denitrification. As compaction increases, reduction of macro-pores enhances the development of anaerobic spaces (Torbert & Wood, 1992). Insufficient aeration of compacted soils leads to anaerobic respiration in roots and insufficient energy for maintaining the basic root functions, namely nutrient uptake (Kozlowski & Pallardy, 1997).

4. Impact of compaction on plant growth

Several studies have shown that tree growth and wood production decrease with increasing compaction (Froehlich, 1976; Cochran & Brock, 1985). Growth inhibition as well as the death of woody plants caused by soil compaction has been documented in zones of recreation, harvesting areas (Sand & Bowen, 1978; Cochran & Brock, 1985), agro forestry (Wairiu et al., 1993) and tree nurseries (Boyer & South, 1988).

Soil compaction strongly reduces plant growth as it limits root growth (Rosolem et al., 2002; Gebauer & Martinková, 2005). There is a non-linear relationship between root elongation and soil resistance in the majority of plants (Misra & Gibbons, 1996). Because compaction usually occurs in the upper soil levels, species with a surface root system are disadvantaged (Godefroid & Koedam, 2003). Generally in the case of large trees, root growth is limited by increasing soil bulk density and excessive soil resistance (typical in dry and skeletal soils) or insufficient aeration if the soil is heavily saturated by water (Greacen & Sands, 1980). The greater the root growth reduction and the smaller the soil space occupied by roots, the slower the growth of a tree in its above-ground parts (Halverson & Zisa, 1982; Tuttle et al., 1988).

The exposure of roots to mechanical pressure induces a number of physiological changes that have been well described on the macroscopic level. For example, the elongation growth decreases, and the response period varies from several minutes (Sarquis et al., 1991; Bengough & MacKenzie, 1994) to many hours (Eavis, 1967; Croser et al., 1999). The root tip generally rounds, becoming concave, the root width behind the meristem increases and the root meristem and the elongation zone shorten (Eavis, 1967; Croser et al., 2000). The data on root thickening behind the root tip demonstrate the effects of long-term mechanical pressure on the root tips (Abdalla et al., 1969; Martinková & Gebauer, 2005). The growth of roots is reported to be a more sensitive indicator of soil disruption than the growth of the above-ground parts (Singer, 1981; Heilman, 1981) because the reduction of root growth precedes the phase when the extreme soil resistance is achieved (Eavis, 1967; Russell, 1977; Simons & Pope, 1987).

The critical value of soil resistance that can lead to significant physiological changes is measured by penetrometers (Atwell, 1993; Greacen & Sands, 1980) (Fig. 7), which better express conditions of root growth as penetrometers also measure the influence of bulk density and soil moisture (Siegel-Issem, 2002). Heavy, humid soils are more easily penetrated by roots due to lower soil resistance, while in arid soils of the same density, the growing resistance limits root growth. Critical values of compaction, expressed by penetrometric soil resistance, for different kinds of soil are listed in Table 3. It has been determined that a soil resistance of 2.0 MPa or more causes root shortening in most plant species (Atwell, 1993). The critical soil resistance on compacted sands limiting root growth measured for *Pinus radiata* was 3.0 MPa (Sands et al., 1979). However, roots usually have a lower resistance to soil penetration than the resistance measured by penetrometers, due to the radial expansion and smaller diameter of roots and the ability to curl and minimise friction by means of polysaccharide slime.

Only a few studies, mainly using herbs, have measured the soil resistance against roots directly in soil (Eavis, 1967; Misra et al., 1986; Bengough & Mullins, 1991; Clark & Barraclough, 1999). Roots were found to be capable of exert the outer pressure from 0.9 to 1.3 MPa (Gill & Miller, 1956; Barley, 1962; Taylor & Ratliff, 1969). Eavis (1967) demonstrated that elongation of roots in peas was reduced by 50% at a pressure of 0.3 MPa. Our measurements show that soil compaction causes reduced root elongation growth in Norway spruce by 50% compared with control seedlings (Gebauer & Martinková, 2005) (Fig. 8). In the case of one-year-old buds of Scotch Pine (*Pinus sylvestris*), the soil compaction did not have a significant impact, but for Macedonian Pine (*Pinus peuce*) of the same age, the root growth was negatively affected by soil compaction (Mickovski & Ennos, 2002; 2003). The authors of this study reasoned that the weak impact on *Pinus sylvestris* was due to the fact that its roots have thinner diameters than those of *Pinus peuce*.

Fig. 7. Measurement of soil resistance by penetrometer

Soil type	**penetrometric soil resistance (MPa)**
Sandy loam and sand	more than 4
Sandy clay	4 – 3.7
Silt	3.7 – 3.5
Silty clay	3.5 – 3.2
Clay	less than 3.2

Table 3. Critical values of penetrometric resistance of soil types

The above study shows that compaction significantly reduces plant growth; yet, other studies show that the compaction of soils with a coarse structure (sandy soils) might have a positive impact on the growth of conifers. This contradiction may be because the compaction of sandy soils creates microscopic spaces and enhances water retention in the soil (Troncoso, 1997; Gomez et al., 2002; Siegel-Issem, 2002). Mild soil compaction in sand supports the contact between roots and soil, resulting in higher absorption of water and nutrients (Gomez et al., 2002; Alameda & Villar, 2009). Alameda & Villar (2009) found that a mild compaction positively affected the growth of 53% of seedlings from 17 species (including both foliage and coniferous seedlings) growing in controlled conditions. Miller et al. (1996) found that in forwarding lines with an increased soil bulk density of 40% or more, growth was not affected at all, and 8-year-old seedlings of *Pseudotsuga menziessi* and *Picea sitchensis* survived.

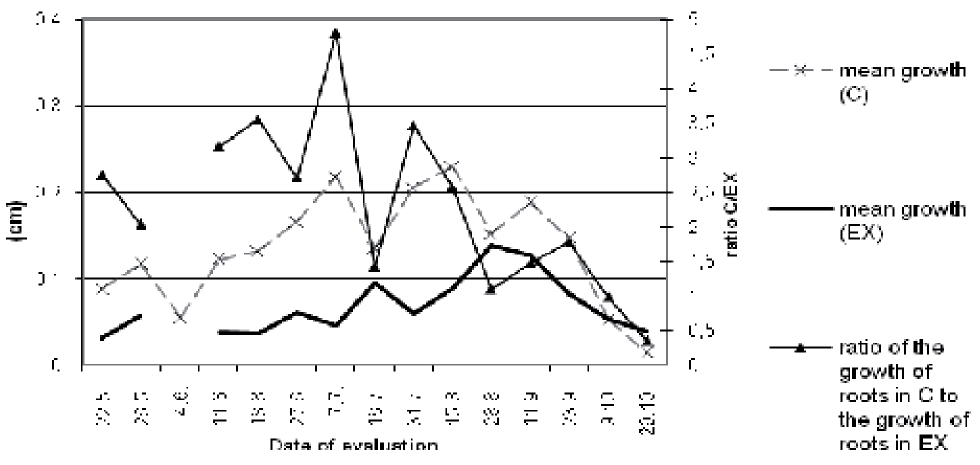

Fig. 8. Root growth and dynamics of Norway spruce seedlings grown in control non-compacted root boxes (C) and in root boxes exposed to a long-term pressure of 5.1 kPa (EX). A C/EX ratio above one indicates higher root growth in the non-compacted soil (Gebauer & Martinková, 2005).

In general, soil compaction is a stress factor that negatively affects the growth of plants, but the rates of compaction and differences among soil types need to be taken into account in these analyses (Kozlowski, 1999; Alameda & Villar, 2009). For instance, Alameda & Villar (2009) showed that growth increases in most seedlings grown in a sandy substrate with rising compaction of 0.2-0.6 MPa, but exceeding this value generally led to a reduction in growth.

5. Recording of harvesters' and forwarders' pressures on soil

During the passage of heavy vehicles on unsurfaced soil, the soil environment gets disrupted and roots are mechanically injured. A method for measuring and recording the immediate pressure on soil was developed and tested by the institute of Forest and Forest Products Technology of MENDELU in Brno (Czech Republic). This method is applicable in forest stands that grow on mild soil surfaces where large and extremely heavy machines (forwarders) pass. Pressure sensors were placed in the soil near the surface, and a unique measuring chain was used to measure the immediate pressure on the soil.

The pressure on a concrete point (e.g., a root or stress sensor) exerted by a wheel is short-lived (approx. 0.04 s) and has a stress impulse character (Fig. 9). The impulse does not have a permanent value, so its rise, apex and fall can be clearly observed. The apex values of stress impulses were used in measuring the stress on the soil. This method is helpful for determining suitable precautions in forestry management, e.g., the effect of different covers on soil protection and the optimal height of the layer. Moreover, this method establishes the optimal inflation of tyres because over-inflated tyres, even the low-pressure type, lead to higher stress on the soil.

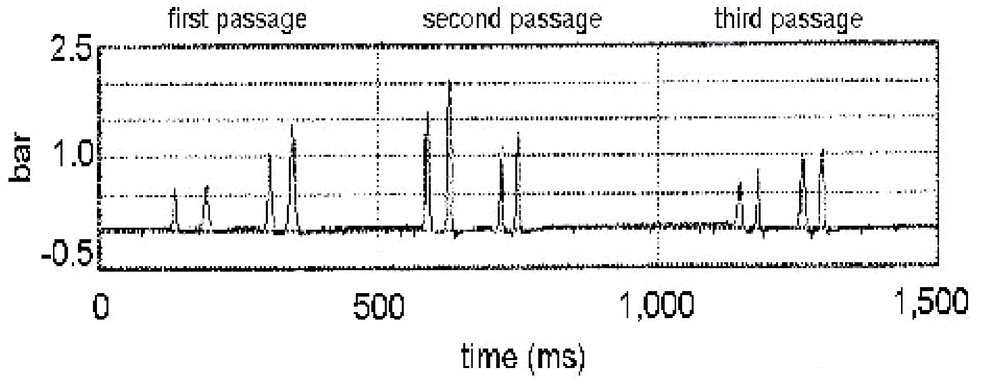

Fig. 9. Recording of the measurement of soil pressures during a forwarder's passage

6. Recovery of compacted soil

Revitalisation and amelioration of compacted soil is a long-term process and it is not known if it is fully achievable (Heninger et al., 2002). The regeneration period after the compaction may be less than 10 years near the soil surface (Thorud & Frissel, 1976; Lowery & Schuler, 1994), but others claim it could last several decades (Wert & Thomas, 1981; Jakobsen, 1983; Froehlich et al., 1985). It is necessary to fully understand the process of compaction, its impact on soil and plant growth and to find means and technologies that minimise the influence of compaction (if at all possible).

The recovery of compacted soil is a result of the combination of root activity, freeze-melt cycles and humid-dry cycles (Reisinger et al., 1988). After a period of 5 years, the bulk density of the surface, which consists of fine sandy-silt soil, was higher by 12% in former lines compared with places unaffected by the compaction (Lockaby & Vidrine, 1984).

The revitalisation of compacted soil also depends on the content of the organic matter in the soil, as it has a significant impact on the soil structure, aeration, water retention and chemical properties. Soil bulk density and porosity increase or decrease with the growing content of organic matter (Childs et al., 1989). Differences of 2-5% may significantly affect soil properties such as bulk density and porosity in sandy soils (Rawls, 1983).

We do not know of any ways to revitalise compacted forest soil on a large scale by technical means or technologies. Thus, it is necessary to prevent soil compaction by forestry management.

7. Prevention of soil compaction

The rate of soil compaction varies considerably depending on the method of felling, the type of soil preparation, the terrain conditions, the timing of the activity and the preparation and personal responsibility of the workers. Soil disruption by harvesting is also affected by soil conditions during the activity (e.g., soil resistance, humidity, frost, snow cover), concrete features of the activity (e.g., frequency of passages) and the impact (stress and vibration) on the soil by harvesters and forwarders.

During the movement of heavy tractors through areas with little bearing capacity of the subsoil, permanent deformations of terrain (lines 20 – 50 cm deep) arise. Even though these lines might be relatively short (5 – 15 m), they make the given section permanently impassable and inaccessible to wheeled or tracked tractors. Such sections include friable sand, drift sand, wet sand, permanently flooded places, passages to bridge inundated areas of watercourses, ford beds, passages in marshy or peaty terrain and dumps. Subsoils at extreme risk include clay soils, because they absorb high amounts of water and their bearing capacity is problematic in the spring and autumn. This highlights the necessity of clearing such a stand prior to activities on weakly bearing terrain.

Preparation of weakly bearing surfaces for harvesting is carried out in two ways:

1. The forwarding route is reinforced with additional material.
2. The road structure is temporarily reinforced (gabions, plastic mobile grids, plastic mobile boards, low-pressure tyres, route reinforcing –old used forest fences, harvesting waste). The extent of the reinforcement needed mainly depends on the axle pressure of the vehicle, construction and strength of the road, mechanical and physical properties of the terrain and the required number of passages of the vehicle.

The advantage of grids and screens is that they are quick and easy to use (Fig. 10). Local reinforcement of a road by means of screens can be achieved along the whole route for minimal costs. After pressing through the bottom layers of the soil, the skid of the wheels on the screen falls rapidly too. The producer recommends 8 tons as the maximal bearing capacity of screens; however, they have been successfully tested with forwarders loaded with 10 – 15 tons (Schlaghamersky, 1991; Ulrich & Schlaghamersky, 1997). Placement of a screen can open the way to a very wet biotope without soil damage by deep lines. One disadvantage of screens is that they cannot be placed directly on unprepared terrain; the lines resulting from the wheels need to be filled with brushwood or harvesting waste, for example. After a long period, soil gets through the screen and needs to be removed by a blade.

Fig. 10. Plastic mobile grids are quick and easy to use.

Besides the proper preparation of the terrain for the passage, there are other ways of minimising soil compaction by the modification of harvesting technologies. For example, the application of lighter technology (Jansson & Wästerlund, 1999), lower inflation of tyres (Canillas & Salokhe, 2001), placement of harvesting waste in locations where harvesting and forwarding is planned (Hutchings et al., 2002) (Fig. 11), harvesting in winter on frozen soil (Alban et al., 1994), planting species tolerant to compaction (Bowen, 1981; Ruark et al. 1982) and limitation of drawing logs using a winch can all help reduce soil compaction. Limitation of the number of passages would not help because 80% of soil compaction occurs during the first passage (Holshouser, 2001). The most efficient precaution is prevention against soil compaction, as the other methods might be ineffective and, furthermore, could do harm to the roots (Howard et al., 1981).

Fig. 11. Placement of harvesting waste in places of forwarders' and harvesters' passages is one way to minimise soil compaction.

8. Conclusion

The passage of forestry machines causes soil compaction, leading to significant changes in the soil structure and moisture conditions. When soil is compacted, soil bulk density increases, porosity and water infiltration decrease, erosion speeds up, and all of these processes lead to changes in plant physiology. Photosynthesis, transpiration, nutrient uptake, mycorrhizas and plant hormones are all possible avenues for these changes.

Soil compaction is influenced by endogenous soil factors (distribution and size of soil elements, soil bulk density, pore continuity, water content, etc.) as well as exogenous factors (choice of equipment, loading of wood, length of loading, intensity and means of harvesting, site preparation, etc.). When soil is compacted, the soil resistance grows; resistance over 2.0 MPa, as measured by penetrometer, limits elongation root growth in most plant species. Our measurements have shown that this critical value is often exceeded when forestry machines pass through an area without any preparation of the site.

Poor aeration of soil caused by soil compaction also prevents the development of root systems and limits the water penetrability of roots. Our measurements show that the critical value of CO_2 in the soil air (defining the rate of aeration) was exceeded as a result of the passage of forestry machines in almost all cases. To establish the optimal inflation of tyres the pressure sensor (a sensor developed and tested by us) was found to be very useful tool.

This sensors are also applicable in forestry management because it aids in the determination of suitable precautions, e.g., whether the soil surface is covered with a sufficient layer of brushwood.

Although compaction is usually considered to be a factor of growth deceleration, some studies of conifers show that compaction of certain soils with a coarse structure (sandy soils) may, on the contrary, enhance growth due to the multiplication of microscopic pores, thus increasing the soil's capability to retain a higher amount of water.

Since the revitalisation and amelioration of compacted soil is a long-term process, and it is not unknown if it is fully achievable, compaction should be minimised as much as possible. Its minimisation could be achieved by the modification of technologies in forestry activities; for instance, by using lighter machines, reducing tyre pressure, placing harvesting waste in places where forestry machines are expected to pass, harvesting in the winter on frozen soil and controlling tractor movement. We should also mention that human factors play often a critical role in the soil compaction.

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Section 5

Biological Diversity

11

Close-to-Nature Forest Management: The Danish Approach to Sustainable Forestry

Jørgen Bo Larsen
Forest & Landscape, University of Copenhagen
Denmark

1. Introduction

How should tomorrow's forests look and which future climatic conditions should they prevail? What kind of goods, services and experiences should they be able to provide; what kind of functions should they be able to perform? These are some of the multifaceted questions forest management faces today.

Forestry policy objectives have grown into a broad range of benefit provisions, other than serving exclusively as the traditional timber suppliers. Today we thus address multiple-use forestry. Production of wood commodities and securing carbon storage is central, but does not necessarily rate above the creation of non timber forest products. Increasingly highly esteemed qualities, such as protecting landscape amenities and cultural heritage, nature conservation and environmental protection, as well as the entire chapter of recreational interests are considered. Consequently, economic and technical efficiency is still prioritized, but ecological and social parameters are progressively taken into account to ensure the multiple use.

For these reasons, silvicultural strategies are required to develop economically productive forests with a high potential for nature conservation, ecosystem protection, and social values. One promising management strategy is to incorporate structural qualities and functional features of natural forest ecosystems – "to follow and assist nature in her development" as already stated 230 years ago by the Danish forester von Warnstedt (Decree of 1781 regarding the management of the Royal forests). This approach can be summarised by the term "nature-based silviculture" or "close-to-nature management" (Gamborg & Larsen, 2003). In North America, on a more general forest management level, 'ecosystem management' and 'adaptive management' can be recognized as part of this trend (Franklin et al., 2002). The aim is to reform current practices so that they are still profitable, but more environmentally benign and even more sensitive to the complexities of nature conservation and the multiple, varying and steadily increasing demands of society by mimicking natural forest structures, their processes as well as their dynamics (Angelstam et al., 2004; Lindenmayer, et al., 2006; Hahn et al., 2007; Larsen et al. 2010).

The disturbances and regeneration processes in natural forest ecosystems, which cause structural heterogeneity at both large and small scale levels are linked to regional characteristics of climate, soil, and species compositions. These processes are being expressed as e.g. infrequent, large-scale storm and fire-driven disturbances in boreal ecosystems and as frequent, small-scale disturbances in Central-European forests. Hence models, describing the region-specific disturbance patterns, such as the forest cycle model, should be used in the development of applied silvicultural methods in such natural ecosystems (Hahn et al., 2005).

In central and western Europe the forest cycle models have been successfully applied to describe the temporal and spatial dynamics and cyclic preoccupation of a specific forest type in natural forest reserves (Leibundgut, 1984; Christensen et al., 2007; Larsen et al., 2010). Such models could serve as an adequate basis for close-to-nature forest management.

The use of natural disturbance regimes to guide human management (i.e thinning and cutting systems) must, however, be complemented with other measures to restore naturalness in forest management. Lindenmayer et al. (2006) emphasize the importance of maintaining aquatic ecosystem integrity for biodiversity protection in managed forests. Hence, maintaining and restoring natural hydrology in forests previously subjected to stand management operations (such as drainage) is important. Therefore promoting species and forest structures that reflect and emphasize the variation in hydrology is an integral part of close-to-nature management, thus contributing to habitat richness in forest landscapes.

One of the basic axioms of nature-based forestry is the mimicking of natural structures and processes in order to obtain a high degree of stability within the ecosystem and thus a high degree of flexibility. All of this necessary to opening up for possible future demands and needs from various players, such as landowners, interest groups and society in general. The logic of this assumption might be best illustrated by considering the contrary position: *without stability – which functions will we be able to sustain in the future.*

One of the major problems experienced with the classical forestry approaches, is the lack of ecological and structural stability and the limited flexibility, towards addressing various at present unknown future demands (who would 20 years ago have predicted the present focus on biodiversity in forest management?). An approach, which has alerted us of the necessity to search for better management systems aiming at increasing functionality as well as flexibility in forest ecosystems; both in relation to multiple uses. In other words: We seem to face a high potential for both ecological adaptability (resilience) and functional flexibility of forest ecosystems, when opening up for greater functional integration – a central aspect of close-to-nature management.

To illustrate the differences between the traditional plantation approach, the close-to-nature approach, and a strict nature protection approach, a "goal-fulfilment assessment" and comparison of the three different management approaches is shown in table 1.

Table 1 indicates that the plantation approach and the conservation approach both are rather narrow and inflexible in their goal-fulfilment, while the nature-based wood production approach is broad and flexible in its goal-fulfilment. The weaknesses of the plantation approach focussing on timber in short rotations and neglecting most natural, cultural and social values are clearly reflected in the table. Further, the plantation approach often leads to less robust forests stands. The conservation approach obviously performs strongly in all nature protection goals but is consequently not, or less able, to deliver on socio-economic goals thus scoring rather low in terms of flexibility to changing goals.

The maintained focus on production economy in combination with relative high scores in ecological as well as social values addressing the needs for stability, explains why the nature-based approach to sustainable forest management has been chosen in many countries. The integrative ability and flexibility of the nature-based approach to fulfil different management goals is a key feature of this management type. Because of this feature, it is possible to gradually adjust the course of management to address the ever-changing objectives and aspirations of society.

Management approach	**Plantation (production) approach**	**Nature-based (integrative) approach**	**Nature protection (conservation) approach**
Specific management goals	Focus on timber production and direct economic outcome	Flexible wood production, nature protection and recreation	Strict forest reserves following natural structures and processes
Production of timber	+++++	++++	+
Economic outcome, long term	+++	+++++	+
Economic outcome, short term	+++++	+++	+
Production of quality timber	++++	++++	+
Biodiversity protection	+	+++	+++++
Protection of wetlands	+	+++	+++++
Ecosystem integrity	+	++++	+++++
Aesthetic qualities	+	+++++	+++++
Landscape integration	++	++++	+++++
Historical and cultural values	+	++++	+++
Space for public recreation	++	++++	++
Place of quietness/meditation	+	+++	+++++
Hunting qualities	+++	++++	+
Robust and resilient forests	+	++++	+++++
Flexibility to changing goals	+	+++++	+

Table 1. Different management approaches and their respective fulfilment of different specific management goals. The scale from 1 to 5 plusses, and '+' = low goal fulfilment, whereas '+++++' = high goal fulfilment.

2. The history of nature-based forest management – In short

In Europe there have been attempts and local traditions to literally follow nature-near principle, to follow and steer the natural development in order to meet some more or less specific goals (Leibundgut, 1984; Schütz, 1990; Otto, 1993). However, the main trend in European forestry has followed the principles of organised forestry with a strong emphasis on clear-cutting, planting, thinning, homogenisation of structures, as well as rationalisation of working procedures. Organised forestry has had longstanding strong advantages in terms of overview, planning, standardisation, prediction and control.

While organised forestry has become the dominating concept in most parts of Europe, the more nature-inspired forestry approaches have been left to survive in the shade. Such concepts have not been given much attention, nor has much research been carried out to highlight possible advantages of this branch of silviculture. The ideal "to follow and assist nature in her development" has often been cited – but in reality rarely been followed in practice. Until recently, most attempts to apply nature-based forestry have been mainly exceptions from the rule. They have been carried out under special conditions and have been conducted by individuals mainly driven by conviction. A belief which has led to the assumption that nature-base forest management could turn out to be a more promising approach than traditional plantation forestry. People practising nature-based forestry have thus in the recent past often been given the image of being some kind of "religious freak" (Heyder, 1986).

Close-to-nature forestry is unquestionably focused around the idea of selection forest. The single tree and group selection system represent a clear contrast to the even-aged forests of organised forestry. Many foresters have tried to develop such uneven-aged mixed forests and have searched for appropriate methods to evaluate management successes, in order to compare them objectively to even-aged systems. The French forester Adolphe Gurnaud (1825-1898) once succeeded with the French *Méthode de contrôle*. His method based on regular inventories of forests parameters, especially diameter distribution and increment. Although not successful in implementation of his ideas, Henri Biolley (1858-1939) later succeeded in managing the community forest of Couvet with this "modern" selection system (Biolley, 1920).

Another important source of nature-based forestry started around the ideas of Karl Gayer (1822-1907), a silviculture professor in Munich. At that time, organised forestry with clear-cut systems and introduction of conifers had already expanded over large forest areas. Consequently, following this process, soil degradation, fungi and insect outbreaks, as well as frequent windbreaks had been observed in those areas. As a reaction, Gayer then developed his idea of mixed forests, which were about to be achieved merely through natural regeneration (Gayer, 1886), often in combination with the irregular shelterwood system. Using irregular regeneration over a longer time-span would thereby enable various different species to establish and thereby creating mixed forest structures.

His ideas were further developed in Switzerland. At that time Swiss forests suffered severely from torrents, landslides and windbreaks, as a result of spruce monocultures and clear-cut management systems. Arnold Engler (1858-1923) succeeded in gradual change of the Swiss forestry paradigm, which was untied from the regeneration scheme of organised forestry.

Today, variations of the Swiss irregular shelterwood systems are the most widely applied nature-based silvicultural systems all over Central and Eastern Europe. This is mainly thanks to the great flexibility of the system, which is based on the principles of adapting the felling temporally and spatially to the regeneration ecology of various tree species. Apart from the selection system, the irregular shelterwood system for nature-based forestry and the "free-style silvicultural technique" are significant as well; especially when it comes to managing degraded forests or transforming uniform and even-aged forests into mixed uneven-aged forests.

The third "wave" of nature-based forestry has developed arund 1920 in northern Germany when Alfred Möller published the book "Der Dauerwaldgedanke" (Möller, 1922). His paradigm of a continuation forest differs essentially from other nature-based concepts. Möller's approach is based on an organismic and holistic conception of forests and it follows stricter felling rules. His ideas had been shaped in forests where careful, continuous forest cover forest management had been applied for many years (Bärenthoren in eastern Germany). Möller carries out different inventories and publishes his results in favour of continuous forest cover management (German: Dauerwald), which, according to his conviction, offer improved forest sites, abundant regeneration as well as increased wood production.

Möller's forest approach was welcomed with great sympathy during the first years after his book had been published. 'The Dauerwald concept' was embraced with great enthusiasm all over Germany. When Möller died soon after publishing his book and his ideas proved unable to deliver the hoped success in the field, his approach became increasingly questioned and in the end even doubts about his scientific credibility ended this chapter of nature-based forestry in Germany during the 1930's.

With the foundation of a working group for close-to-nature forestry (Ger. Arbeitsgemeinschaft naturgemäße Waldwirtschaft - ANW), in 1950 yet another force steps onto the forest management scene. The ANW was rooted in the Dauerwald movement, and the groups ideas based on a set of principles rather than a management system. The group's members are mainly practising foresters and forest owners. Decisions on how to manage forests and strategies are empirical and often intuition based.

The call for for expanding the ANW-movement outside Germany resulted in the foundation of Pro Silva Europe in Slovenia in 1989. Pro Silva advocates close-to-nature forest management based on natural processes. Most European countries (at present 24) have joined and established national, independent Pro Silva sub-organisations. Their common ground on the national level is to develop and promote the principles of sustainability. These principles are considered to allow for the full development of the forests ecological and social roles, while a simultaneous economic production of high quality forest products can take place - all by mimicking natural processes. Members are forest owners, foresters, students and others who wish to practice and learn more about nature-near forestry.

2.1 The toolbox of classical and nature-based forestry

Basically classical plantation silviculture and close-to-nature approaches make use of the same toolbox in managing forests. However, the importance of single tools differs between the two concepts. Table 2 illustrates how different silviculture tools can be applied and combined under different management approaches. The plantation approach is displayed in two versions: traditional and modified (to achieve a higher degree of sustainability). Accordingly, the nature-based approach is displayed in a more economic, and a conservation focussed version.

Table 2 shows how management approaches determine what tools might be appropriate and most widely used. It further shows that although it is meaningful to differentiate between the various management approaches, it is neither possible, nor is it meaningful to draw a strict watershed line between those definition categories. Naturally transgression corridors occur. For each strategy however, it is possible to provide a set of relevant silviculture tools. Depending on management styles and aims within plantation, respectively nature-based management, the relative importance of the different tools can be adjusted. Each forest owner and each policy maker must critically choose his or her favourite tools for the situation and objectives which are being focussed upon.

Silviculture tool / Anticipated stand structure	Traditional plantation (production) approach	Modified (sustainable) plantation approach	Nature-based economic production approach	Nature-based nature conservation approach
Clear cutting at rotation age	+++++	++++	++	+
Single tree/group cutting	+	++	+++++	+++++
Planting or sowing	+++++	++++	++	+
Natural regeneration	+	++	++++	+++++
Use of soil preparation	+++++	++++	++	+
No soil preparation	+	++	++++	+++++
Use of pesticides	+++++	++	+	+
Ban of pesticides	+	++++	+++++	+++++
Use of exotic species	+++++	++++	+++	+
Use of native species	+	++	+++	+++++
Stand management	+++++	++++	++	+
Single tree management	+	++	++++	+++++
Harvest when ripe	+++++	++++	++++	+
Preserving old trees	+	++	++	+++++
Wood salvage	+++++	++++	++++	+
Leaving dead wood	+	++	++	+++++
Draining for production	+++++	++++	+++	+
Maintain wet habitats	+	++	+++	+++++
Monoculture	+++++	++++	++	+
Species mixtures	+	++	++++	+++++
Even-aged stands	+++++	++++	++	+
Uneven-aged stands	+	++	++++	+++++

Table 2. Examples of silviculture tools and anticipated stand structure and their relative importance in nature-based as well as classic (plantation) forest management:
+++++ greatly used, ++++ frequently used, +++ regularly used, ++ rarely used;
+ hardly ever used

Nature-based approaches in general refrain from larger clear cuts, but in specific cases - often in order to promote light demanding (pioneer) species - clear-cuts can be applied. Nature-based management relies heavily on natural regeneration but includes planting or direct seeding if natural regeneration is insufficient and/or if desired species are missing (enrichment planting). Nature-based approaches often make use of single tree selection based on target-diameter cutting, which should not be misinterpreted as "high grading" known from overexploitation of natural stands. Thus we here focus on a system to provide a sustained yield by making thinning among the various age classes in order to ensure their desired proportions and to maintain a suitable mixture of species. It should further be stressed that the heterogeneity of the stands is not just an end in itself, but rather a way of allocating species to various soil conditions and creating good forest floor conditions for natural regeneration.

The toolbox concept implies the refrain from any specific (religious) interpretation of what nature-based forest management is or should be - rather, the toolbox should be open to anyone finding the tools appropriate for any use he or she might wish for. The tools from

this nature-based toolbox can be used for nature-protection, for wood-production or to develop new types of urban forest (Larsen and Nielsen, 2011).
However, the main prerequisite for defining an approach "nature-near" or "close-to-nature" should be *that the practices are founded in, or inspired by, the structures and processes that occur in natural forests of a specific (reference) region.* This principle can be used to achieve all kinds of different management goals and objectives, including timber production, nature protection and social values.

3. Close-to-nature forest management in Denmark

The forests in Denmark amount to a total area of 570.800 ha, equivalent to some 13 percent of the total land cover. Originally, most of the land has been forested, but after centuries of uncontrolled logging and deforestation for agriculture, forest areas begun to decline drastically and consequently collapsed to a mere 2 to 3 percent around the 1820's. Since then the forest area is increasing due to large forestation efforts from 1860 and onwards and expected to reach around 20 % within this century.
Originally the Danish forest consisted mainly of deciduous trees - especially beech and oak. Over the past 200 years of forest management - including the large forest plantings in Central and West Jutland – the species distribution changed radically. Today, more than 50 % of forested areas are covered with non native conifers such as Norway and Sitka spruces, Douglas fir, as well as different Abies-species. Deciduous forests cover not more than 44 %, with beech and oak as the most common species, and ash, sycamore maple, Norway maple, birch, alder, wild cherry and lime as minor species.
As a general trend, forestry in Denmark has followed the overall European development when focussing on timber production in mostly plantation like structures. As a result, highly productive forests have been promoted, a process, which simultaneously created the matrix for increasingly intense conflicts with nature protection interests. First and foremost, the stability of the forests suffered through the development of even-aged monocultures. During the last 40 years, in 4 storms (1967, 1981, 1999, and 2005) a total of 15 million m^3 were blown down, whereas "only" 1 million m^3 fell down during the first 60 years of the past century. Hence, a major reason for the increasing impact of storms in Danish forests is the increasing use of storm sensitive conifer species.
In order to realize sustainable forestry at the management unit level (to achieve a proper balance between economic, ecological and social functions), a set of overall aims and operational guidelines has been developed in a stakeholder driven process during 2001. The National Forest Programme (Skov- og Naturstyrelsen, 2002) now consequently prescribes that Danish public forests should be managed in accordance with close-to-nature principles. The essence in these close-to-nature principles can be summarized as follows: Increase the stability and prepare the forests for an unknown future of changing climate, changing values and a variety of goals.
This close-to-nature approach is in particular focussed on:

1. Creating optimal conditions for natural regeneration by maintaining the permanent forest climate by refraining from clear-felling.
2. Stability improvement and risk diversification (resilience) through the creation of uneven aged mixed forest stands of site-adapted tree species.
3. Active stand improvement through frequent and weak thinning.

4. Protection of natural equilibriums among forest organisms, including pests, with the aim of promoting biodiversity and avoid the use of pesticides.

The close-to-nature forest management, combined with an increased use of climate robust deciduous and coniferous species and the reduction of climate change intolerant conifers (i.e. Norway spruce and Sitka spruce), are here identified as the overarching principles to secure sustainability, safeguard stability, and prevent the negative effects of climate change. Consequently, The Forest Act from 2004 supports the change from classical mono-species and even-aged management of stands into close-to-nature management characterised by more single tree and group management, incorporating and supporting natural regeneration and structural differentiation.

This decision to transform ''classical'' age-class forests (plantation forestry) towards nature-based forest stand structures implied no less than a paradigm shift in the management of state owned forests. Realizing that the complex character of these near-natural forest structures and dynamics require integrative and flexible management frameworks, as well as tools, a two step process was established: Firstly, the need for defining and describing long term goals for nature-near stand structure and dynamics was recognized and taken into the picture (where are we going?). Secondly, methods for transformation from plantation to nature-near structures were specified (how do we get there?).

3.1 The long-term goals – Creating Forest Development Types (FDT)

The concept "Forest Development Type" (FDT) was considered as an adequate framework for advancing and describing long-term goals for stand structures and dynamics in stands subjected to close-to-nature management (Larsen and Nielsen, 2007). An FDT describes the direction for forest development on a given locality (climate and soil conditions) in order to accomplish specific long-term aims of functionality (ecological-protective, economical-productive, and social-/cultural functions). It is based upon an analysis of the silvicultural possibilities on a given site in combination with the aspirations of future forest functions. It will serve as a guide for future silvicultural activities in order to "channel" the actual forest stand into the desired direction. Such a common understanding and agreement upon the desired development is crucial, since the conversion from age-class to nature-based stand structures is a continuous process.

In Denmark, a participatory process lead and described by Larsen and Nielsen (2007) resulted in the creation of 19 FDT's, which can be grouped into 9 broadleaved dominated, 6 conifer dominated, and an additional 4 "historic" types (Table 3). Whereas all "nature-based" FDT encompass a balance between productive, protective and recreational/social functions, the other four "historical" types mainly serve to protect recreational, natural and cultural functions. Especially the historical Forest Pasture (FDT No. 92) and Forest Meadow (FDT No. 93) can be actively used to create habitat diversity and experiential richness in forest landscapes.

Each FDT is described as follows (See also Figure 2, describing FDT No. 12 "Beech with ash and sycamore"):

- Name: The name encompasses the dominating and co-dominating species. The first digit in the FDT-number indicates the main species (1 = beech, 2 = oak, 3 = ash, 4 = birch, 5 = spruce, 6 = Douglas fir, 7 = true fir, 8 = pine, and 9 indicating a "historic" FDT). The second digit is numbered at random.
- Structure: A description of how the forest structure could appear when fully developed. This description is supplied with a profile diagram depicting a 120 m transect of the

anticipated forest structure at "maturity" (In Figure 1 profile diagrams of all 19 FDT´s are displayed and in Figure 4 the profile diagrams of four FDT´s are with different forest-edge types shown: No. 11-Beech, No. 21-Oak with ash and hornbeam, No. 71-Silver fir and beech, and No. 92-Forest pasture).

- Species distribution: The long-term distribution of species and their relative importance.
- Dynamics: The regeneration dynamics described in relation to the expected succession and spatial patterns (species, size).
- Functionality: Indication of the forest functionality (economic-production, ecologic-protection, and social/cultural functions).
- Occurrence: Suggested application in relation to climate and soil. For this purpose the country is divided into 4 sub-regions each with their typical climatic characteristics. Further, the application of the specific FDT in terms of soil conditions is stated in relation to nutrient and water supply.

Broadleaved dominated:		Conifer dominated:	
11	Beech	51	Spruce with beech and sycamore
12	Beech with ash and sycamore	52	Sitka spruce with pine and broadleaves
13	Beech with Douglas fir and larch	61	Douglas fir, Norway spruce and beech
14	Beech with spruce	71	Silver fir and beech
21	Oak with ash and hornbeam	81	Scots pine with birch and Norway spruce
22	Oak with lime and beech	82	Mountain pine
23	Oak with Scots pine and larch	**"Historic" forest types:**	
31	Ash with alder	91	Coppice forest
41	Birch with Scots pine and spruce	92	Forest pasture
		93	Forest meadow
		94	Unmanaged forest

Table 3. The 19 Danish Forest Development Types.

Matching forest development types to site

While different forest development types possess different site requirements it is possible to address and utilise potential variation in site conditions by matching FDT to site. This requires a thorough site survey, in which analyzing the basic growth conditions such as geology and soil types, nutrient and water supply, as well as specific site factors (such as compact layers and insufficient drainage) are taken into account. A hydrological status analysis on site is necessary, and it should include a survey of existing drainage systems, in combination with a plan of the historic landscape with former wet-lands, prior to any draining process. This hydrological analysis will provide an important tool and inspiration for delineating the landscape into ecological functional units. The site classification map works correspondingly as a frame for applying FDT to the site, thus facilitating the creation of forested landscape where site adapted forest and nature types reflect and emphasize variations within landscape.
Further, different FDTs possess different combinations of goal fulfilment - some are more production oriented, some more oriented towards nature/biodiversity protection, while others focus on enhancing landscape and recreational values. This variation in goal achievement can correspondingly be used to select FDTs - all according to specific functional requirements defined by the forest owner and - in case of public forest - by society/interest groups.

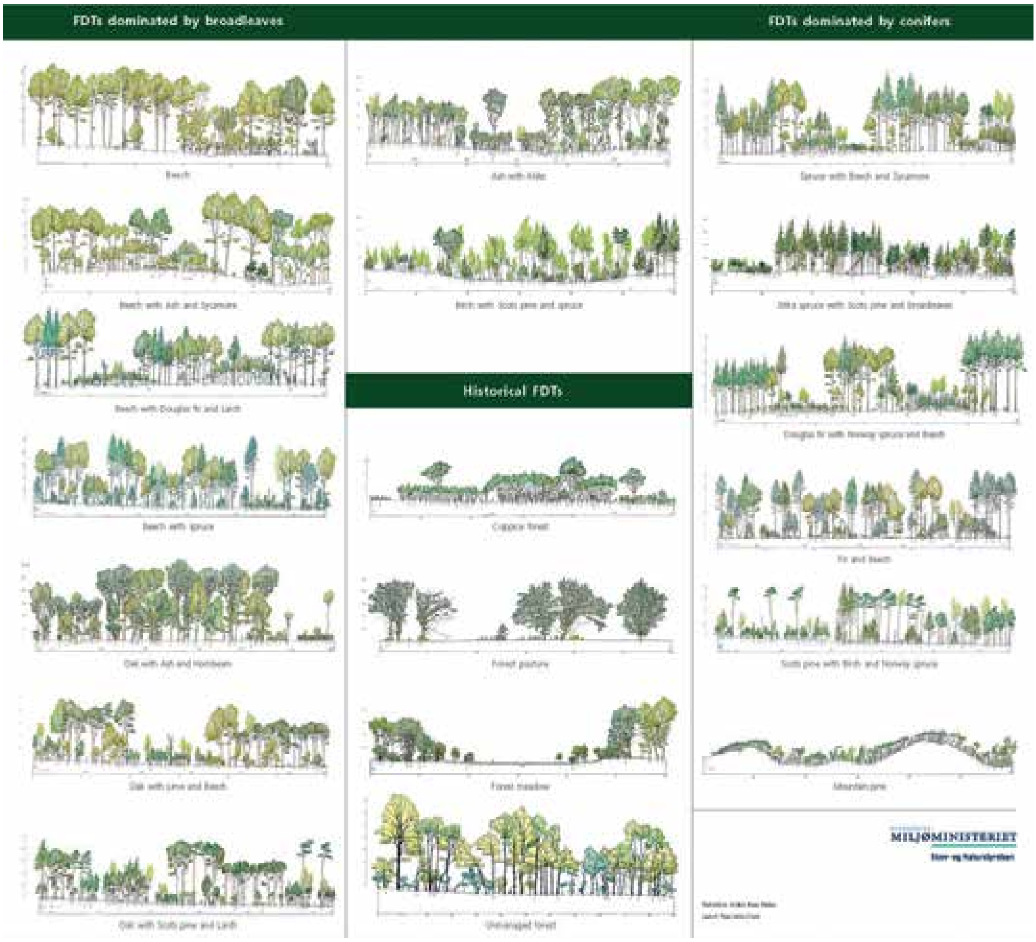


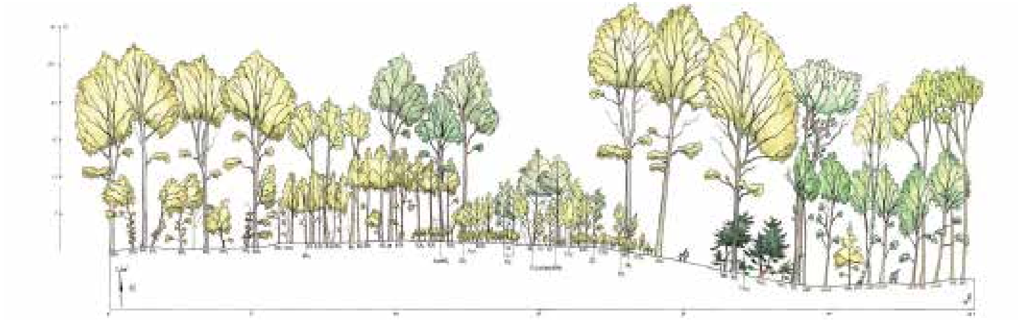
Fig. 1. Profile diagrams of the 19 Danish Forest Development Types.

At present the Nature Agency, responsible for the management of the Danish state forests, is laying out a grid system of forest development types on all public forests. This grid system will provide the local forest manager with information about the long-term goals he should aim at in each and every part of "his" forest. The managers job as local silviculturist will consequently be to observe the natural development and only then, after having conducted his observational research, to start making adjustments (cutting, planting, weeding, fencing, soil scarification etc.) in case the stand is due for short-term economic intervention (commercial thinning) and/or the actual development compromises the long term goal, as described in the attributed forest development type.

As mentioned above, the process of marking out FDTs on a management unit level is at present ongoing in Denmark. To illustrate this process, as well as the outcome, an example will be shown below. This example inspects the FDT-plan for the eastern part of Vestskoven as proposed by a group of students attending the international master course in Urban Woodland Design and Management (plan described in detail in Larsen & Nielsen, 2011).

Vestskoven was established in the 1960´s west of Copenhagen to create a large recreational forest that could separate and structure the intense and rapid urban sprawl, and provide for

Forest Development Type 12: Beech with ash and sycamore



Structure: Species rich, well structured forest with beech as dominating element mixed with ash and cherry and in south-eastern Denmark additionally with hornbeam and lime. The in-mixed species occur mainly in groups. The horizontal structures arise between groups of varying size and age. Where the light demanding species such as ash, sycamore and cherry dominate, vertical structures occur periodically with shade trees (beech, hornbeam, elm, and others) in sub-canopy strata.

Species: Beech. 40 – 60 %, ash and sycamore: 30 – 50 %, cherry, hornbeam, oak, lime, and others up to 20 %

Dynamics: Beech regenerates mainly in groups and smaller stands. Ash and sycamore as gap specialists regenerate in openings later followed by beech. Hornbeam belongs to the sub-canopy stratum and regenerates under shade, whereas the pioneer species (cherry and oak) only regenerate after lager openings and/or in relation to forest edges.

Functions: Productive: The forest development type has a high potential for production of hardwood in larger dimensions and of good quality.

Protective: In most parts of the country the beech dominated forest represents the potential natural vegetation; consequently, many indigenous species are connected to this forest development type. It has a great potential for conserving biodiversity connected to the NATURA 2000 habitat type 9139 and 9150.

Recreational: Through ist mixture of (indigenous) species in combination with pronounced variation in size the forest development type gives a multitude of recreational experiences and intimacy.

Occurrence: The forest development type belongs on protected sties in the eastern and northern parts of Denmark on rich, well drained soils with good water supply as illustrated below.

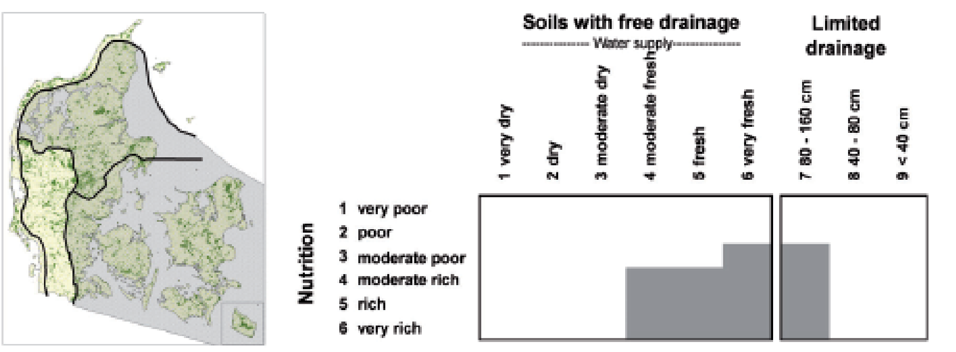


Fig. 2. Description and illustration of Forest development Type 12: Beech (*Fagus sylvatica*) with ash (*Fraxinus excelsior*) and sycamore (*Acer pseudoplatanus*).

important recreational qualities for the 300.000 new citizens in the western parts of Copenhagen. Fields were planted successively as they were purchased; little consideration was given to the overall composition and interlock zones between stands, or those parts dividing forested from open areas. The fields were planted according to traditional manuals with monoculture stands or simple species mixtures, using the species that were available at nurseries. The forest thus consists of small stands with abrupt species transitions and edges, all together lacking valuable interlock zones between the forested and more open areas. Today the area functions as a traditional Danish timber production forest with some large open spaces for recreation sprinkled onto it (Figure 3).

Fig. 3. Photo (from east towards west) of the eastern part of Vestskoven, showing the fragmented composition of uniform blocks of geometrically shaped stands and open spaces.

The above description demonstrates that Vestskoven incorporates most of the potentials, but even many problems, which urban woodlands inherited from the commercial forest management tradition with its uniform stand structures and its fragmented blocks of geometrically formed stands and open areas. The absence of smaller openings and glades, and the lack of valuable wetlands thus mould a fragmented, disconnected forest landscape.

Since Vestskoven is a public forest it will be managed according to close-to-nature principles and it is currently in the process of being charted into the FDT grid. Figure 4 presents a conversion/restoration plan where four Forest Development Types (FDT´s) have been laid out in respect of existing values in the young plantations and adjacent plains. The four selected FDT´s (FDT 11, Beech; FDT 71, Silver fir with beech and spruce; FDT 21, Oak with ash and hornbeam; FDT 92, Grazing forest), each with distinct experiential and ecological characteristics, unify the many small stands within larger units. The variety of size in open areas is increased by adding small, intimate glades in the forested parts. Some of the open areas have been linked to add further spatial variation and to increase coherence.

Parts of the forested, as well as the open areas have been converted into grazing forest through heavy thinning and some additional planting of trees. The borders between forested parts and open areas have been re-shaped organically by cutting out some of the existing stands, and instead giving room for edge species in those corridors. Thereby important interlock zones are being shaped between the denser forested and the more open areas, allowing for more diverse and complex edge structures. Ponds have been restored at emerging wetlands to render valuable landscape attractions, both in regard to landscape interpretation by visitors, as well as in regard to biodiversity in general. This landscape re-shaping takes place in the vicinity of small glades, at forested edges and in larger plains.

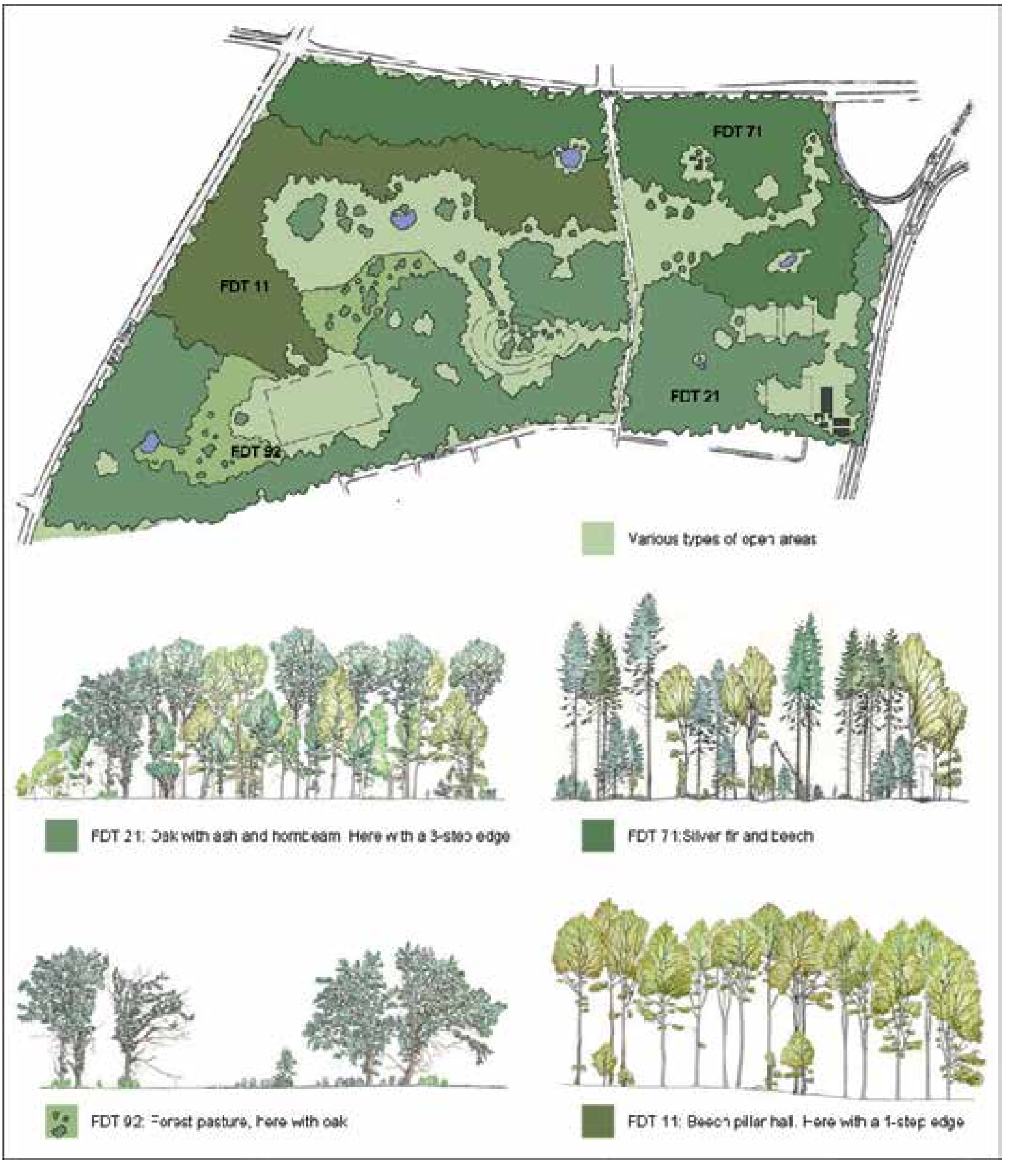


Fig. 4. Restoration plan for the eastern part of Vestskoven:

This plan was developed by a group of students attending the international master course in Urban Woodland Design and Management (Larsen & Nielsen, 2011). The chart, in combination with the profile diagrams of the four FDT´s, including examples of different edge-types, gives an instant impression of the anticipated urban forest landscape goals.

Furthermore, it provides an outline for appropriate developments in different parts of the forest. Such a developed and augmented plan, in combination with an FDT-map and profile diagrams of the different forest development types applied, can be used in multiple participatory planning processes.

3.2 Conversion principles and methods

Having defined the long-term goal at each part of the forest, the practitioners' principal task is to "guide" the forest from the current structure toward the targeted FDT. To help the local manager in this new endeavour, a number of conversion models haven been developed through a participatory process with local practitioners, forest workers and entrepreneurs. The primary purpose of this process is to come up with ideas as to how the conversion of a number of typical output models toward the desired forest type of development can take place. Since the conversion of uniform stands of spruce and beech are the main challenges in the transition to close-to-nature forest management in Denmark, the emphasis is on models for these species. Therefore, it is important to emphasise that these models are intended only to be used as inspiration, and they will always have to be adapted to any local situation, as well as to the concrete economic and technical possibilities. Especially the pace, at which the conversion is to be preformed, must be thoroughly analysed in regard to any economic aspects, paying special attention not to compromise expectation values for wood production in the transition phase. Therefore, in most cases, the full transformation to nature-near structures might take up to one or two tree generations.

Deciding on conversion strategy and tools there are two fundamentals, which must be kept in mind: Firstly, stand stability must be ensured and natural regeneration conditions must be improved. Thus creating various options and "freedom," timely to initiate rejuvenation (including bringing in new species), if required. Secondly, it is essential to initiate these elements at the ecologically and economically right moment in time. Thus, we speak of 1) a preparation phase, where the forest is stabilised and prepared for regeneration - mainly through selective thinning operations, and 2) a transformation phase, characterized by passive or active initiated regeneration, respectively by introduction of new species (and the procedure of ensuring their development). The preparation phase is usually associated with income (or at least cost neutrally implemented); whereas the transformation phase often entails costs (investments). Although, according to the principles of biological rationalization (a central economic aspect of close-to-nature-management), these costs could be kept on minimized levels by letting nature itself do as much of the "work" as possible.

If the forest development type prescribes species which are not present, or their genetic constitution (provenance) is not acceptable, additional seeding or planting (enrichment) in groups (typically, beech, ash, maple, birch, bird cherry, fir, larch, Douglas fir, etc.) is foreseen. These groups can later on contribute to a more widespread distribution of the species (done through seed dispersal). In order to allow rejuvenation of stable, but often frost-sensitive species (beech, firs including Douglas fir, etc.), a continued forest climate is regarded vital. Under such circumstances a stable forest canopy is paramount; especially in critically exposed, storm sensitive spruce monocultures. If a more complete conversion to new main tree species is aimed at already in the first generation, an extra widespread planting or seeding is envisaged, but often at higher cost. However, the close-to-nature approach is in general more inclined to exploit cheap regeneration methods, thereby accepting a longer conversion phase.

Generally, we distinguish between passive and active conversion strategies. The passive strategies are primarily based on existing vegetation, in order to convert as economically efficiently as possible. This implies mostly long conversion periods (up to several tree generations). The active approach is used where stability does not allow a slow (pending) conversion and/or there are other motives (ecological, aesthetical, and recreational) that advocate for a fast conversion.

Passive strategies

The purpose of the passive strategies is to implement as low-cost rejuvenation as possible, while maintaining optimum production in the upper canopy and the area as a whole.
Exhibit stands a high degree of stability; a passive strategy can be used that largely exploits the stand productive potential for transition to target diameter cutting without losing the possibility of a conversion. Transition phase can likewise extend over a long period of time, utilizing the system's own forces (natural regeneration), supplemented with scattered introduction of "new" species, if needed in the emerging gaps. Under such conditions, there are usually no major conflicts between the long-term objectives and the operating economy of the conversion phase. Gap size, and thus the potential light radiation, plays a crucial role in the choice of implanted species where they do not appear spontaneously. Thus, light demanding species such as larch, Douglas fir, oak, birch etc. require larger gaps (above 0.4 ha), while in the smaller gaps (0.1 - 0.2 ha), more shade tolerant species such as beech, maple and fir will be suitable.

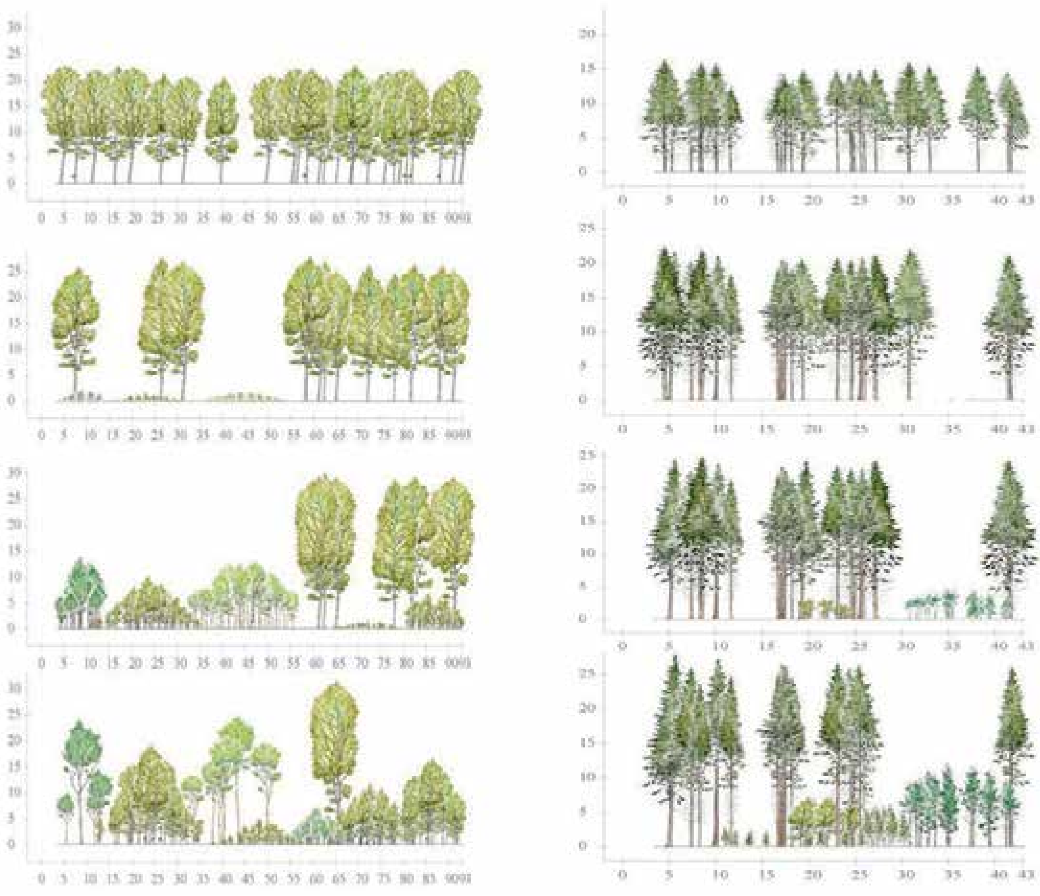
Fig. 5. Passive approach; Spontaneous regeneration of fir, spruce, birch, larch and Mountain ash in wind-throw gaps in a Norway spruce stand (group regeneration), Klosterheden Statsskovdistrikt. Photo: J.B. Larsen

Active strategies

The active conversion approach is used under conditions, where lack of stability does not allow a passive conversion. Active strategies are used in unstable stands primarily of spruce. In potentially unstable stands which have not yet reached a height that makes them storm exposed (below approx. 14 m), it is important to conduct an active thinning to promote stability and structural variation. This can happen partly through an early shelterwood formation or by liberating a number of future trees, thereby creating stable single trees (anchor trees). Important is that the thinning is conducted "from above" (removing dominating and co-dominating trees) thereby promoting variation in tree size (diameter, height) and a more heterogeneous stand structure. Group felling, in combination with early introduction of regeneration are also examples of active strategies. It is common to these approaches that a portion of the potential production in the stand will be sacrificed to safeguard the success of regeneration. In some cases the only economically realistic approach for regeneration/conversion of unstable spruce stands will be a clear cut; a measure, which also can be considered as an active strategy. In situations, when clear-cutting is the only way to regenerate the stand, frost hardy pioneer species such as Scots pine, oak, larch and birch will be introduced by planting/sowing to supplement, to improve the frequent natural regeneration of spruce and birch, thereby increasing future silvicultural options and thereby successively moving towards the planned long term goal – the FDT.

Fig. 6. Active approach; 9-year old beech planted under a canopy of Norway spruce (shelterwood regeneration). Klosterheden Statsskovdistrikt. Photo: J.B. Larsen.

The choice of conversion strategy depend on the starting point including the potential stability of the concrete stand, the objective defined by the FDT, and the time available for the conversion according to the economic perspective of net-present values of anticipated functions, together with the conversion costs. In total 10 different conversion and regeneration models have been developed for converting monocultures of beech, spruce and oak into nature-near structures. In Figure 7 such two models are displayed by means of



Left: Passive approach showing the conversion of a 54-year old even-aged beech stand to FDT 12 – Beech with ash and sycamore maple. The so-called "qualitative group cutting" is applied. Thinning is preformed by cutting trees according to their quality disregarding an even distribution of the remaining trees. This will create openings in the closed beech stand, where ash and maple is introduced. The regeneration is completed by natural regeneration slowly creating a group-wise structure of beech, ash and maple.
Right: Active approach showing the conversion of a 24-year old Norway spruce plantation to FDT 61 – Douglas fir, Norway spruce and beech. The thinning regime aims at creating variation in the overall thinning density in the area. It is done to open up for creation gaps, to be filled with Douglas fir and beech. The rest of the area is regenerated naturally with spruce and birch.

Fig. 7. Conversion models displayed with profile diagrams.

profile diagrams, depicting a possible development from a uniform plantation like structure towards the decided nature-near forest development type.

4. Conclusion

The management of forests "closer to nature" has increased significantly in recent decades, simultaneously accompanied by ever more reliable and refined models, promoting its efficient implementation. The basic idea is to reach a better balance between productive, protective and social functions. Other important goals are to increase economic competitiveness by cost reduction and increase robustness to climate change.

In Denmark, the Nature Agency started to manage all public forest according to close-to-nature principles in 2005. To facilitate the transition from classical even-aged plantation forestry to close-to-nature silviculture a total of 19 Forest Development Types (FDTs) and different conversion models have been developed in a participatory process with forest practitioners, scientists, forest workers, contractors and other stakeholders.

Now, almost 10 years after the political initiation, and 6 years after the state forest once started to be managed according to close-to-nature principles, the picture is multifaceted: The conversion process in the state forests is continuing with special focus on developing nature rich recreational forest landscapes, by means of the FDT planning scheme. A massive effort to restore natural hydrology is one of the most significant ingredients in the process; as well as the integration of permanent open spaces in the forest (forest meadows – FDT 93), the introduction of grazing animals (forest pasture – FDT 92), and the delineation of larger reserves (unmanaged forest - FDT 94). Furthermore, different methods and models for converting spruce plantations have been used. Still, it seems too early to draw any final conclusions in regard to his last aspect. The lack of funding for a scientific follow-up is a potentially jeopardising aspect.

Many forests belonging to municipalities have also changed management strategies fundamentally and they now apply the close-to-nature silviculture guidelines. Especially the FDT planning tool-box has proven highly effective to generate discussion platforms to define goals and ways of forest management among various stakeholders in urban forests.

The private forest sector is still rather reluctant in applying close-to-nature management. Some forest owners are doing it with great enthusiasm, while a majority still sticks to the classical age-class plantation system. However, the running debate about the pros and cons has had its effect on the size of clear-cuts and the use of natural regeneration.

We are learning by doing: Some of the pending issues are: How much reduction in professional input/contribution is possible without loosing the advantages of close-to-nature management? To what extend is it possible to educate private forest contractors to apply close-to-nature silviculture with their big machines? Is it possible to create the same high wood quality in un-even aged forest systems as in plantation like structures – and to what costs? How can the close-to-nature managed forest cope with the increased need for bio-energy production?

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Ecological and Environmental Role of Deadwood in Managed and Unmanaged Forests

Alessandro Paletto[1], Fabrizio Ferretti[2], Isabella De Meo[1], Paolo Cantiani[3] and Marco Focacci[4]

[1]Agricultural Research Council – Forest Monitoring and Planning Research Unit (CRA-MPF), Villazzano di Trento
[2]Agricultural Research Council – Apennine Forestry Research Unit (CRA-SFA), Isernia
[3]Agricultural Research Council – Research Centre for Forest Ecology and Silviculture (CRA-SEL), Arezzo
[4]Land / Forestry Resources Consultant, Sesto Fiorentino
Italy

1. Introduction

According to the Global Forest Resources Assessment 2005, forest deadwood encompasses all non-living woody biomass not contained in litter, either standing, lying on the ground, or in the soil (FAO, 2004). This definition considers the non-living biomass which remains in the forest regardless of the portion removed for production purposes (i.e. biomass-energy production). All different components of deadwood such as snags, standing dead trees (including high stumps), logs, lying dead trunks, fallen branches, fallen twigs and stumps are comprised in this account (Hagemann et al., 2009).

The role and importance of deadwood in forest ecosystem has been recognized by the international scientific community since the early 80's. The first scientific studies focused on the role of deadwood as a key factor for biodiversity conservation thanks to its ability in providing microhabitats for many species (Hunter, 1990; Raphael & White, 1984). Other issues were then discussed and analysed such as the protective role of coarse woody debris in stabilizing steep slopes and stream channels (Densmore et al., 2004), the contribution of deadwood to carbon, nitrogen and phosphorus cycles (Laiho & Prescott, 1999) and the influence on stand dynamics and regeneration of natural and semi-natural forests (Duvall & Grigal, 1999).

At the political level the recognition of its role was more recent, and raised in importance a decade after the scientific recognition. In particular, deadwood was included within the five carbon pool list (above-ground and below-ground biomass, litter, deadwood and soil) provided by Intergovernmental Panel on Climate Change (IPCC)-Good Practice Guidance for Land Use, Land Use Change and Forestry (IPCC-GPG) (2003). The change in C-stock in deadwood is required for reporting to the Kyoto Protocol (1997), Marrakesh Accords (7th Conference of the Parties, 2001) as well as to the United Nations Framework Convention on Climate Change (1992) (Tobin et al., 2007).

In Europe the importance of forest deadwood has been identified for the first time by the Ministerial Conference on the Protection of Forests in Europe (MCPFE) (2002) during the definitions of a set of Pan-European indicators for sustainable forest management. Deadwood is one of the indicators under the criterion "Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems" and can be usefully considered in order to measure the level of biodiversity (Indicator 4.5: volume of standing deadwood and of lying dead-wood on forest and other wooded land classified by forest type).

The amount of deadwood in forest depends on a set of variables (Lombardi et al., 2008): forest type, stage of development, kind and frequency of natural or anthropogenic disturbances, local soil, local climatic characteristics and type of management. Regarding the latter, the qualitative and quantitative presence of deadwood in forest ecosystems is influenced by both forest system (either coppice or high forest) and the intensity of management (Green & Peterken, 1997; Fridman & Walheim, 2000). In managed forests, potential deadwood volumes are reduced by the extraction of timber and biomass. (Andersson & Hytteborn, 1991; Christensen et al., 2005; Green & Peterken 1997; Kirby et al., 1998; Verkerk et al., 2011)

Similarly, also the qualitative features are altered in comparison with those of natural dynamics. Furthermore, in traditional management practices, the accumulation of deadwood may not be desirable because of the increasing risk of either insect pests (such as bark beetles) or forest fires. In biodiversity oriented forest management one of the main purposes is to reduce the difference in deadwood volume between managed and unmanaged forests, while the close-to-nature forest management aims at maintaining a certain level of deadwood (Müller-Using & Bartsch, 2009).

For these reasons, in order to support technicians in developing suitable forest management plans and selecting the best silvicultural option, qualitative and quantitative data on deadwood should be collected. The silvicultural treatment can play a fairly significant role in balancing necromass volumes. Ad hoc solutions and well-designed planning of different silvicultural actions may increase, wherever is thought to be important, the presence of deadwood in the system. However, this achievement should be obtained without affecting costs and related management components (cost of harvesting, pest and fire hazard). With these premises, the authors provide a method to quantify stumps, standing and lying deadwood in forest with the aim at supporting multifunctional planning and management practices.

1.1 Functional and structural role of forest deadwood

Deadwood is an essential multifunctional and structural component of forest ecosystems (Harmon et al., 1986), being a key factor in the nutrient cycle (N, P, Ca and Mg) (Holub et al., 2001), a fundamental element in the ecological, geomorphological and soil hydrological processes (Bragg & Kershner, 1999), a relevant forest carbon pool (Krankina & Harmon, 1995), a potential resource for biomass-energy production and an important habitat for many species (mammals, birds, amphibians, insects, fungi, moss and lichen communities) (Nordén et al., 2004).

From the ecological and environmental point of view, deadwood increases the structural and biological diversity of the ecosystem since many organisms are adapted to utilise this resource. In particular, two types of organisms, which depend on its presence in the forest ecosystem, can be distinguished (Wolynski, 2001):

- directly dependent organisms: use the deadwood as substrate for germination, power source and nesting site;
- indirectly dependent organisms: find occasionally shelter in the coarse woody debris or in standing dead trees at either first or advanced level of decomposition.

Among the organisms of the first group, saproxylic insects occupy the most important position, being a major part of biodiversity in forest ecosystem (Schlaghamersky, 2003). The saproxylic organisms, either those classified as obligatory or facultative, depend, at some stage of its life cycle, on deadwood of senescent trees or fallen timber.

Considering the relationship between deadwood and bird species, a particularly important role is played by the "habitat trees". Normally, "habitat trees" are large size individuals, with a diameter greater than 30 cm, which contain hollows used by forest fauna (Humphrey et al., 2004). The bird species that are hosted by dead trees can be primary excavators of cavities (i.e. wood peckers) or secondary cavity nesters (Hagan & Grove, 1999). The importance of deadwood as an indicator of biodiversity is provided by the diameter of the tree which is closely related, in turn, to the size of the nest holes. Thus, some bird species, such as *Parus palustris, Parus caeruleus*, *Passer montanus*, and *Sitta europaea* require small cavities (hole diameter less than 5 cm), whereas some other species such as *Strix aluco*, *Upupa epops* (Longo, 2003), *Dryocopus martius*, *Picoides leucotos* and *Picoides major* need larger cavities (hole diameter more than 5 cm).

Moreover, several mammal species use hollows, cavities, roots, fallen branches and deadwood such as bear, lynx, fox, martens, squirrels, bats and many small rodents (Radu, 2006). In particular, many Mustelids use the deadwood as a shelter: the stone marten (*Martes foina*), marten (*Martes martes*) and wolverine (*Gulo gulo*). The tree holes are also used by common dormouse (*Muscardinus avellanarius*) and fat dormouse (*Myoxus glis*) as nesting site (Paolucci, 2003).

Deadwood plays also a role in soil stabilization, since the lying logs on the soil surface control the movements of soil and litter across the ground surface (Harmon et al., 1986). With special regard to the protective function of forests (indirect protection), the fallen logs may retain soil and water movement either on slopes or through the ground (Kraigher et al., 2002). Similarly, the fallen tree trunks provide good protection against avalanches and rockfalls (direct protection - Berretti et al., 2007). Considering the latter, lying deadwood has a positive effect in the short-medium term, whereas the decomposition of the wood can bring back the movement of stones accumulated over time.

Moreover, deadwood can act as a temporary storage site for carbon (C), because of its slow carbon dioxide release ability, thereby showing a potential in moderating global warming (Keller et al., 2004). Deadwood, as a carbon pool, can account for a substantial fraction of stored carbon, but only few studies have provided quantitative features and the length of the turnover in comparison with other C storing components of the forest ecosystem (above-ground biomass, below-ground biomass, litter and soil organic C) (Kueppers et al., 2004). The few efforts on this topic show that standing and lying deadwood accounts for about 6% of total carbon stock in forest (Ravindranath & Ostwald, 2008), but, according to a set of qualitative and quantitative features, it does so with a certain variability.

1.2 Quantitative and qualitative features of forest deadwood

The importance of deadwood in forest ecosystems can be analysed by considering some qualitative and quantitative features such as: volume and its distribution by component and size, origin (species or botanical group), decay class and spatial distribution.

The main variable to be considered, in order to evaluate and analyze the ecological importance in forests, and its influence on other ecosystem components, is volume. Volume was measured during the forest inventory by applying two main procedures (Morelli et al., 2006): the line transect method (Line Intersect Sampling - LIS), which was applied in order to quantify directly the amount of deadwood on the ground (Van Wagner, 1968) and the measurement of the metric attributes (length and diameter) in ordinary sample plots, in order to calculate both standing and lying deadwood volumes (Harmon & Sexton, 1996).

As indicated in literature, deadwood volumes vary greatly in forest: unmanaged natural or semi-natural forests show the highest values with more than 100 m^3 ha^{-1} (Green & Peterken, 1997), while intensively managed forests manifested the lowest outputs with 5-30 m^3 ha^{-1} (Kirby et al., 1998). This variability is influenced by the classification of components sizes as well as the diametric threshold used in the inventory and the different measurement methods. These data are confirmed by the results of the National Forest Inventories (NFIs). As explained in Table 1, the different management traditions existing in Europe have a direct consequence in the accumulation of deadwood. The highest values are recorded in central European forests (Austria, Germany and Switzerland), whereas the lowest values are found in France and Finland. In Italy the volume of all deadwood components (stump, standing and lying deadwood) amounts to 8.8 m^3 ha^{-1} (INFC, 2009).

At local level, different situations can be found since potential deadwood volumes are reduced in managed forests either by the extraction of timber and biomass (Verkerk et al., 2011) or by sanitation cuttings. Also the qualitative features are altered in comparison with those of natural dynamics (Hodge & Peterken, 1998). In the traditional forest management the presence of deadwood is considered negatively for several reasons. Historically, deadwood has been removed in order to decrease firewood risk as well as to protect timber from insect and fungal attacks (Radu, 2006). Nowadays, the newest paradigms in forest management have recognized the ecological role of deadwood and developed strategies to both maintain or increase the amount of deadwood in forest ecosystem (i.e. by increasing volume of lying deadwood in order to favour population of invertebrates).

Country	Volume (m^3 ha^{-1})	Note
Austria	13.9	NFI 2000-2002 Threshold considered 20 cm
Belgium	9.1	Standing and lying deadwood of Wallonia region
Finland	5.6	NFI 1996-2003. Threshold considered 10 cm
France	2.2	NFI 2002. Threshold considered 7.0 cm
Germany	11.5	NFI 2001-2002. Threshold considered 20 cm
Italy	8.8	NFI 2005. Threshold considered 10 cm
Norway	6.8	Threshold 10 cm
Sweden	6.1	NFI 1993-2002. Threshold considered 10 cm
Switzerland	11.9	NFI 1993-199. Threshold considered 7.0 cm

Source: Brassel & Brändli, 1999; Fridman & Walheim, 2000; INFC, 2009; Mehrani-Mylany & Hauk, 2004; Pignatti et al., 2009; Vallauri et al., 2003.

Table 1. Volume of deadwood in the main European Forest Inventories

Deadwood can be subdivided into three main components (Næsset, 1999): (1) snags or standing dead trees, (2) logs or lying deadwood and (3) stumps. All of them occupy a

different ecological role in the forest ecosystem. Standing dead trees play an important role in increasing natural diversity and, in general, in the functioning of forest ecosystems, since a wide number of plants and animals has been strongly associated with their presence (Marage & Lemperiere, 2005). Lying deadwood provides important habitats for numerous insect species including flies, beetles and millipedes, while nurse logs facilitate the germination of conifers in mountain forests (Vallauri et al., 2003). Referring to the origin of deadwood, each piece can be classified on the basis of the species or by simply distinguishing between coniferous or deciduous. The tree species can be easily identified if the plant has recently died by observing the bark and the wood structure. When these parameters are no longer recognizable because of the advanced state of decomposition, a simple distinction conifer/broadleaved should be applied (Stokland et al., 2004).
Decomposition is the process through which the complex organic structure of biological material is reduced to its mineral form, and it is the result of the interactions between biotic and abiotic factors such as non-enzymatic chemical reactions, leaching, volatilisation, comminution and catabolism. Decay processes depend on species (hardwood or softwood species), site conditions (microclimate) and exposure; these characteristics can be quantified visually by using a decay class scale. The decay rate influences the dynamics of carbon release and sequestration, and it is measured with a decay class scale that takes into account species, microclimate and exposure. Moreover, the stage of decay is a very important parameter in order to analyse ecological dynamics and quantify carbon pools (Zell et al., 2009). Several ways to classify the level of decomposition can be found in literature. Normally the most widely accepted classification considers different (three, four or five) decay classes determined on the basis of the following variables (Montes et al., 2004): structure of bark, presence of small branches, softness of wood and other visible characteristics. The most common classification system is a 5-class system (Hunter, 1990) used in the American Forest Inventory (Waddell, 2002) and in the main European forest inventories (Paletto & Tosi, 2010; Sandström et al., 2007). The five decay classes used in the international standard are reported in Table 2.

Decay class	Bark condition	Small branches	Woody consistency	Other visual characteristics
1- Recently dead	Entire and attached	Present	Intact	Little rotten area under bark
2- Weakly decayed	Entire but not-attached	Partly present	Intact	Rotten areas < 3 cm
3- Medium decayed	Fragments of bark	Absent	Partly broken	Rotten area > 3 cm
4- Very decayed	Absent	Absent	Broken	Large rotten area
5- Almost decomposed	Absent	Absent	Dust	Very large rotten area, musk and lichens

Table 2. Decay classes of deadwood (five-class system)

Considering the size, lying deadwood is normally divided into two categories: coarse woody debris (CWD) which includes the logs with minimum diameter of 10 cm and fine woody debris (FWD) which refers to logs smaller than this threshold (Densmore et al., 2004). The same values are used to classify standing dead tree and stumps. Woldendorp et al. (2002) suggested to consider litter those small woody debris which have diameters below

2.5 cm, whereas other authors classify them as very fine woody debris (VFWD – Hegetschweiler et al., 2009). The distinction between these two (or three) categories of size is important when the habitat requirements of animal, fungi and plant species must be identified. Normally, FWD is relevant for diversity of wood-inhabiting fungi, especially ascomycetes in boreal forest whereas CWD favours many species of basidiomycetes. VFWD, in particular, may be associated to wood-inhabiting basidiomycetes, especially in managed forests where there is little availability of other substrates (Küffer & Senn-Irlet, 2005).

The spatial density, as a parameter, (Comiti et al., 2006) indicates how CWD and FWD are distributed on the ground. The spatial distribution is the result of human activities (i.e. cutting) or natural events (i.e. landslides). This variable can be qualitatively divided into three classes: (1) homogeneously concentrated, (2) concentrated in small groups, (3) scattered.

2. Materials and methods

The quantitative and qualitative features of snags, stumps and logs were estimated and analysed according to the forest types and forest systems (coppice and high forest) in three case studies. Hence, the authors examined the relationship between the presence of deadwood - species, size and decay class distribution - and the forest management practices.

The analysis of the influence of forest management on deadwood were investigated in a four-phases research:

- Classification of land uses/cover;
- Qualitative and quantitative description of forest formations;
- Dendrometric measures including the qualitative and quantitative information on forest deadwood;
- Analysis of the relationship between carbon storage and intensity of management.

The CORINE land cover (EC, 1993) European classification - level III - was adopted as a reference classification system for basic cartography. A specific classification was assembled for the forests that was based on the use of a homogeneous cultivation subcategory. This feature was ranked as an intermediate between the forest category and the forest type, and took into account both the forest system and the possible treatments of the wood. This classification was obtained according to the existing regional forest types and was coherent with higher superior reference systems (Italian National Forest Inventory-INFC, European Nature Information System - EUNIS, CORINE).

Regarding the qualitative and quantitative description of forest formations (woodlands and shrub lands), stratified samplings were conducted on the basis of homogeneous cultivation subcategory. The information was then entered in a Geographical Information System (GIS) built on the regional forest map.

The main dendrometric and management parameters were calculated in a sub-sample of woodlands using a circular area with a radius of 13 m measured onto a topographic map for a total surface of 531 m^2. The parameters measured were: number of trees, diameter at breast height (dbh), tree height of some sample trees, regeneration, deadwood, and qualitative attributes linked to the forest management and harvesting operations.

This method was tested on three study areas (forest districts) located in three different administrative regions of Southern Italy (Figure 1): (1) Arci-Grighine district in Sardinia region, (2) Alto Agri district in Basilicata region and (3) Matese district in Molise region.

The Arci-Grighine district (39°42′7′′ N; 8°42′4′′ E) is located in the Centre-East area of the Sardinia island. The district has a total surface of 55,183 ha and a population of 26,207 inhabitants (density of about 0.47 persons ha^{-1}) subdivided in 21 municipalities. The forests cover a surface area of 26,541 ha, comprising 48.1% of the Arci-Grighine territory. The forest categories, in order of prevalence, include: Mediterranean maquis (57.0%), Mediterranean Evergreen Oak forests (*Quercus ilex* L.) (14.7%), Cork Oak forests (*Quercus suber* L.) (9.9%), Eucalyptus forests (*Eucalyptus spp.*) (5.3%), Mediterranean pine forests (*Pinus spp.*) (3.1%), Downy oak forests (*Quercus pubescens* Willd.) (2.3%) and Monterey Pine (*Pinus radiata* Don) (1.7%)

The Alto Agri district (40°20′25′′ N; 15°53′52′′ E) is located in the Province of Potenza and characterized by a population of 33,739 people and a surface of 72,469 ha (density of about 0.47 persons ha^{-1}) divided into 12 municipalities. The forest areas cover 42,367 ha, comprising 58.4% of the Alto Agri territory. The forest categories, in order of prevalence, include: Downy oak forests (*Quercus pubescens* Willd.) (28.4%), followed by Turkey oak (*Quercus cerris* L.) forests (17.8%), shrub lands such as broom thicket, mixed thorny thicket and thermophile thicket with *Phillyrea sp.* and *Pistacia lentiscus*, (12.7%) and Beech (*Fagus sylvatica* L.) forests (9.6%).

The Matese district (41°29′12′′ N; 14°28′26′′ E) is located in the Province of Campobasso and characterized by a population of 21,022 people and a surface area of 36,539 ha (density of about 0.58 persons ha^{-1}) divided into 11 municipalities. The forest areas cover 15,712 ha, comprising 43.0% of the Matese territory. The forest categories, in order of prevalence, include: Turkey oak (*Quercus cerris* L.) forests (42.3%), followed by Beech (*Fagus sylvatica* L.) forests (30.5%), Hop hornbeam (*Ostrya carpinifolia* Scop.) forests (10.9%).

Fig. 1. Location of the case studies in Italy

The number of sub-plots were proportionally chosen according to the different forest surfaces: 218 sub-plots in Arci-Grighine, 235 sub-plots in Alto Agri and 117 sub-plots in Matese.
The quantitative presence of forest deadwood (volume) was investigated in each sub-sample plot taking into account four main integrative features: components, origin, decay class and size.
The volume of each log or snag included in the sub-sample was measured by applying a geometric system and, only for the snags, the stereometric equation of Italian National Forest Inventory 1985.
The forest operators registered lengths and diameters in three cross sections (minimum, maximum and medium) for lying dead wood while for the standing dead trees also the tree height and diameters at breast height (dbh) were considered.
Standing dead tree volume (V_s) was calculated from stand basal area (BA) whereas tree height was obtained from the hypsometric curve (h), by using the standard biometric equation (Cannell, 1984):

$$V_s = f \cdot BA \cdot h \tag{1}$$

which includes a standard stem form factor (*f*) of 0.5.
Lying deadwood volume (V_l) and stump volume (V_{st}) was calculated using the following formula:

$$V = \frac{\pi}{4} \cdot h \cdot \frac{D+d}{2} \tag{2}$$

Where:
h = height or length measured (m)
D = maximum diameter (m)
d = minimum diameter (m)
The total volume of deadwood in forest (V_d) was the sum of three components:

$$V_d = V_s + V_l + V_{st} \tag{3}$$

3. Results and discussion

3.1 Volume by components and decay class

The quantitative data on deadwood (snags, logs and stumps) in the three districts showed interesting differences linked to the different traditions of management (Table 3 & Figure 2). The maximum value of deadwood was found in the Matese district with 47.1 m^3 ha^{-1} being stumps (30.4 m^3 ha^{-1}) and snags (9.9 m^3 ha^{-1}) the major contributors. In the Arci-Grighine district the total volume was 21.2 m^3 ha^{-1}, almost exclusively concentred in snags (19.2 m^3 ha^{-1}). The Alto Agri district showed the lowest volumes of deadwood (8.8 m^3 ha^{-1}), the majority (61%) comprised of snags (5.4 m^3 ha^{-1}).
The variable number of deadwood pieces and volume provided an average of 0.54 m^3 $piece^{-1}$, with a minimum value in the Alto Agri district (0.23 m^3 $piece^{-1}$) and a maximum value in the Matese district (0.90 m^3 $piece^{-1}$).
The results obtained were also compared with the Italian NFI. The volumes in the three districts were higher than those provided by NFI (INFC, 2009). In addition, the quantitative

and qualitative differences were found to be comparable. A total of 1.7 m^3 ha^{-1} were recorded in Sardinia (0.8 m^3 ha^{-1} snag, 0.4 m^3 ha^{-1} stump and 0.5 m^3 ha^{-1} of log), 2.2 m^3 ha^{-1} in Basilicata (1.1 m^3 ha^{-1} snag, 0.5 m^3 ha^{-1} stump and 0.6 m^3 ha^{-1} log) and 4.3 m^3 ha^{-1} in Molise (2.7 m^3 ha^{-1} snag, 1.0 m^3 ha^{-1} stump and 0.6 m^3 ha^{-1} log). Other studies conducted in Italy show various values: 71.3 m^3 ha^{-1} were estimated in a site of Basilicata (Cozzo Ferriero) and in three sites of Molise of 95.6 m^3 ha^{-1} (Abeti Soprani), 17.4 m^3 ha^{-1} (Collemelluccio) and 26.5 m^3 ha^{-1} (Monte di Mezzo) (Lombardi et al., 2010). Moreover, in 21 study areas in North-West of Molise, Marchetti and Lombardi (2006) measured 15.1 m^3 ha^{-1}. These data show how the great variability in volumes is associated with specific site conditions and management practices.

District	Snag		Log		Stump		Total
	N ha^{-1}	Volume (m^3 ha^{-1})	N ha^{-1}	Volume (m^3 ha^{-1})	N ha^{-1}	Volume (m^3 ha^{-1})	Volume (m^3 ha^{-1})
Alto Agri	11.3	5.4	16.3	3.3	11.0	0.1	8.8
Arci-Grighine	17.1	19.2	7.3	1.2	17.9	0.9	21.3
Matese	16.2	9.9	20.4	6.8	15.8	30.4	47.1
Mean	*14.9*	*11.5*	*14.7*	*3.8*	*14.9*	*10.5*	*25.7*
St.dev.	*3.1*	*7.0*	*6.7*	*2.8*	*3.5*	*17.3*	*19.5*

Table 3. Volume and number of pieces for the different components of deadwood by district

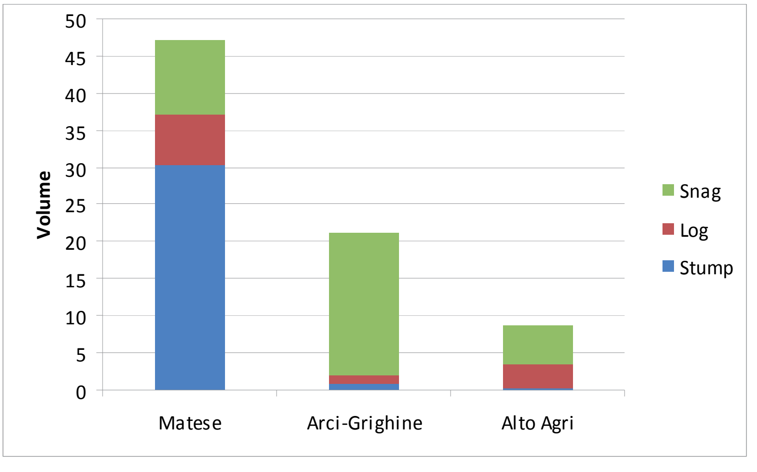

Fig. 2. Distribution of deadwood components volume (m^3 ha^{-1}) by district

The variation in decay class distribution provides an indication of the temporal variation in both tree mortality and tree felling and this variable can be used as an indicator of the history of a forest (Rouvinen et al., 2005). Generally, when fallen dead trees show all decay classes, the death of the plants have probably occurred evenly over a long time. *Vice versa,* when decay stages are concentrated in one or few classes, external events (naturally or human-induced) have concentrated the mortality in specific moments. Two different situations were observed in the case studies (Figure 3). In the Arci-Grighine district the volume of deadwood was concentrated in the first decay class (around 80% of total deadwood) and was composed almost exclusively of standing deadwood. Probably, the dead material might have been deliberately left in the forest for ecological or economic reasons after recent cuttings. Conversely, deadwood was regularly distributed along the five decay classes in the other two districts. The Matese scored higher values in the strongly decayed classes (fourth and fifth class) with around 67% of total volume, while the Alto Agri showed an opposite trend with 76% of the volume concentrated in the first two classes.

In general, the relatively scarce presence of highly decayed material in the Alto Agri and Arci-Grighine districts may be related to the effect of repeated clearing of the undercover vegetation which was carried out in order to prevent forest fires. Furthermore, heavy forest grazing in the Alto Agri caused the removal of dead material in the past.

The difference distribution of deadwood volume by decay classes in the three case studies were compared using the Kruskal-Wallis non-parametric test (Table 4). The results showed statistically significant differences only for stumps. In particular, the differences were related to the different distribution of deadwood in the Matese district compared with the other two districts.

	Observed K	Critical value	Degrees of freedom	p-value	α
Snag	0.081	5.991	2	0.961	0.05
Log	1.940	5.991	2	0.379	0.05
Stump	12.020	5.991	2	0.002	0.05

Table 4. Kruskal-Wallis non-parametric test: difference among the three districts concerning the three deadwood components

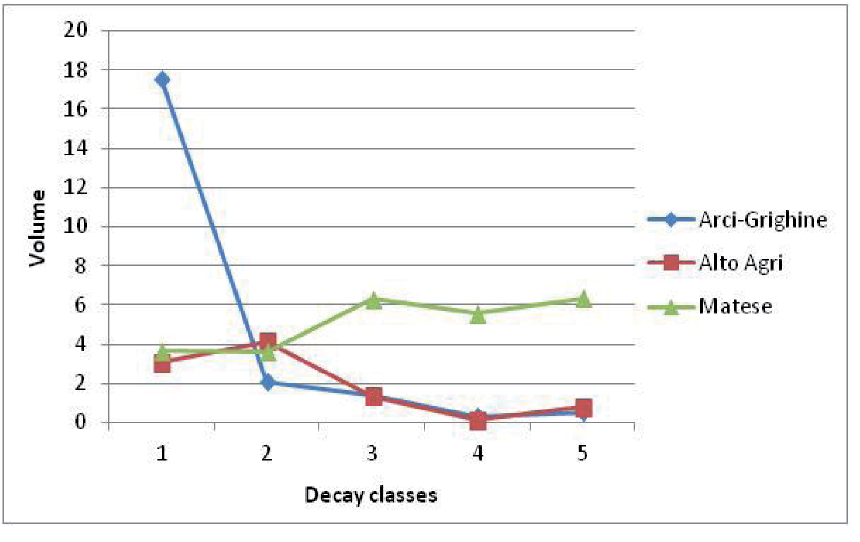

Fig. 3. Distribution of deadwood volume (m^3 ha^{-1}) by decay class in the three districts

3.2 Diametric and species distribution

The diametric distribution of deadwood (Figures 4, 5 & 6) provides important information on the presence of habitat trees and the differences among the three components of deadwood itself.

Regarding the tree habitat, a minimum number of 5-10 trees ha^{-1} is required for biodiversity conservation, especially for saproxylic organisms (Mason et al., 2005). The situation found in the three districts varied greatly since in the Arci-Grighine 31 dead trees ha^{-1} and 44 logs ha^{-1} with a minimum diameter of 30 cm were recorded, whereas both in Alto Agri and Matese habitat trees were only 1.7 per hectares.

In addition, the results on diametric distribution showed two different situations, being the Alto Agri more represented in small diameter classes and the Arci-Grighine and Matese in high diameter classes. In particular, in the Alto Agri district all three components were concentred in the first diametric class (40.4% of snags and 58.6% of logs). Instead, in the Arci-Grighine around 36% of snags and 41% of logs and stumps fell above the 30 cm diameter class. Similarly, in the Matese district 51% of logs and 69% of stumps were distributed in the highest diametrical class. Probably, the Alto Agri differed so significantly from the other two districts because almost 30% of its forests is constituted of young evergreen oak coppices, with high densities and a continuous mortality of thin dominated individuals.

The difference between the diametric distribution of the deadwood components in the three case studies were compared in pairs through the use of Kolmogorov-Smirnov non-parametric test. This test is based on the difference in the cumulative distributions of the two datasets. The results showed in all cases no statistically significant differences.

In order to test these differences by forest district, the Chi-square test (χ^2) was applied to the three deadwood components. The results obtained (Table 5) showed a statistical difference in sampling distribution of stumps and logs, while for the snag distribution the difference among the three districts was not significant.

	Observed chi-square value	Calculated chi-square value	Degrees of freedom	p-value	α
Snag	11.738	15.507	8	=0.163	0.05
Log	94.668	15.507	8	< 0.0001	0.05
Stump	51.274	15.507	8	< 0.0001	0.05

Table 5. Chi-square test (χ^2): difference among the three districts concerning the three deadwood components

Regarding the deadwood distribution per species in the Arci-Grighine district, a total of 60% of non-living biomass was concentrated in a single species (Monterey pine). Similarly, in the Matese district 64% and 9.7% of deadwood belonged to European beech and Turkey oak respectively. These results may be explained by the active firewood collection in oak forests and the substantial abandonment of beech forests. The species in the Alto Agri district were more evenly distributed: 34.4% of deadwood belonged to European black pine, 15.1% to chestnut and 12.2% to European beech.

In the Matese district, the beech deadwood consisted mainly of stumps (45.9%) and logs (48.7%), probably originated by old cuttings. In the Arci-Grighine instead, the presence of Monterey pine deadwood was almost exclusively composed by standing trees coming from abandoned old plantations.

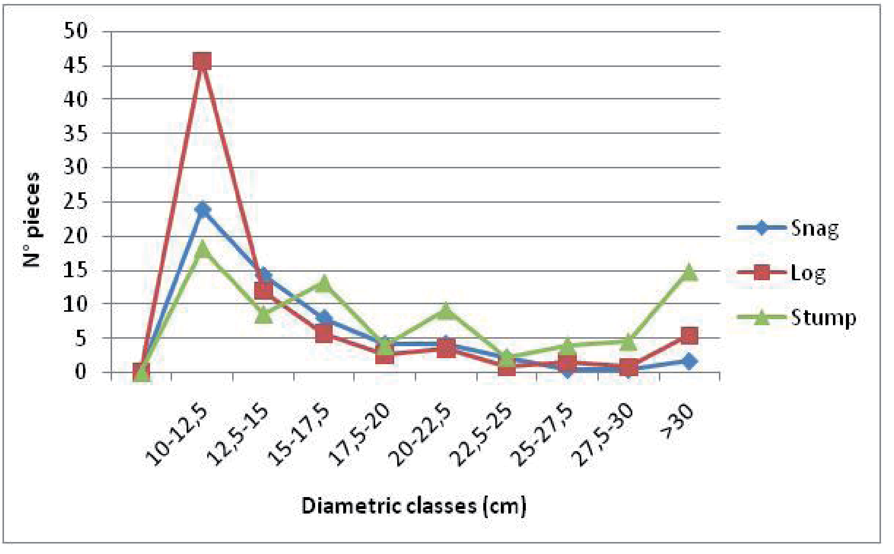

Fig. 4. Diametric distribution of deadwood in Alto Agri district

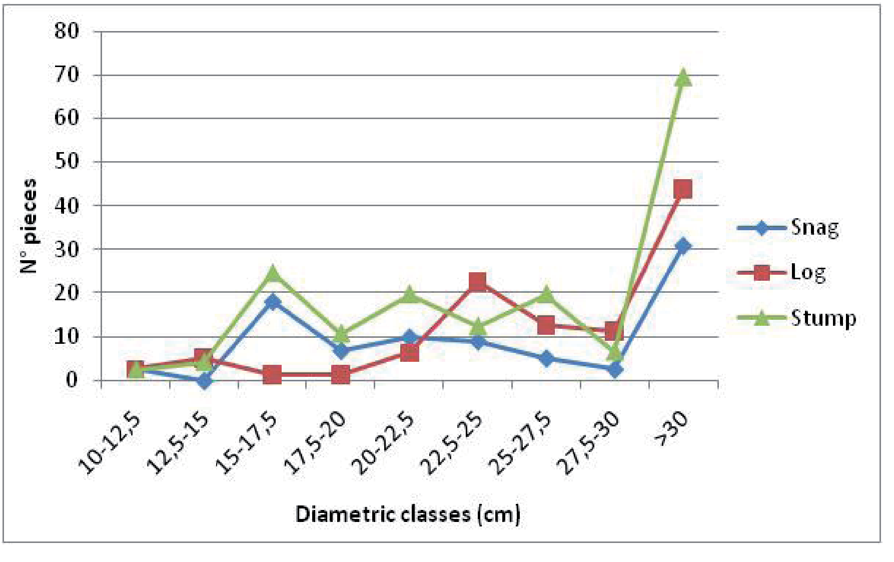

Fig. 5. Diametric distribution of deadwood in Arci-Grighine district

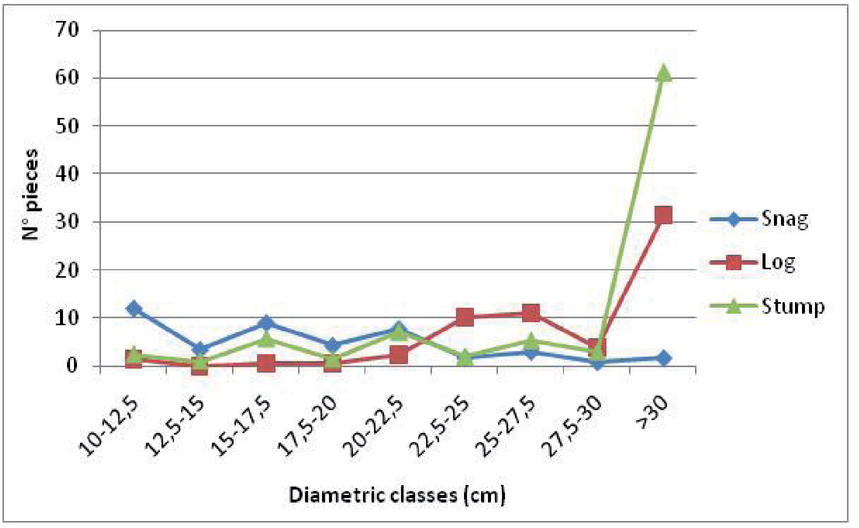

Fig. 6. Diametric distribution of deadwood in Matese district

3.3 Forest type and forest system

The type of forest system applied to the forest is a key factor to understand the impacts of forest management on deadwood and, consequently, on biodiversity conservation. The results showed that high forests had on the average higher volumes of deadwood for all three components in comparison with coppices (Table 6). In particular, the greatest differences were found for stumps in the Matese district (coppice: 7.4 m^3 ha^{-1}, high forest: 53.1 m^3 ha^{-1}) and for snags in the Arci-Grighine district (coppice: 3.8 m^3 ha^{-1}, high forest: 25.9 m^3 ha^{-1}). Only in the Matese district, snags scored higher values in coppices (11.1 m^3 ha^{-1}) rather than in high forests (8.7 m^3 ha^{-1}). This result was probably caused by a higher number of abandoned coppices in the area.

	Matese		Arci-Grighine		Alto Agri	
	Coppice	High forest	Coppice	High forest	Coppice	High forest
Stump	7.4	53.1	0.8	0.9	0.1	0.2
Log	4.4	9.1	0.2	1.6	1.6	5.9
Snag	11.1	8.7	3.8	25.9	3.9	7.6

Table 6. Volume (m^3 ha^{-1}) of deadwood components by forest system

Regarding the forest type, interesting differences were retrieved: in the Matese district *Fagus sylvatica* forests showed higher values than those of *Quercus cerris* forests, except for snags (Figure 7). In the Arci-Grighine district (Figure 8) very high volumes of snags were recorded in two forest type: *Pinus radiata* forests (93.5 m^3 ha^{-1}) and *Eucalyptus sp*. forests (54.1 m^3 ha^{-1}). In the Alto Agri district, instead, (Figure 9) Mediterranean pine forests (29.5 m^3 ha^{-1}) and, secondly, Mixed broadleaved forests (12.7 m^3 ha^{-1}) and *Castanea sativa* forests (12.8 m^3 ha^{-1}) showed the highest values of deadwood.

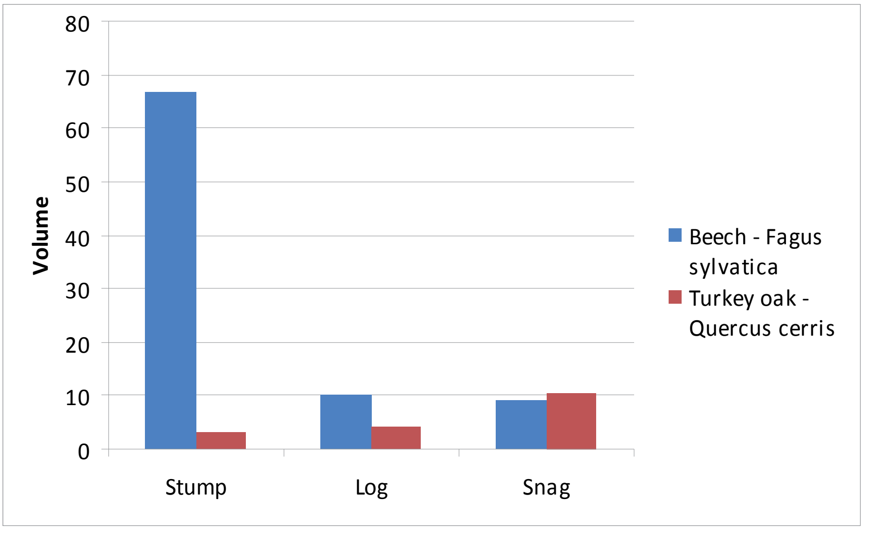

Fig. 7. Volume (m^3 ha^{-1}) distribution per forest type in the Matese district

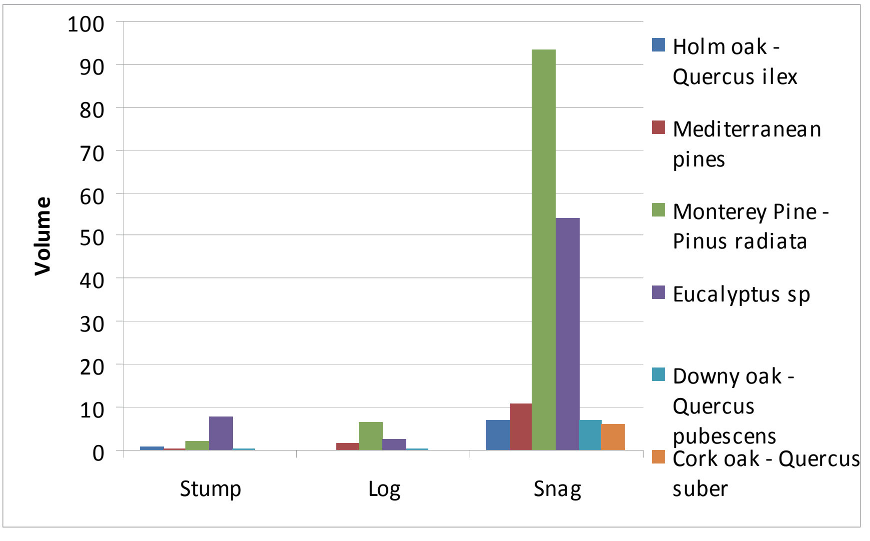

Fig. 8. Volume (m^3 ha^{-1}) distribution per forest type in the Arci-Grighine district

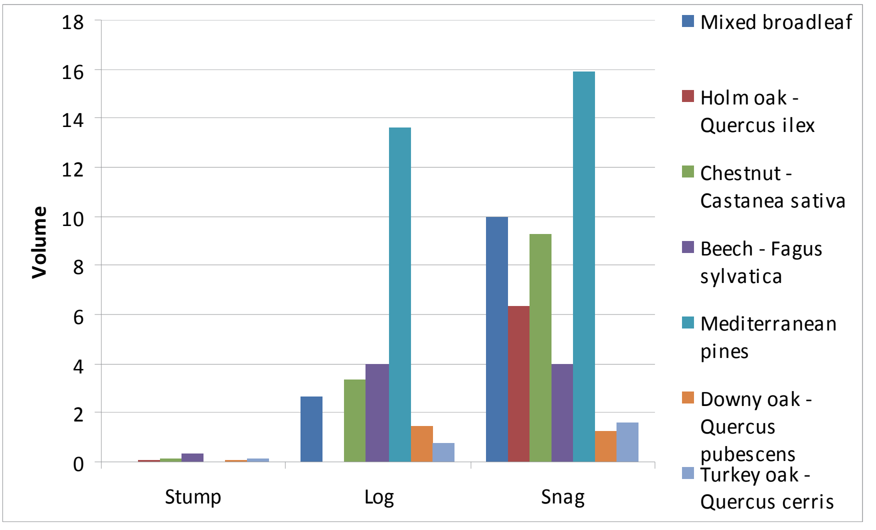

Fig. 9. Volume (m^3 ha^{-1}) distribution per forest type in the Alto Agri district

The higher number of stumps in the beech forests in comparison with those of the Turkey oak forests in the Matese district was a direct consequence of the different silvicultural treatments. As a matter of fact, these residues in the *Fagus sylvatica* forests, which belonged mainly to old standard trees, were originated by the conversion to high forests of abandoned coppices.
In addition, these types of forest had traditionally undergone to a less active management because of a minor economic interest on its main product (firewood) and a generally difficult accessibility. Similar considerations explain the high number of snags in the Arci Grighine *Pinus radiata* and *Eucaliptus spp.* forests. The abandonment, in the last decades, of these plantations increased the competitions among the individuals, thereby promoting high mortality rates and big sized deadwood material.

4. Conclusions

A method to collect quantitative and qualitative features of deadwood was a useful tool in order to define management strategies and silvicultural treatments aimed at optimizing the presence of deadwood in forest. In addition, its importance is remarked by the relevance of deadwood in carbon sequestration.
The expeditious method for the quantification of deadwood has an effective management relevance in supporting the choice of silvicultural treatments for the different forest type. The planners, with the analysis of the deadwood stock distinguished by type, specie and modality of active management, may acquire fundamental elements in order to define the sustainability of their technical proposals. Hence, appropriate interventions, aimed at valorising the specific functions of deadwood, can be defined case by case.
The different techniques may prescribe, wherever necessary, either the release of standing dead trees or other particular actions to increase deadwood. In coppices, for instance, a few standards may be left to indefinite ageing as well as some declining or dying individuals may be chosen as standards. In high forests instead, snags may be artificially increased by

girdling some plants. The number of stumps and logs may be improved by releasing dominated plants without economic value that will rapidly die and fall down. However, the increase of deadwood should be carefully planned along with all the remaining management considerations such as production, protection etc, giving particular attention on fire hazard and pest control.

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Section 6

Socioeconomic Functions

13

Multiple Services from Alpine Forests and Policies for Local Development*

Ilaria Goio[1], Geremia Gios[1], Rocco Scolozzi[2] and Alessandro Gretter[2]
[1]*Department of Economics, University of Trento*
[2]*Research and Innovation Centre, Fondazione Edmund Mach –Michele all'Adige (Trento)*
Italy

1. Introduction

The starting point of the analysis here presented is the concept of ecosystem services, which could help us appreciate natural systems as vital assets, recognizing the central roles that they play in supporting human well-being, either at the local or global level. In fact, ecosystem services provide benefits, in terms of goods and services, both to people living in the mountains and to people living outside them. At the moment, these services are seriously threatened, and "their global degradation is increasingly jeopardizing development goals"(OECD, 2008). As a consequence, it is necessary to reverse this trend while, at the same time, meeting the increasing demands of and interests in such services.[1]
The focus of our study are the alpine forest ecosystems, which represent a fundamental resource for people living in mountain areas and for human society, in general.[2] In fact, it is commonly known that forests nowadays fulfil several other functions, in addition to what has been perceived as their main function (the productive one). These functions include the protective function, the landscape and recreational function and the ecological function. This functionality means that forests not only produce goods but also various social and environmental services,[3] contributing, in many different ways, to the welfare of humans. This capacity is well summarized in the concept of "multi-functionality". It is clear that ''better understanding of the full range of goods and services supplied by forests is essential for optimal utilization of forests, and it may provide an economic rationale for sustainable forestry'' (Lange, 2004).

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* This paper is the result of its authors' common reflections. However, single sections have been written, as follows: Ilaria Goio wrote 1, 3, 4.1 and 6.1, Geremia Gios wrote 4, 5 and 6; Rocco Scolozzi wrote 2; and Alessandro Gretter wrote 6.2.

[1] "One of the most important problems that our society currently faces is how to strike a suitable balance between the conversion of natural capital to economic production and its conservation to provide ecosystem services" (Farley & Costanza, 2010).

[2] According to the Millennium Ecosystem Assessment (MEA, 2005) the "environmental conservation and sustainable land use in the world's mountains are not only a necessary condition for sustainable local livelihoods, but also for well-being of nearly half the world's population who live downstream and depend on mountain resources".

[3] Historically, the nature and value of these services have largely been ignored until their disruption or loss has highlighted their importance (Daily et al. 1997).

Within this framework, the main objective of this work is to define the management policies that allow efficient and effective use of goods and services produced by forests.
Clearly, these policies will differ in relation to the kinds of goods or services considered and also in relation to the specific socio-economic and environmental context of a given area. In particular, in our analysis, we will make reference to the landscape and recreational function and to its economic assessment, as partly learned by our working experience in the Alpine context of the Autonomous Province of Trento (Italy).
We would like to suggest to the public and local policy makers of the southern part of the Alps, some general economic policy instruments. The objective of these policy instruments is twofold: on the one hand, they permit policy makers the use ("with a sufficient flexibility in order to operate within constantly changing circumstances" [OECD, 1999]) of the above-mentioned goods and services. On the other hand, these policy instruments provide a useful support for orienting their action towards a territorial policy, which is able to give, from the perspective of sustainable development,[4] equal justice to the economic, social, and environmental components of forests.
Within the process just outlined, a key role is played by the local and non-local stakeholders. That is, some stakeholders are the actors who provide environmental benefits, and therefore, have to be remunerated. Other stakeholders should pay for taking advantage of the environmental benefits. For this reason it is necessary to understand how the cited actors perceive the factors connected with sustainability, facilitating and promoting an enriching exchange of views, knowledge and initiatives. To provide a complete and reliable overview, these point of view exchanges should involve both public and private actors, creating new synergies and new partnerships in the area.
This chapter is structured as follows. The next section explains the principal characteristics of ecosystem services and section 3 provides some considerations about the multi-functionality of forests. In section 4, specific reflections on forest joint-productions are presented, including brief considerations of market factors and payment for ecosystem services. Section 5 focuses on the particular case of landscape and recreational services and section 6 illustrates some policy implications for public decision makers in the alpine areas, with particular reference to the need for a participative approach, and offers evidence from Alpine examples.

2. Ecosystem services from mountain areas

Ecosystems are complex systems that provide humanity with vital services through interacting ecological processes. With regard to mountain areas, forest ecosystems provide wood products and a wide range of non-wood products and services, e.g., regulation of the climate and water supply, purification of the air and drinking water, protection against soil

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[4] The Brundtland Commission's report, published in 1987, defined sustainable development as "development which meets the needs of current generations without compromising the ability of future generations to meet their own needs" (Brundtland, 1987). Recently, the Research Institute for Humanity and Nature (Kyoto, Japan), proposed a reinterpretation of the sustainable development concept, referring to the idea of futurability. "Sustainability is a static and conservative concept that focuses on the continuation of the present-day anthroposphere (i.e., *sustainable parasitism*), although dynamic co-evolution between human and nature could be an alternative definition" (Newman, 2005). "In contrast, futurability is a more dynamic and ambitious concept that seeks truly sustainable and futurable human–environment interactions, namely *futurable mutualism*" (Handoh & Hidaka, 2010).

erosion and the support of soil fertility. Forest ecosystems also play an important role in the aesthetic and recreational values of landscapes, supporting increasing worldwide tourism.

Studies conducted since the 1980s indicate that forest values may be much higher than timber values per hectare (Peters et al., 1989). As a consequence, there has been an increasing realization that many other products and services generated by forests are essential to the well being of local communities and are required by society at large. In particular, the FAO (2005) defined non-wood forest products as products and goods "that are tangible, of biological origin other than wood, derived from forests, other wooded land and trees outside forests." These non-wood forest products include mushrooms, fruit, leaves, plants and animals collected or grown in forests, and they are used as food, fodder, medicine and raw materials for handicrafts. They have significance as cultural objects and as a source of income. This definition of non-wood forest products neglects intangible forest services (e.g., ecotourism, bio prospecting) and forest benefits (e.g., soil conservation, watershed protection and maintenance of biodiversity), which are clearly more difficult to assess and quantify than goods. Therefore, a new open system of terms for forest-dependent resources was proposed (Mantau et al., 2001, 2007): "Forest Goods and Services (FOGS), defined as resources of biological origin, associated with forests, other wooded land and trees outside forests".

Specific typologies were proposed to describe the forest transactions (uses) of interest to facilitate analyses or marketing. They consider three basic levels: 1. resource, 2. product and 3. user.[5]

Each of these levels may be internally classified into many hierarchical levels. For example, the "resource" plant may provide a "product" such as erosion control for the "user" state, but may also offer a different "product," such as fuel wood to the "user" local community. In effect, each resource may be structured into several products, and these products, in turn, are handled and consumed by many different user groups. A systematic taxonomy definition of goods and services (Mantau et al., 2001) may help in the examination and description of the value chains that are increasingly being developed as a basis for interventions to promote successful commercialization of FOGS.

Besides the FOGS, as defined above, the concept of ecosystem services better recognizes potential values for ecological/ecosystem processes *per se*. The MEA (2005) breaks ecosystem services into four different classes:

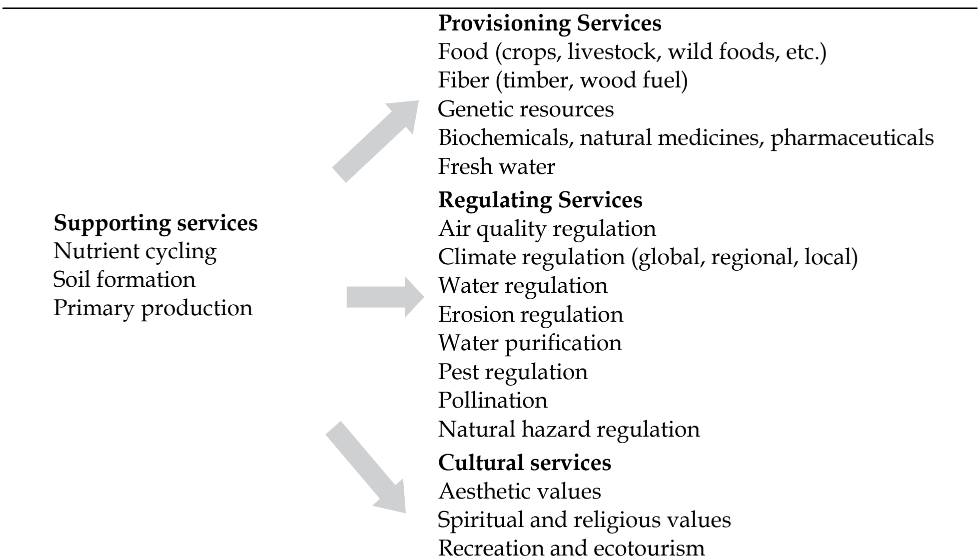
- *Provisioning services,* which are the products obtained from ecosystems, including food, fiber, fuel, genetic resources, ornamental resources, freshwater, biochemical, natural medicines, and pharmaceuticals.

[5] In more depth:

- Resource: in the context of the forest, anything of biological origin that is of use to humans and the basis for any output. For instance, resources for goods are energy, carbon, land, water, materials, plants, foodstuff, fibre, medicine, extractives and live plants or animals.
- Product: anything that can be offered to a market that might satisfy a want or need. A product can be a simple marketable good (e.g., fuel wood) or service (e.g., recreation) or combination of both (i.e., composite products or commodities, such as Christmas tree markets and guided mushroom-picking walks).
- User: any group of people that benefits from a product. This category includes collectors, processors, middlemen, retailers and the end-user or client. It therefore describes the market or value-chain for a given product.

- *Regulating services,* which are the benefits obtained from the regulation of ecosystem processes, including air quality regulation, climate regulation, water regulation, erosion regulation, water purification and waste treatment, disease regulation, pest regulation, pollination, and natural hazard regulation.
- *Cultural services,* which are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences, including cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation, and ecotourism.
- *Supporting services,* which are necessary for the production of all other ecosystem services.

Figure 1 summarizes well the different classes with reference to the mountain ecosystems. Mountains and their ecosystems provide all services from each of the four main MEA categories, as widely documented in the RUBICODE project.[6]



Source: modified from Patterson, 2009

Fig. 1. Broad categories of mountain ecosystem services

3. On forest multi-functionality

As described in the previous sections, it is commonly known that forests are defined as multi-functional[7] assets, providing, at the same time, different goods, connected with the productive

[6] This project, aiming at rationalizing biodiversity conservation in dynamic ecosystems, focuses on assessing the ecological resilience of those components of biological diversity essential for maintaining ecosystem services. It provides this focus in order to suggest priorities for biodiversity conservation policy on the basis of dynamic ecosystems and the services they provide (http://www.rubicode.net).

[7] According to the OECD (2001), the term «multi-functionality» "refers to the fact that an economic activity may have multiple outputs and, by virtue of this, may contribute to several societal objectives at

function (such as timber and non-timber products), and services connected with the ecological functions (such as soil conservation and protection, watercourse protection, hunting, fishing, protection of biodiversity, and the carbon cycle). As presented in the figure 2, these outputs (goods and services) can be classified differently with respect to the parameters of rivalry and excludability (Fisher et al., 2009; Gios & Clauser, 2009; Patterson & Coelho, 2009) and, thus, with reference to forests. These outputs could be purely private (excludable and rival – timber and non timber products) or purely public (non-excludable and non-rival[8] hydro-geological services). Moreover, there is a spectrum of forest goods ranging between purely private and purely public goods. Some of these "intermediate goods" are qualified as club goods (excludable and non-rival – landscape-recreational function) or as open access resources (non-excludable and rival) or "common-pool resources" (Hardin, 1968).). This latter category characterizes many (not just natural) resources. Their need for management has led to the establishment of various institutions in the Alps since the 11th century. Notably, the Autonomous Province of Trento treats almost 60% of overall surface (over 3,000 Km^2) land and goods as collective property and manages them as common-pool resources.

Usually, the productive function is defined as the «market function» while the others categories are referred to as «non market functions». In the first case, the forest generates some inputs for the productive processes that can be exchanged in the market and subsequently have a monetary value. Conversely, in the second case, the forest provides public goods (such as carbon sinks) and mixed goods (such as landscape and recreational values) that cannot be exchanged in the market, and therefore, cannot be priced. Moreover, as many studies have demonstrated and as we have already mentioned, "forests have a higher value than that solely connected to production aspects"(Goio et al., 2008).

We should consider that central to the debate on multi-functionality is the degree of the conjunction of the production of secondary goods compared to that of primary goods and the inevitability of this conjunction. Since the 1950s, some authors (Carlson, 1956; Marshall, 1959) have tried to define «joint-production» as involving things that cannot be produced separately, and are joined by a common origin. More recently, according to Shumway et al. (1984), "the joint production encompasses all production situations in which two or more outputs or products are interdependent". These inter-linkages could arise for three different reasons:

1. "there are technical interdependencies in the production process;[9]
2. outputs are produced from a non-allocable input;
3. outputs compete for an (allocable) input that is fixed at the firm level"[10] (OECD, 2001).

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once. Multi-functionality is, thus, an activity-oriented concept that refers to specific properties of the production process and its multiple outputs."

[8] To briefly clarify, the term «non-rival» means that a unit of the good can be consumed by one individual without diminishing the consumption opportunities available to others, from the same unit. In contrast the term «non-excludable» refers to the situation in which it is physically or institutionally (i.e. through laws) impossible, or very costly, to exclude individuals from consuming a good.

[9] Instead of $Y_1 = f_1 (L_1, K_1, T_1)$ and $Y_2 = f_2 (L_2, K_2, T_2)$, it is $Y_1 = f_1 (Y_2, L_1, K_1, T_1)$ and $Y_2 = f_2 (Y1, L2, K_2, T_2)$. Where:

Y = production
L = labour
K = capital
T = land

This means that, in the case of joint production, the production Y_1 is function not only of the usual production factors (L, K, T) but also of the production Y_2 and vice versa.

In the case of technical complementarities, (1) the products have to be produced together, or, in the other cases (2 and 3), outputs can be produced separately. However joint-production is cheaper because of the presence of economies of scope.

	Excludable	EXCLUDABILITY	**Non excludable**
Rival	*PRIVATE GOODS*		*COMMON POOL RESOURCES (OPEN ACCESS)*
	timber and non-timber products		local fishing and hunting
	patented processes from genetic resources		access to genetic materials
RIVALRY	eco-tourism	Hunting and fishing licensing	carbon sequestration
	natural parks with entrance fees	water quality trough ecosystem protection	
	flood control trough ecosystem protection		existence of species and ecosystems
Non rival	*CLUB GOODS*		*PURE PUBLIC GOODS*
	Local	LOCALITY	**Global**

Adapted from Landell-Mills & Porras (2002); OECD (2001)

Fig. 2. Different utility flows provided by forests

A second critical aspect is that the time horizon is different depending on the output that is evaluated. The emphasis is usually on the market failures resulting from the difficult assignment of an adequate property rights system.

Within this framework, the multi-functionality aspects that assume greater significance, may be identified as the following: the "type and strength of the link between forest production and secondary products; synergies and trade-offs between the various forest products; specificity of the forest in the provision of services and products not directly commercial; and the fact that the market is unable to assign a price to many secondary products, thereby requiring public intervention" (Henke, 2004). In many cases, as stressed by Janse & Ottitsch (2005), "synergies and the integration of these various components/products is not always without conflict".

[10] $Y_1 = f_1 (L_1, K_1, T_1)$ and $Y_2 = f_2 (L_2, K_2, T_2)$ but $L < L_1 + L_2$, $K < K_1 + K_2$, $T < T_1 + T_2$. In this case Y_1 and Y_2 can be produced separately but the costs connected with the production factors are higher than those of the joint production.

Evidently, inappropriate means of development, such as excessive intensification, mechanization, over-exploitation of resources, environmental pollution and urbanization are only some of the factors that could increasingly threaten the multi-functionality of forest ecosystems. Hence, this ecosystem can continue to provide their goods and services, in a rapidly changing world, only if multi-functionality is taken into account in their management. As a consequence, as we discuss in detail in the following sections, with the objective of properly defining the management options and opportunities, it is important to characterize, precisely, the different utility flows performed by forests and to evaluate them.[11] It is clear that the choice of the policy tools that have to be adopted will be: "a) different, depending on whether the goods are private, public or mixed and on the kind of joint production carried out; b) strictly connected with the specific socio-economic and environmental context of a given area; and c) flexible in order to operate within constantly changing circumstances" (OECD, 1999).

4. Some specific reflections on forest join-productions

In the case of forests, we are dealing with the two aspects of joint-production examined in the previous section. In particular, there is technical complementarities with reference to timber production and carbon fixation, and economies of scope with reference to the production of timber and the landscape. As a consequence, the non-commodity outputs are joint products with timber production. This circumstance means that joint products, that clearly create benefits for people living inside and outside the local areas, have different characteristics. In other words while timber is a market product that is "paid", the others are non-market products and are, therefore, "unpaid". It is important to point out the strong impact that the production of non-market outputs, has on the structure of the private costs related to market forest production. In fact, if the increase in private costs exceeds certain levels, this increase may affect the sustainability of the system. In such cases, the most effective solution is to pursue economies of scope rather than scale, because the costs to produce two or more outputs together are less than those for obtaining the same outputs through different production processes.

With the objective of maximizing the environmental externalities associated with forests, the production process should, consequently, be organized in a precise and defined way. The choice of many alpine areas has been and is so now, that of «natural forest management». The problem is that this kind of management causes an increase in the use costs, due to: a) higher cutting and logging costs, related to constraints on the maximum cutting area and to the need to adopt more environmentally friendly techniques.; b) constraints on the characteristics of forest's roads (reduced width, practicable by less efficient equipment); c) the acquisition of heterogeneous material (by species, diameter, features) imposes higher costs for the selection and start-up to the sale.

There are no specific researches able to quantify, in the alpine areas, the additional cost related to the natural forest management. However, some experts (Pollini et al., 1998) estimated that the increase is about 20-30 % of the forest utilisation. In addition, the natural forest management determines also a lower level in the production, and as a consequence, less available wood mass for the next links in the productive chain.

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[11] The assessment and valuation of ecosystem services, since the seminal papers (Costanza et al., 1997), has "recently focused on an extensive research, with the number of publications increasing almost exponentially" (Fisher et al., 2009).

In particular cases, the cited circumstances lead to the abandonment of the cultivations or to damages to the non-market functions.
In the table 1, we try to link the possible private and social benefits, the different typologies of goods and services provided by forests. This classification occurs, largely, in the alpine context and, in particular, in the area that we are considering as our "case study".

Typology	Private benefits	Social Benefits/Externalities
Forest products	Market based value	Production-chain activities
Hydro-geological	Preservation of forest soil, protection from erosion, landslides, floods	Preservation of lowland soil, water regulation
Climate regulation		Carbon fixation, air depuration
Landscape and recreation	Tourism	Recreational and aesthetic benefits

Table 1. Forest goods and services and private and social benefits

4.1 Brief considerations about markets and payment for ecosystem services

In regard to sustainable forest management,[12] it is necessary to pay particular attention to the costs of multi-functionality and to identify techniques that can internalize the positive externalities provided by forests. This effort will help ensuring a fair distribution of costs and benefits among the local population, the economic actors, the other stakeholders and the entire society. Many authors believe that important opportunities exist for provisioning forest ecosystem services, whether through "governance" (Gibson et al., 2000, 2005), "payment systems and markets" (Engel et al., 2008; Johnson et al., 2001), adjustments to life cycle processes, "or other means " (Patterson & Coelho, 2009). Clearly, as previously mentioned, this process is not easy because of the presence of utility flows that have the characteristics of public or mixed goods.
A category that has been widely analyzed in this context is that of the "Payment for Ecosystem Services" (PES). According to Muradian et al. (2010), the PES[13] "are a transfer of resources between social actors, which aim to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources". Wunder (2005) in particular, attributes the following features to the PES:

1. "a voluntary transaction where,
2. a well-defined environmental service (or a land use likely to secure that service),

[12] During the Second Ministerial Conference on Forest Protection in Europe, held in Helsinki in 1993, the following definition of «sustainable forest management» was introduced: "the correct management and use of forests and forest land in such ways and to such a degree as to conserve their biodiversity, productivity, renewal capacity, vitality and a potential that guarantees their important ecological, economic and social functions both now and in the future, at a local, national and global level without bringing damage to other ecosystems (www.mcpfe.org)". The European Commission (2001), subsequently, stressed that "sustainable forest management is the fundamental aim of development in the forestry sector, where the term «sustainability» refers not only to the regular production of timber, in the forestry sense, but also to the whole range of environmental, economic and social services performed by forests".
[13] Having in mind that "democratic mechanisms for allocating essential and non-substitutable resources may be preferable to markets, at least until basic needs are met" (Farley & Costanza, 2010).

3. is being 'bought' by a (minimum one) service buyer,
4. from a (minimum one) service provider,
5. if and only if the service provider secures service provision (conditionality)."

Very few PES schemes achieve the standards proposed by Wunder (Muradian et al., 2010; Porras et al., 2008). "Generating adequate resources or ensuring a just distribution of payments may require non-voluntary approaches such as taxes or mandatory service charges" (Patterson & Coelho, 2009). Whether payments should be voluntary or coerced through taxation should in fact be determined by the physical characteristics of the resource (Farley et al., 2010; Kemkes et al., 2010). "Services dominated by private good characteristics are amenable to voluntary payments, while services with public good characteristics are not" (Patterson & Coelho, 2009).

5. The landscape-recreational function

In this framework, an example is represented by the last typology presented in the table 1. It is the landscape-recreational function, which we now further analyze.

Natural resources[14] and, thus, forests, under certain conditions[15] that we identify in this and the following sections, could guide the local development of mountain areas, ensuring that income arising from the territory remains with local communities. Although the concept of local development is very broad, according to Greffe (1989, 1990) it can be considered "a process through which a certain number of institutions and/or local people mobilise themselves in a given locality in order to create, reinforce and stabilise activities using, as well as possible, the resources of the territory." In addition, "local development policies can help to achieve sustainable development goals. In fact, they are based on facilitating structural adjustment and enabling economies and societies to adapt to changing conditions, combating social exclusion and maintaining social equilibrium, and making the best use of social, economic and environmental resources in the local area" (OECD, 1999). It should be noted that, the increasing globalization of the economy and changing technologies have opened new markets and new competition with regard to which local development policies need to offer new responses.

According to the paradigm of the total economic value (TEV)[16], which mainly differentiates between use and non-use value, the landscape-recreational function can be subdivided into different components (Table 2). Specifically, "the recreational and scenic values require the direct use of the good: the first one derives from the possibility of carrying out tourist-recreational activities in environmental contests of quality, and the second one is related to the benefits produced by observing certain typologies of landscape" (Goio et al., 2008). In contrast, the evocative value "derives from the desire that a landscape encompassing aesthetic functions should exist, and from knowing that its

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[14] According to Barbier (2002), "these resources should be viewed as important economic assets, which can be called natural capital".

[15] We are referring, for example, to the control of natural resources, of investments and of legal and administrative rules.

[16] "The concept of total economic value (TEV) is one framework that economists have developed for categorizing the various multiple benefits arising from natural systems" (Barbier, 2002). In particular, it is a tool for the assessment of the intrinsic value of environmental goods aimed at economic evaluation of all functions regardless of their market interest.

associated traditions, culture and lifestyles continue to exist through its conservation" (Novelli, 2005).

From the perspective of local development, each component is related to different management options and to different benefits for people living inside and outside the local area.[17]

Recreational value - Areas with user-oriented management[18] - Areas with resources-oriented management[19]	Use value
Scenic value	Use value
Evocative value	Non-use value

Source: Gios & Clauser, 2009

Table 2. Different components of the landscape-recreational function

In table 4 we present some of the possible ways for "internalizing" the landscape-recreational function.

As shown in table 4, these ways are related to the different kinds of goods or services considered. In the case of private goods related to "user-oriented management" the internalization could be a ticket or a fee, while in the case of public goods what is needed is public support. Finally, for mixed goods connected with resources-oriented management, an approach based on the management of "commons" is required.

Typology of goods	**Target**	**Form of internalization**
Private	Areas with user-oriented management	Ticket
Public	Landscape as scenery	Public support
Mixed	Areas with resources-oriented management	Approach based on management of "commons"

Table 3. How to internalize the landscape-recreational function

With reference to the landscape-recreational function it is necessary to introduce an element characterizing many mountain areas: the tourism activity. Although this activity can foster the economic development and is a source of employment for the local population, in some cases, it can, also, lead to an imbalance among the various components of ecosystems, producing negative trade-offs (Dollinger, 1988). These trade-offs, sometimes, become very difficult to manage.

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[17] "One particular landscape typically has different functions for different people" (Heilig, 2003).

[18] We refer to areas that generate direct revenue. These include the following:

- areas with quick and excludable admission (e.g. adventure parks and golf courses) (Type I) and
- areas characterised by the provision of direct use[18] services and ad hoc facilities accessible through the payment of fees (e.g. hunting, fishing and mushroom collection) (Type II).

[19] Includes areas characterized by the provision of direct use, free of charge services (that is, open-access protected areas) that, under certain conditions, allow the creation of other sources of revenue (e.g. restaurants, hotels and guides) (Type III).

6. Some policy implications for decision-makers

In this framework, if the «ultimate aim» is the enhancement of the landscape-recreational function, a strategy able to incorporate jointly, forest management, the kind of landscape-recreational components and, finally, the characteristics of the tourism system has to be adopted. For this purpose, it is really important to take into account not only the specific characteristics of the forests, but also the system in which they are included. These considerations are summarized in the following table (n°4).
When the landscape is referred to as a specific resource for a well-defined project (for example an adventure park), the arrangements for tourist activities require large investments in equipment and structures with related management costs. In the area under consideration, cash flow can create, both directly and indirectly, jobs and sources of income. It also represents the underpinning of the traditional development pattern of some touristic districts, which has occurred since the 1960s in alpine areas. In other words, investments transform a public good into a private one. In contrast, in cases where the investments needed to utilize the natural resources are of small dimension, it is impractical to implement mechanisms of excludability from consumption, even if such mechanisms were technically feasible. It has to be noted that the forms of tourism, so-called "green" or "soft", fall mainly into this category.

Forest Landscape as:	**Touristic system**	**Forest utilization**	**Type of intervention**
1) Specific resource (adventure park[20])	Specific touristic project	Specialize areas oriented to a prevalent use	Active: equipment investments
2) Scenery	Weak and uni-directional links with the touristic system	maximization of biomass	Passive: diminishing the utilization, check fire and pest
3) "Complex" resource (visit to natural park)	Strong and bi-directional links with the touristic system	Naturalistic selvi-culture	Active: knowledge and dissemination investments

Table 4. Intervention related to tourism exploitation of forests

The central objective is to find, even in the case of landscape and recreational activities that do not require large investments, mechanisms that allow the enjoyment of those activities after a specific payment is made as a compensation supporting local development.[21]

[20] They are acrobatic paths realized in forested areas that allow direct contact with nature and the possibility of directly exploiting the trunks of the trees for the preparation of the various paths. These paths are very well developed in France, the United Kingdom (www.ttadventure.co.uk) and Italy (www.agilityforest.it).
[21] A good example, in this context can be represented by the "Sentiero del Castagno" (Alto Adige, Italy, http://www.valleisarco.info/it/attivita/estate/escursionismo/sentiero-del-castagno.html) or "Les Route du Bois in Belgium, (www.lesroutesdubois.be) .

In addition some further observation can be made. In the case of specific resources, the forest is managed to extract timber. However, this benefit is a "sub-product" because the dominant use of the forest is as an adventure-park. In the second case, where the forest is considered as scenery (for example in relation to sport use), an increase in the overall timber quantity produced, is a positive aspect. Its perceived value in fact increases, but the associated management costs, which clearly grow (for example for checking fires and pests), are not compensated by the market. Consequently a public support is needed. Finally, with respect to "complex resources" a double objective, related to tourism activity and timber production, has to be achieved. In this case, higher costs related to timber production should be compensated by tourism revenues. However that initiative requires investments to disseminate and share knowledge amongst the general stakeholders who use natural resources.

6.1 The need for a participative approach

The future development, especially of mixed goods in the Alps, will depend, largely, on the ability to involve local stakeholders in the environmental protection and promotion processes, establishing, at the same time, the priorities for each single area. Clearly, to make this involvement work, local actors should be able to direct the management of natural resources, in general, and of forests in particular, towards their own interests and needs. They also need to control the management options adopted.[22] As known, the participative approach of local stakeholders has emerged in the last 15 years, following, precisely, the evolution of the concept of sustainable development. It is based on the belief that citizens are able to shape their own future. It is "thought up on the conviction that people are capable of defining their own future" (Jennings, 2000). As a consequence, "it uses capabilities and local knowledge to guide and define the nature of actions and strategies" (Jennings, 2000). Through efficient participative development processes, it is possible to take into account some territorial dimensions that are often neglected or not considered, such as traditions, beliefs and habits, thereby creating the preconditions for the implementation of spontaneous action by the communities involved. In the development initiatives related to the management of natural resources, however, this participation cannot be confined to the mere application of techniques to facilitate the involvement of larger social groups. On the contrary, it is fundamental that the stakeholders become aware of the issues related to natural, social and human capital, thus creating a shared sense of the problems and the basis for potential collective actions. Since sustainability refers to three different dimensions (environmental, social and economic), integrated and comprehensive territorial development is required. In order to pursue it, as already noted, it is necessary to understand how the stakeholders perceive the factors connected with sustainability, facilitating and promoting an enriching exchange of views, knowledge and initiatives. To obtain a complete and reliable overview, these exchanges should involve both public and private actors, producing new synergies and new partnerships in the area.

This field is very complex and a pre-eminent participative process does not exist, because the process "is perceived and implemented in different ways" (Buchy & Hoverman, 2000). In fact

[22] According to Carpenter and Folke (2006) "management actions should be viewed as experiments that can improve knowledge of social–ecological dynamics if the outcome is monitored and appropriately analyzed".

in the literature there are many different classifications of participation[23], "because of the concept give rise to a wide range of interpretations" (Lawrence, 2006). Some writers take into account the degree of involvement, which can be strong or weak. For the World Bank (1996), in fact, "participation is strong if there is a real influence on development decisions by local actors and weak in the case of a simple informative involvement concerning the implementation or benefits of a particular development activity". Other classifications (Rowe & Frewer, 2000) focus on the nature rather than the degree of engagement, identifying different types of public engagement by the direction of communication flows between parties. According to this view, information dissemination to passive recipients constitutes ''communication'', gathering information from participants is ''consultation'' and ''participation'' is conceptualized as two-way communication between participants and exercise organizers in which information is exchanged in some sort of dialogue or negotiation. Others (Biggs, 1989) describe the level of engagement as a relationship that can be ''contractual'', ''consultative'', ''collaborative'' and ''collegiate''. Finally, some engagements stand between pragmatic participation and normative participation. "The first focuses on process, suggesting that people have a democratic right to participate in environmental decision-making", while the second "arguments focus on participation as a means to an end, which can deliver higher quality decisions " (Reed, 2008).

For this reason, different kinds of participation can be implemented, in relation to: "a) the characteristics and conditions of any specific context, b) the aims that have to be realized, and c) the ability of the stakeholders to influence the final results" (Richards et al., 2004; Tippett et al., 2007). The literature urges to move towards a high degree of participation (Arnstein, 1969; Johnson et al., 2004) or to a strong participation, as defined by the World Bank (1996). In addition, several authors believe that the success of participatory processes should be institutionalized[24] (Richards et al. 2004). They require a) "a sufficiently detailed and clear description of the context and objectives" (Reed, 2008), b) "to identify appropriately and adequately the role of each actor, be it public or private "(Purnomo et al., 2005), to manage any conflicts, and c) to encourage the actors to develop an adequate motivation and ability to participate, triggering a process that might be called «educational»[25]. This approach requires

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[23] For example:

- "Participation is concerned with . . . the organised efforts to increase control over resources and regulative institutions in given social situations on the part of groups and movements hitherto excluded from such control" (Pearse & Stiefel, 1979);
- "Participation can be seen as a process of empowerment for the deprived and the excluded. This view is based on the recognition of differences in political and economic power among different social groups and classes. Participation in this sense necessitates the creation of organisations of the poor which are democratic, independent and self- reliant!" (Ghai, 1988);
- "Participation is a process through which stakeholders influence and share control over development initiatives and the decisions and resources which affect them" (World Bank, 1996);
- "Participatory development stands for partnership which is built upon the basis of dialogue among the various actors, during which the agenda is jointly set, and local views and indigenous knowledge are deliberately sought and respected. This implies negotiation rather than the dominance of an externally set project agenda. Thus people become actors instead of being beneficiaries" (OECD, 1994).

[24] If participation is a democratic right and not just a legislative goal.

[25] The issue is really one of education and politics: neither the general public nor decision makers appear to be well-informed concerning the relative contributions of ecosystem services and economic growth to our well-being" (Farley et al., 2010).

knowledge of each actor to check the degree of understanding and awareness of the project, with the aim of filling any gaps. Today, it is increasingly necessary to include in the participatory processes the so-called «experts»: a) "local stakeholders, namely those who have proven experience and knowledge (location-specific), not only scientific but also operational, in reference to the area and location of interest, b) the external stakeholders, namely those who truly understand the phenomena and who have a more scientific/universal knowkledge" (McCall, 2003; Fraser & Lepofsky, 2004). These categories are generally involved "simultaneously" (Reed, 2007), because local knowledge combined with what science can contribute, leads to a more complete understanding of complex systems and processes (Johnson et al. 2004), as well as the learning pathways within each category and between the two categories.

6.2 Evidence from alpine examples: Logarska Dolina

As presented in table 4, specific economic and political tools need to be created in order for stakeholders to pursue local development. Payment for ecosystem services is rather common for some resources, but not for scenery and landscape services.

There are examples of fees connected with the touristic and recreational uses of alpine territories, but they are mainly not resource-related. Many local communities are requiring daily-payment from tourists, but they cannot be individuated as in the categories of Table 4 because their practices have general purposes and, sometimes, are not supportive of the maintenance of landscape. In fact, it is necessary to aim for payments capable of financing management activities devoted to preserving and enhancing the natural and forest features of the Alps.

The role of the participation of local inhabitants is clearly presented with reference to "Logarska Dolina": a valley seven kilometers long, covered by meadows and forests, with some waterfalls, is located in the northern part of Slovenia. The attraction of this area lies in its abundant natural sights, coupled with an almost pristine environment. It has been attracting hikers since the late 1800s. Also characteristic are the farmsteads, which have, over the centuries, aided in building a cultural landscape. This valley has recently been part of the network of the "European Destination of Excellence".[26]

In 1987, the local council of Mozirje decided to create a "Landscape Park". However, there were not enough financial resources for protecting the local flora and fauna and developing the recreational and management structures needed. An additional problem has been the increasing number of tourists visiting the valley by car and creating such diseconomies as pollution, chaos, fires and rubbish.

The small local population (at that time, ten families with 35 persons) decided to prevent an excessive use of the local amenities. According to local oral norms, the land is private and it has not been divided over the centuries through inheritance. The only public property is the

[26] The population of this district is primarily engaged in forestry, animal husbandry and, most recently, tourism, the prosperity from which is largely supported by this area's great natural beauty. An unspoiled natural environment, coupled with the fact that this region had not been overdeveloped, has worked to the advantage of the local community. However, the people of the Solcava District are well aware that this pristine environment must be preserved at all costs. For this reason, they have chosen to develop high quality tourism, which emphasizes the individual, offering him peace as well as the opportunity to enjoy an active holiday in harmony with nature" (from official website www.logarska-dolina.si).

traffic road situated in the middle of the valley. Although some families were able to make a living through their occupations in agriculture and forestry, others might be interested in developing some services for tourists. The local population agreed that the option of direct management was the optimal choice, even though it involved delegating this activity to external bodies. In 1992, the council of Mozirje created a public company devoted to the management of the park of the Logarska Dolina valley. There are 14 members: ten local families, two companies owning houses in the valley for the recreation of their staff, the manager of the local hotel and a member of the local tourist board. Thus, decisions are made only by persons who are living in the valley or who have a strong interest in the valley. They are members who are aiming at long-term results by taking care of the cultural landscapes as well as the employment of locals.

A decision to introduce a fee for visiting the valley was made to reduce the number of cars and to restore a sense of quiet. The valley should be crossed, preferably, by foot, with tourists leaving their cars at the free parking places located on private land at the beginning of the valley. The entering fees generated most of the money needed to manage the public company. However, more recently, most financial support has been coming from external sources, such as those from the European Union or national government. The small managing board permitted a review of a sustainable model of development, which rejected the ideas of creating golf courses, tennis halls, or new buildings on land not utilized. Few developments were permitted other than nearby local farms in order to avoid landscape fragmentation. "We are creating our house, not a leisure park," reflects the main aim of the board of the public company. What has been created is a heating network, using biomasses, and a purification plant. Five permanent positions have been created for managing the local resources and the activity of the public company.

In 2008, the Logarska Dolina Valley was incorporated into a large park that included the surrounding three valleys of the Solcava district. The cooperative system of management was used as an example. However, it has not been incorporated into the new authority. "The sense of trust and reciprocity is easier when there are few persons involved and all of them are sharing almost the same interests; the largest and most direct right of participation is the better way of management" declared Avgust Lenar, the Director of the Logarska Dolina public company (CIPRA, 2007).

Similar communities, which required the payment of a fee for entering the territories, could be found within the Alps in Natural Parks or in some municipalities where some outstanding recreational or landscape features are located (Cimoliana Valley in Friuli, Italy, nearby the peaks of Cime di Lavaredo).

Dimensioning the population involved, the property regimes and the structure of the political and management organization is playing a relevant role in defining the participatory tools and the design of the development plan and strategies. As previously mentioned in Paragraph 6.1, there are no models that can easily be adapted for the Alps. Thus, there is need to adapt or create new participatory tools and strategies, paying particular attention to the characteristics of territory and stakeholders.

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14

Economic Valuation of Watershed Services for Sustainable Forest Management: Insights from Mexico

G. Perez-Verdin[1,*], J.J. Navar-Chaidez[1], Y-S. Kim[2] and R. Silva-Flores[3]
[1]*Instituto Politécnico Nacional*
[2]*Northern Arizona University*
[3]*Private Consultant*
[1,3]*Mexico*
[2]*United States of America*

1. Introduction

Ecosystem services are the benefits that people obtain from ecosystems (Brauman et al., 2007). Recognizing the importance of the services provided by ecosystems for human well-being is not a new idea, going as far as Plato (Feen, 1996) and the economic conceptualization of ecosystem values (Coase, 1960; Feen, 1996). However, the scientific and practical interests in assessing and trading ecosystem services have not gained momentum until the 1990s when pioneering works by Daily (1997) and Costanza et al. (1997) galvanized the field. Among the ecosystem services that received increasing attention in the recent years are the hydrological services due to the role of water as a vital, and sometimes decisive, element in human life (Pare et al., 2008). Hydrologic services encompass a range of benefits that terrestrial ecosystem produces in terms of freshwater. These services can be grouped as: improvement of extractive water supply, improvement of in-stream water supply, water damage mitigation, provision of water related cultural services, and water-associated supporting services (Brauman et al., 2007).

The majority of hydrological services take place in the highlands of forest watersheds (Messerli et al., 2004). In these areas, upland forest watersheds work as a source that collects, manufactures, and distributes water and provides hydrological services to lowlands (Neary et al., 2009). Various components of the water cycle (i.e., evaporation, infiltration, surface run-off) critically depend on forest cover. If the forest cover is affected, so it will be the quality and quantity of the water provided to downstream users (Brown et al., 2005). In developing countries, such as Mexico, changes in forest cover are caused among other things by the local economic conditions in which landowners live. While searching for basic needs (food and shelter), they exercise excessive pressure over the forests eventually triggering forest fragmentation and deforestation (Perez-Verdin et al., 2009).

Based on the methods used for their economic valuation, hydrological services can be classified into two broad categories of values: marketed and non-marketed. The economic

* Corresponding Author

value of the former is reflected through the market price determined mainly by its demand and supply (i.e., drinking water) while the latter, traded under imperfect markets, requires a more complex evaluation that involves evaluating consumer's preferences and behavior (i.e., evaluation of recreation sites). The sum of these services gives the total economic value (TEV) of a forest watershed. Because of the quasi-public good nature of hydrological services and the presence of externalities, failure to recognize the TEV of a watershed can lead to depletion, degradation, and overexploitation of forest resources and eventually loss of social welfare (Plottu & Plottu, 2007).

Recently, research has focused on assigning economic values to environmental services to redirect policies for sustainable forest management. The intention is to help landowners reduce the impact of externalities by giving monetary incentives and implement best management practices to regulate the quality/quantity of water (Pagiola et al., 2003; Muñoz-Piña et al., 2008). Among the new schemes include the formal articulation of incentive-based instruments, such as Payments for Ecosystem Services (PES) and Markets for Ecosystem Services (MES) (Jack et al., 2008; Gómez-Boggerthun et al., 2010). While the design and operation of various international PES and MES programs have been started by local governments, many of them now promote the participation of the private sector, non-government organizations, and the general public (Paré et al., 2008).

The major objective of this chapter is to underline the importance of assigning economic values to hydrological services as a means to achieve sustainable forest management. The paper first introduces critical inputs of the water balance and best management practices for watershed resources. It also describes the types of watershed services and how they can be valued. The paper then analyzes the cases where non-market valuation techniques have been implemented for various types of watershed services in Mexico. And finally, it discusses the operation of a Mexican PES program and its impact on watershed services.

2. Water balance and best management practices

The assessment of available water resources is central to economic valuation of hydrological services. The economic valuation of water resources involves knowledge of the supply and demand sides and eventually to the search for effective management policies. The determination of available water within a watershed is given by the water balance and depends on the magnitude of inputs and outputs and the storage capacity. The basic input is precipitation (P_T) and is either lost to evaporation (E_V) and transpiration (T_R) or routed through small pathways of overflow and interflow to give surface runoff (Q) and infiltration (I) (Hiscock, 2005). Thus, the water balance model, estimated for a given period of time ($\partial A/\partial t$), is the difference between inputs and outputs. The larger the difference between inputs and outputs, the more supply water there is to end users. In this case, Inputs= P_T and Outputs = $I + E_V + T_R + Q$. Therefore, the water balance can be expressed as:

$$\partial A/_{\partial t} = P_T - (I + E_V + T_R + Q) \quad (1)$$

In mountainous forest watersheds, precipitation is partitioned into throughfall, interception loss, and stemflow (Navar, 2011). Throughfall is the rainfall portion that reaches the ground by passing directly through or dripping from tree canopies. Interception loss is the rainfall retained on the canopy that evaporates back to the atmosphere; it is composed mainly on the amount of precipitation stored by canopies and the evaporation of stored canopy water.

Stemflow is the rainfall portion that flows to the ground via trunks or stems (Dunkerley, 2008). Litter retains part of the throughfall and stemflow and infiltrate into the mineral soil increasing soil moisture content. Evapotranspiration is the amount of water vapor that leaves soil and vegetation via evaporation and transpiration. Factors that control evaporation from soils are the current water content, the water content at wilting point, and the soil water content at field capacity. Factors that affect transpiration are the type of vegetation, density, and age.

Conventional forest management practices, that include logging and grazing, affect tree density, canopy cover, and tree composition and structure (Brown et al., 2005). Hydrologic studies in the United States have demonstrated that selective harvesting and clear-cutting promotes increased discharge because of a reduction of stand density and canopy cover that demand less water for transpiration (Swank et al., 1988; McBroom et al., 2008). Non-conventional forest disturbances that cause tree mortality include forests fires, pests and diseases, strong winds, etc. Forest fires of large spatial scales and severity, in addition to tree mortality, also cause soil water repellency (Martin & Moody, 2001). Water repellency reduces infiltration and often promotes surface runoff and soil erosion beyond any other forest disturbance (Pierson et al., 2008). In general, tree mortality beyond natural causes reduce interception loss and transpiration leaving more net precipitation (throughfall) for other processes such as soil moisture content, aquifer recharge, and surface runoff (Brown et al., 2005; Ikawa et al., 2009). In addition, streamflow and aquifers are enriched with sediments and chemicals washed out from the soil that reduces usability. Other human-related disturbances are road construction and maintenance, and harvest-related activities that promote soil compaction and reduce soil infiltration at specific places in the watershed.

The aim of best management practices (BMP) is to reduce the effect of non-point and point sources of degradation that affect water quality and quantity (McBroom et al., 2008). Examples of non-point sources, which are characterized by a widespread and diffused generation, include cropland, harvesting areas, animal feedlots and grazing lands, impervious surfaces (e.g., roads, land rocks, deforested sites, urban areas), and construction sites (Neary et al., 2009). Transport of sediments, organic matter, and nutrients, such as nitrogen and phosphorus are examples of point sources. Harvesting, grazing, and agriculture can lead to increased rates of runoff and erosion. Rates of material export from impacted watersheds to water resources, while highly variable within and between land uses, exceed those for natural or undisturbed land uses (Andreassian, 2004). Because of this characteristic, the application of BMP is mainly oriented to reduce the effect of non-point sources.

Effective BMPs to reduce the effect of non-point source loads should target changes in current land-use practices, construction and operation of equipment, machinery, and the use of structures to retain or otherwise control the movement of water and material (McBroom et al., 2008; Neary et al., 2009). Also, effective BMPs need to consider the local conditions (e.g., geology and soils, topography, climate, and hydrology), landowner expectations, and the nature of the source of the polluting material (e.g., harvesting, grazing, or agricultural land uses) in which impacts are occurring. Overall, watershed BMPs are oriented to (1) minimize soil compaction and bare ground coverage, (2) separate exposed bare ground from surface waters, (3) exclude fertilizer and herbicide applications from surface waters, (4) inhibit hydraulic connections between bare ground and surface waters, (5) avoid disturbance in steep convergent areas, (6) provide a forested buffer around streams, and (7) build stable road surfaces and stream crossings (Jackson & Miwa, 2007; Neary et al., 2009).

In Mexico, the national water, environmental protection, and forest laws are the basis for regulating watershed management practices. Coupled with the federal laws, almost every state in the country has specific regulations that complement those issues where the federal laws do not apply. Based on this set of laws and regulations, common examples of BMPs that involve forest vegetation and water include: the provision of forested buffer around streams, stabilization and closure of third-order roads immediately after harvesting, construction of culverts on primary and secondary roads crossing streams, pre-harvest planning for cutting, skidding and loading zones to avoid increasing hydrologic and sediment source connectivity to stream channels, and the perpendicular arrangement of forest residues to reduce soil erosion, among others.

In the past, the implementation of these BMPs was adopted by landowners who would evaluate the cost and benefits in either doing another activity or doing nothing. Since these practices, which we have identified as externalities, would reduce their economic profits, many landowners did not comply with the regulations leading to increased rates of erosion and sedimentation (Muñoz-Piña et al., 2008). Nowadays, the cost of BMPs is mostly shared with the government; however, the private sector, non-government organizations, and the general public are participating as well. This type of cost-share programs, which embrace the known concept of internalizing externalities, is discussed in section 4 of this chapter.

3. Economic valuation of watershed services

The need of economic valuation of watershed services stems from their quasi-public and non-rivalry nature, the presence of externalities, and scales of production (Pattanayak, 2004; Brauman et al., 2007; Plottu & Plottu, 2007). In a market economy, watershed services without economic values will not be provided at optimal levels. The quasi-public, non-rivalry nature implies that it is difficult, if not impossible, to exclude an individual from using watershed services (e.g. soil retention), and several individuals can use them simultaneously without diminishing each other's use values. The presence of externalities means that the economic benefits of users of these services will not be deviated to compensate providers. And regarding the scale of production, these services are characterized by economies of scale in production; the larger the watershed, the lower the marginal costs (Pattanayak, 2004).

Valuation of watershed services also implies understanding the different types of benefits a watershed offers to ecosystems and society. A forest watershed not only functions like a basin which receives and stores water from precipitation, surface runoff, or infiltration, but also cleans water, retains sediments, provides habitats for wildlife, sinks CO_2, and offers many environmental amenities for humans (Brauman et al., 2007; Locatelli & Vignola, 2009). Some of these benefits can be valued through conventional methods that use market-based approaches. For example, the useful life of a dam can be valued through estimations of the rate of sedimentation and the years left to sustain fish. Other benefits require detailed information and more complex approaches that estimate for example the value of environmental services for present, future generations, or consider the presence of externalities (Field, 2008). For example, if fewer recreation opportunities are provided in the watershed, due to water loss resulting from harvesting or grazing, recreationists may act and eventually offer a fee to preserve the watershed and recover the loss of recreation opportunities. In this section, we provide a brief summary of the different watershed values and the means to estimate them.

3.1 Watershed values

For the purpose of this work, we will focus on two main types of watershed values: use and non-use values (Freeman, 2003; Field, 2008). Use values, which consist of consumptive and non-consumptive uses, refer to the situations where people directly or indirectly interact with resource use (Field, 2008). Consumptive use values are derived from extractive resource uses such as timber, commercial fishing and hunting, and the use of water for irrigation and drinking. Examples of non-consumptive uses values are benefits from resources with a minimal or imperceptible extraction and include those from boating, swimming, ecotourism, and camping.

Non-use or passive-use values refer to the situations in which people place monetary values on resources independent of their present or future use (Field, 2008). For example, people may be willing to support a long-term program intended to maximize water quality even though their offspring, not they, will receive the benefits. Despite the controversy that these types of values should not be considered in mainstream economics, because they reflect altruism and difficulty to assess, Freeman (2003) argues that non-use values can be defined within a utility theoretical framework and should be considered as public goods. Freeman further contends that ignoring non-use values could lead to wrong policies and resource misallocation.

The rationale for assigning values to watershed services also lies on the many biochemical cycles that take place in the watershed, the water and soil conservation functions, and the provision of wildlife habitats and amenities (Pearce, 2001; Pattanayak, 2004; Brauman et al., 2007). Water is the principal medium in which many chemical reactions occur and watersheds provide a variety of conditions in which those chemical reactions take place (Ward & Trimble, 2004). Water, Carbon, Nitrogen, Oxygen are among the key elements whose maintenance depends on the management of forest watersheds. Altering these cycles could interrupt the flow of environmental services, particularly water, to downstream communities (Figure 1). Therefore, the main question is how these hydrological processes, defined by a local drainage unit, can be manipulated to be fairly useful to society.

Figure 1 shows the relationship between hydrological processes and economic values to humans. A change in physical or chemical properties of water causes a change in the quality and quantity of the liquid provided. Discharges from non-point pollution sources affect the quality of water and force resource managers to use expensive processes, equipment to clean the water. Conversely, to address the feedback loop, excessive fishing may cause a change in the fish population. Estimating an improvement of watershed benefits involves the use of economic models to determine the monetary units people place on both use and non-use values (Freeman, 2003).

The TEV is a concept that illustrates the whole worth of ecosystem services. Due to the nature of some services, hypothetical markets are created to elicit values through a variety of economic techniques, including: (a) direct market valuation approaches, (b) revealed preference approaches, and (c) stated preferences approaches (Freeman, 2003; Champ et al., 2003;). Direct market valuation methods use data from actual markets and thus reflect actual preferences or costs to individuals. Revealed preference techniques are based on the observation of individual choices in existing markets that are related to the ecosystem service subjected to valuation. Stated preference approaches simulate a demand for ecosystem services by means of surveys on hypothetical changes in the provision of ecosystem services (TEEB, 2010). Selection of the best technique depends on the objectives of the researcher, the type of use values, and the type of ecosystem services under evaluation.

Again, due to the nature of some watershed services, uncertainty is an issue that must be considered in every valuation work. As suggested by TEEB (2010), one way to deal with uncertainty is the use of the data enrichment or data fusion approach which combines the use of revealed and stated preference methods. The main advantage of these hybridized approaches is that they overcome technical uncertainty due to application of valuations tools and uncertainty with regard to preferences about ecosystem services. However, their application generally depends on available financial, human, or time resources.

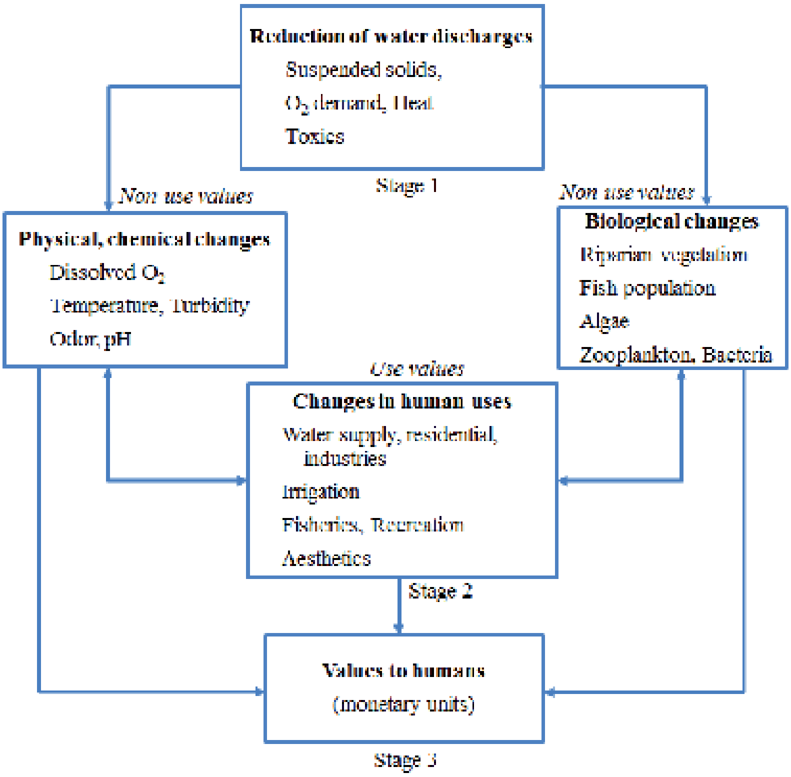

Fig. 1. Types of hydrological values and flow of services to society. The sum of use and non-use values gives the total economic value (From Freeman, 2003, page 31).

3.2 Theoretical framework of economic valuation

Because of the diversity of watershed benefits, which include use and non-use values, placing monetary units depends on the type of services provided, the actual and desired conditions of the watershed, and people´s social status (Freeman, 2003; Brauman et al., 2007). Although valuation of all watershed benefits is possible, many studies focus on few or single services. The most common benefits include drinking, irrigation, wildlife habitats, prevention of soil erosion, flood protection, fisheries, and hydropower (see Pearce, 2001; Locatelli & Vignola, 2007, for a literature review of watershed services). To account for reliable estimations of the watershed value, information on the extent of the change in quality and/or quantity of the service is required. The marginal value, the extra monetary units a person would be willing to pay for an additional unit of the service, depends on the magnitude of the change a person expects with her/his contribution as well as on the beginning and ending points of that change (Brauman et al., 2007).

Concerning to watershed services, willingness-to-pay (WTP) is the maximum amount of income an individual will pay for an improvement in current conditions of the watershed, or the maximum amount of money to avoid a decline in those current conditions (Freeman, 2003). The WTP measure for valuing watershed services is a function of a vector of individual´s social characteristics (such as income, education, family size, among others), the price (p), and quantity of the service (q) (Freeman, 2003). Theoretically, WTP can be expressed as either in terms of an utility function $V(p, q, y)$:

$$V(p, q^*, y - WTP) = V(p, q, y) \tag{2}$$

or in terms of the minimum expenditure function $m(p, q, u)$,

$$WTP = m(p, q, u) - m(p, q^*, u), \text{ when } u = V(p, q, y) \tag{3}$$

where y is income and q^* represents a new condition or improvement in the watershed service ($q^*>q$). The WTP is thus the amount of money to pay that would make such individual indifferent between the current condition (y and q) and the new, improved state $[(y–WTP), q^*]$.

To estimate the economic value of watershed services, particularly non-consumptive or non-use values, typically researchers use a stated preference technique called Contingent Valuation (CV). This technique employs survey-based information to directly elicit households' preferences and build a contingent market through which respondents may state their willingness to pay for a specified provision change in a particular service (Mitchell and Carson 1989). The CV approach first involves describing the current situation of a non-market good, how it can be improved, and then asking respondents whether or not they would pay for the improvement of the good (Boyle 2003). It is called contingent valuation, because people are asked to state their willingness to pay contingent on a specific hypothetical scenario and description of the environmental service (Carson & Groves 2007). The willingness-to-pay results can then be used by decision makers to weigh policy options. Details on CV description can be found in Mitchell & Carson (1989), Boyle (2003), Schlapfer (2008), TEEB (2019), among others.

4. Valuation of watershed services in Mexico

In recent years, various studies have been conducted to estimate the value of watershed services using non-market valuation techniques in Mexico. To document these cases, several sources of information where a consistent valuation approach was used were reviewed in this chapter. The first information source included a literature search from all available databases (e.g. Web of Science) and the web for nonmarket valuation studies. A brief review of the abstracts and introductions served to select articles directly related to watershed services and the valuation approach. Second, all articles relating to the topic were thoroughly reviewed to identify the main watershed services and other information needed to be considered. The search also included the citations of published articles to find any unpublished data or papers. Besides the WTP amount and the watershed service being evaluated, additional information collected in the review was altitude, latitude, longitude, and precipitation. The search eventually gave 13 cases including Mexico City and other cities located across the country. The watershed services ranged from wildlife habitat preservation, soil retention, and recreation, to drinking, irrigation, fishing, and hunting. The cases identified were compiled and georeferenced in a geographical information system (GIS) database.

Table 1 shows the cases included in the literature review. Most of the studies were located in high elevation areas (e.g., more than 1,000 meters above sea level) which gave indication of the relevance of the watershed highlands to provide environmental services, and the need to protect them. The WTP, obtained through the contingent valuation approach, ranged from US$ 0.45 to 15.8 per month and household, being the Mexico City the case with the highest WTP. These figures represent between 0.33 and 11.8% of the 2011 per-capita minimum wage (the minimum wage is US $134/person/month; DOF, 2010). The main types of services provided by the watersheds were wildlife habitat, drinking, and soil retention. The most common management practices proposed in the studies were reforestation, soil conservation works, and reducing harvesting, grazing and risk of fire, among others.

It is important to note that in many studies it was difficult to clearly identify the main watershed service. During the search, several works were discarded due to the inconsistency of valuation approaches, the service being evaluated, and the type of WTP units (for example, WTP was expressed in $/month/person, $/year/household, $/visit, etc). Out of the 25 studies reviewed, only those listed in Table 1 were selected since they allowed cross-site comparisons. Based on the predominant service, each case was classified into two major groups: those with consumptive use values (e.g., drinking, fishing, irrigation and hunting) and those with non-consumptive use values (ecotourism, wildlife habitat, recreation, soil retention); the latter also included non-use values. The classification yielded seven cases in the first group and six in the second. To test for WTP differences in the type of use values, one-way analysis of variance indicated that there was no significant relationship in the WTP[†] (n=13, F=2.541, p=0.14). Neither there was for elevation (n=13, F=0.001, p=0.99) and moisture index (n=13, F=0.978, p=0.34), the two additional physical variables of the cities. The lack of significance in the WTP differences means that individuals appreciate both consumptive and non-consumptive uses similarly. However, in practical terms, the individual benefits estimated for consumptive use values were 47% higher than those for non-consumptive use cases.

4.1 Government-supported watershed markets

Various Latin-American countries have started programs to intensify the production of watershed services in forest ecosystems. In 2003, Mexico launched an innovative PES program to help landowners to protect forest watersheds in critical areas of the country. The program, called in Spanish as *Pago de Servicios Ambientales Hidrológicos* (PSAH), had three main goals: to reduce deforestation in areas with severe water problems, apply best management practices for sustainable forestry, and reduce illegal logging (Muñoz-Piña et al., 2008). The PSAH consisted of direct payments to landowners, whose lands were mostly covered by temperate or tropical forests, during a 5-year period in which landowners executed a series of BMPs to protect the watersheds. Part of the PSAH's innovative approach is that it was funded through an earmarked portion of federal fiscal revenues from water

[†] Due to the small sample size, differences between the use values were also evaluated with the non-parametric Mann-Withney test. Results corroborated the results of no significant differences for WTP, elevation, and moisture.

fees, so the program involved users and producers of environmental services. The payment, offered as an economic compensation or subsidy, was based on the opportunity cost of using the land for agriculture or livestock (Muñoz-Piña et al., 2008), not on the non-market valuations we have discussed above. Initially, it oscillated between US$ 23 and 30 per hectare depending of the type of forest (CONAFOR 2004)‡.

As expected, the PSAH received various criticisms. The government used the opportunity costs of the two primary economic activities (agriculture and livestock) to estimate the compensation. Though there are no official reports, this was probably due to the type of information available initially. Government officials have said that these payments are currently under evaluation and will be reassessed with new information based on market and non-market methods. Also, the PSAH has been regulated by the government itself who

Study site	Watershed service [a]	Type of use value [b]	Elevation (meters)	Moisture index [c]	Adjusted WTP (US$/month) [d]	Source
Ciudad Obregon, SON	WH, F, SBR	NC	35	0.146	6.12	Ojeda, et al. (2008)
San Luis Rio Colorado, SON	F,H, SBR	C	40	0.055	6.39	Sanjurjo (2006)
Parral, CHIH	D	C	1,620	0.089	8.91	Vasquez et al. (2009)
El Salto, DGO	D,SR	C	2,540	0.250	2.08	Silva-Flores et al. (2010)
Tapalpa, JAL	I,D	C	1,950	0.135	9.10	Lopez-Paniagua, et al. (2007)
Mexico City, DF	D	C	2,240	0.064	15.81	Soto and Bateman (2006)
San Cristobal de las Casas, CHIS	D,WH	C	2,120	0.306	1.82	Gutierrez-Villalpando (2006)
Tepetlaoxtoc, EDOMEX	WH	NC	2,300	0.088	4.98	Jimenez-Moreno (2004)
Oaxaca, OAX	WH	NC	1,555	0.105	3.11	Garcia-Angeles (2006)
Tlaxco, TLAX	WH, SR	NC	2,588	0.074	1.83	Orozco-Paredes (2006)
Metztitlan, HGO	WH, SR	NC	2,080	0.091	0.45	Monroy-Hernandez (2008)
Alamos, SON	WH, SR, D	NC	400	0.046	8.23	Chan-Yam (2007)
La Paz, BCS	D	C	10	0.048	10.15	Aviles-Polanco, et al. (2010)

[a] WH, Wildlife habitat; D: Drinking; I, Irrigation; F, Fishing; H, Hunting; SBR, Scenic Beauty and Recreation; SR, Soil Retention

[b] C= Consumptive, NC = Non-consumptive

[c] Based on precipitation and evaporation data (Willmott & Feddema, 1992). The moisture index goes from 0 to 1, where dryer areas tend to zero.

[d] Based on Februrary-2011 price levels (US$1 = MEX$13) and 10-year average of the National Consumer Price Index =4.03%)

Table 1. Willingness-to-pay for watershed services in Mexico

‡ We tried to compare the PSAH payments to those WTP values extracted from literature (See Table 1). The comparison turned difficult due to the differences in methods, sampling issues, and monetary units.

acted as a monopsonistic buyer on behalf of water users (Muñoz-Piña et al., 2008). The government basically established a price and waited for landowners to offer their forests for conservation. In retrospective, some landowners may have rejected the program because the compensation was not enough to fully cover transaction and opportunity costs. In addition, the initiators of the program never considered a baseline to monitor the impacts of the economic compensations on the quality/quantity of water. Today, evaluating the performance of the first periods of the program is difficult due to the lack of a monitoring plan (Consejo Civil Mexicano, 2008).

Despite of these and other criticisms, the PSAH has endured and contributed to sustainable forest management by offering landowners more incentives to provide environmental services, while clarifying and defining property rights, thus reducing the impact of externalities (Muñoz-Piña et al., 2008). In the first years of operation, the program had paid almost US $200 million and protected about 1.5 million has of strategic watersheds (Chagoya & Iglesias, 2009). The PSAH also has received full support from the Mexican Congress which recently authorized the participation of state and local governments, non-government organizations, private entrepreneurs, and society to increase the funds. Examples of this type of mixed funds are found in Centro Montaña de Guerrero; Tehuacan, Puebla; Coatepec and Texizapa in Veracruz; Cupatitzio, Michoacan; and Chinantla Alta, Oaxaca; among others (Paré et al., 2008). Most of the collected fees have been used to implement selected best management practices in the watersheds' highlands.

The examples of multi-stake voluntary participation in the payment for environmental hydrological services have received ample attention due to the commitment of the multiple parties to promote sustainable forest management. Although programs like PSAH are not in themselves sufficient conditions for sustainable forest management, they are necessary conditions for efficient policy making. Assigning property rights to providers and consumers help delineate the responsibilities of each group. The former receives an economic compensation to reduce the effect of externalities in the management of forest resources. The latter express their demand for environmental services through their WTP for receiving a better quality watershed service. The interaction between providers and consumers helps partially correct market failures and eventually reduce forest degradation. Programs like PSAH not only generate the necessary funds for forest conservation, but also will increase the quantity/quality of watershed services (Pagiola, et al., 2003). The future of PSAH and similar programs lies in the clear definition of the real value of watershed services, correct assignment of property rights, and the continuity of funds.

5. Conclusions

This chapter discussed the relevance of valuing watershed services to achieve a sustainable management of forest resources in Mexico. It presented a simple method to estimate water balance and identified BMPs, discussed the main types of values a watershed can offer, how they can be valued, and examples of cases based on non-market valuation and government-supported programs. Due to their non-exclusive, non-rival characteristic, watershed services need to be economically valued using diverse approaches to be produced at optimal levels. Their valuation through opportunity costs may not reflect the total economic value, particularly of non-use values.

The Mexican PSAH, one of the largest in its type, is a clear example of the international concern for redesigning effective management policies for watershed resources (Muñoz-Piña et al., 2008). However, there are still a number of challenges for mainstreaming this type of programs. Turner & Daily (2008) summarized three key constraints that need to be overcome before ecosystem services become operational: 1) information failure, where decision-makers lack scale-relevant detailed information on important ecosystem services and their tradeoffs; 2) institutional failure, where property rights and institutions are lacking to ensure legitimacy and equity; 3) market failure, where investments in long-term ecosystem health can be discouraged due to shared benefits and missing prices for public goods.

We have reviewed several cases of non-market valuation that estimated the benefits of watershed services in Mexico (Table 1). Results indicate that there is no significant relationship between WTP, moisture index, and elevation with the two types of values (i.e., consumptive and non-consumptive uses). Considering the low number of cases found in the literature, more research is clearly needed to evaluate the relationship between WTP and the benefits of environmental services, and motivate the interest for creating markets, particularly of non-use values. Here, researchers must incorporate a diversity of geographical areas and services to scale up these markets and incentive programs. They also must employ appropriate valuation tools to tackle the problems associated with reliability of results such as survey design, definition of contingent valuation scenarios, and testing for survey variations of results (Wittington, 2002). More work is necessary to understand the benefits of use and non-use values of watershed services, disseminate the results of pilot projects (success stories), and incorporate all interested sectors of society. This kind of work would increase public's level of awareness and their perception over changes in the provision of environmental services. The participation of government and other institutions (such as landowners represented by *ejidos*[§]) can help to identify critical watersheds for cities, private companies, or non-government organizations. In incipient markets, such as in Mexico, government participation is essential in promoting the type of service most needed for users.

Devising PES programs, such as PSAH, as a rent based on the watershed services preserved (or on the decline in the rate of its loss), necessitates translating ecological functions as measurable and traceable unit of services provided due to the payment (Wunder, 2007). Providing economic incentives to enhance ecosystem service delivery would be ineffective if policies are implemented without tools to differentiate those who alter their management practices in response to the incentive from those free-riders whose behaviors are essentially unaltered (Gilenwater, 2011). To overcome the constraints from the institutional failure, the government must clarify how the service in question and its value will be measured and monitored. We believe that combining market and non-market valuation techniques clarifies the scale of economic distortions due to uncertainty and should help understand the importance of both use and non-use values. The impacts of non-point sources to streamflow can be monitored by establishing a paired-watershed design, which utilizes a calibration period and a control watershed to detect changes in hydrology of a treatment watershed.

[§] Ejidos is one the agrarian reform outcomes generated by the Mexican revolution in the 1920's. As defined by Alcorn and Toledo (1998) an ejido is as an expanse of land, title to which resides in a community of beneficiaries of the Agrarian Reform. Most of the ejidos are collectively owned or cooperatively farmed and the products are also marketed collectively.

Finally, although programs like PSAH are not the panacea to water quality and deforestation problems (Muñoz-Piña et al., 2008), they should be considered in the design of policies for sustainable forest management. PES programs need to reflect the real value of services so providers allocate their maximum effort to internalize the externalities. The real value will come with the use of appropriate economic methods that consider both use and non-use values of watershed services. The involvement of other actors, such as the private sector and non-government organizations, is necessary to improve decision-making and ensure that these kinds of programs achieve their goals.

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15

Market-Based Approaches Toward the Development of Urban Forest Carbon Projects in the United States

Neelam C. Poudyal[1], Jacek P. Siry[1] and J. M. Bowker[2]
[1]*Warnell School of Forestry & Natural Resources, University of Georgia, Athens, GA*
[2]*USDA Forest Service, Southern Research Station, Athens, GA*
USA

1. Introduction

1.1 Urban forestry in the United States: Status and scope

The United States has observed unprecedented urban growth over the last few decades. Nowak et al. (2005) noted that between 1990 and 2000, the share of urban land area in the nation increased from 2.5% to 3.1%. Existing urban areas in the U.S. maintain average tree coverage of 27% (Nowak et al. 2001), and consist of millions of trees along streets and in parks, riparian buffers, and other public areas. Further, Walton and Nowak (2005) predicted that this urban area will continue to expand through 2050, eventually covering up to 8.1% of the country's area. Some of the expected urban development will come at the expense of currently forested areas. This may further the scope of afforestation and subsequent reforestation as part of urban forest management.

Increasing with the area of urban land is the geographical coverage of urban forests. Urban areas nationwide support more than 3.8 billion trees (Nowak et al. 2002), whereas as many as 70 billion trees are estimated to be growing in the urban and urbanizing areas throughout the nation (Bratkovich et al. 2008). A brief look at urban tree inventory data at individual state and city levels confirms that urban trees are a significant component of forest resources at local and regional levels. Table 1 presents canopy coverage and tree inventory data for five selected states and cities to illustrate the relative stocking of urban trees at individual state and municipal level (Nowak et al. 2001). Some of the states have smaller urban canopy coverage, but are densely stocked. Recent urban forest inventories also suggest that there is substantial variation of tree stocks among the United States cities, which ranges from roughly 15 trees per acre in Jersey City, New Jersey to about 113 trees per acre in Atlanta, Georgia (Nowak et al. 2010).

1.2 Issues facing urban forestry in the United States

Sustainable management of forest resources nationwide, regardless of their ownership and management objectives, is facing a number of challenges. Urban forestry is no exception. Sustainable forest management implies conservation and sustainable use of forest resources across all ownerships including urban forests, which are typically managed by local governments (e.g., municipality, city, metropolitan council, town). While population growth

State	Urban tree cover (%)	Urban trees (thousands)	City	Urban tree cover (%)	Urban trees (thousands)
Georgia	55.3	232,906	Atlanta	36.7	9,420
Alabama	48.2	205,847	Boston	22.3	1,180
Ohio	38.3	191,113	Baltimore	21.5	2,600
Florida	18.4	169,587	Oakland	21.0	1,590
Tennessee	43.9	163,783	New York	20.9	5,220

Note: Adopted from Nowak et al. (2010,p. 39)

Table 1. Tree cover and number of trees for selected U.S. states and cities

and development pressures accelerate the loss of wild lands and expansion of urban and suburban areas, protecting and managing trees for a variety of societal and environmental benefits often remains up to local governments. Urban forest management in the United States and elsewhere is facing substantial challenges which threaten the long-term conservation and management of urban tree and park resources. Major factors currently under consideration in the U.S. include the following:

- Disease and pest infestation
- Invasive species
- Wildfires
- Heavy recreational use
- Fragmentation
- Air pollution
- Lack of community participation
- Insufficient funding

Recent forest disturbance research (Holmes et al., 2008) illustrates a range of biological and socio-economic threats to the United States forest systems. A number of invasive stem borers and sap sucking pests such as Emerald Ash Borer, Gypsy Moth, Hemlock Woolly Adelgid have already killed thousands of trees of high amenity and ecological value. A number of exotic plant species including Kudzu, Chinese Privet, and English Ivy have invaded native landscapes in urban parks and roadside plantations. Increasing air pollution due to auto emissions and atmospheric pollution from industrial plants that are often located near urban areas have negatively affected the physiology and ecology of urban landscapes. Furthermore, with rapid population growth, per capita public open space is declining and existing urban forest resources in some areas are being ecologically destroyed due to heavy use (Poudyal et al. 2009). On the other hand, garnering sufficient community participation in urban tree management is challenging due to changing socio-demographics and ethnic heterogeneity in major metropolitan areas. Residents living in a heterogeneous community usually show varying levels of interest towards the maintenance and management of community resources like urban trees, which makes planning and implementation complicated (Gaither et al. 2011).

Another big challenge facing urban forestry right now is insufficient funding. A perennial source of income could greatly contribute to making the urban forest programs financially self-sufficient and sustainable. Indeed, with sufficient funding, local governments could put together efforts aimed at managing many of the other issues listed above. This is why it is important to address the marketability and revenue generating potential of ecosystem services that urban forests provide.

1.3 Towards a financially self-reliant urban forestry

As stated in the preceding section, local government budget problems, and the lack of adequate funding for tree care and maintenance has been considered a major issue in the United States. Mere tree planting along roadsides or on vacant lots within city limits does not define urban forestry. Rather, it involves tree care and maintenance and management (e.g., pruning, clearing, disposal), for which about two-thirds of an urban forest project budget needs to be typically allocated (American Public Works Association, 2007). However, urban forest projects during tough economic times are often overlooked when setting funding and management priorities. Private individuals, albeit usually appreciative of the amenity benefits of urban trees, do not always support the 'tax approach' to finance tree care and management programs. Urban forests bear some characteristics of 'public goods,' meaning that once an output or service is supplied, nobody can be effectively excluded from enjoying it, thereby leading to free-rider problems (Freeman, 2003). Private firms and for-profit organizations have few incentives to provide and maintain such resource. Therefore, if the good or service is to be provided, government must play a major role, either by direct provision or by providing incentives to the private sector.

The sustainable management of urban trees will require continuous funding and a reliable and well-established income generating mechanism at local level. The Urban and Community Forestry Program of United States Department of Agriculture aims at enabling the development of self-sufficient local urban and community forestry programs nationwide. As the provision of a range of public services and basic infrastructure compete for tax revenue, local governments are required to look for external sources of funding to keep their urban forestry programs operating adequately. In many cases, forest management programs, regardless of their location and ownership, will not be sustainable unless they are financially self-sufficient.

Because of the aesthetic and amenity purposes of urban forest management, neither timber neither timber harvesting nor planting of fast-growing cash tree crops are compatible options, or even a debatable alternatives. However, among a wide range of ecosystem services, carbon sequestration is especially promising. Nowak & Crane (2002) estimated that urban forests in the conterminous United States can store 770 million tons of atmospheric carbon, valued at $14.3 billion, assuming conversion to tradable carbon credits and then-current prices. Translating those numbers into annual terms, the United States urban forests absorb nearly 23 million tons of carbon, which can generate $460 million in revenue -- again assuming conversion to tradable carbon credits and concurrent prices. By appropriately managing urban trees and forests for maximum carbon sequestration, cities can collect revenue from selling credits for carbon absorbed and stored in urban trees. Revenue generated in this manner will not strain local tax revenue collections, and will help fund sustainable urban forest management. Given the fact that markets for carbon offset credits have recently emerged, carbon credits become worth investigating.

Federal and state agencies are trying to promote carbon trading in community and urban forestry as evidenced by a series of recently published policy documents. For example, a recently released USDA Forest Service document on open space conservation strategy has listed promotion of market-based approaches to enhance carbon-credit trading as one of the top thirteen priority actions (USDA Forest Service, 2007). Despite its significant potential and increasing policy emphasis, the market for urban forest carbon credits has not been well developed. This outcome in part is a result of the lack of appropriate and broadly accepted market protocols, and the limited understanding of entrepreneurial principles associated with this product. Developing carbon markets will require a thorough understanding of the preferences and expectations of potential buyers per the characteristics, quality, and price of carbon credits. It will also require information about the technical and managerial capacities of the potential sellers to develop carbon offset projects. This chapter highlights some of the findings of a recently completed comprehensive research project in the United States that examined the capacities, interests, and expectations of both the potential sellers and buyers of carbon credits generated from urban forest projects.

2. Objective

The objective of the material presented in this chapter is to address the feasibility of establishing a market for urban forest carbon credits. This will be achieved by assessing the interest of key stakeholders involved in potential market for this output. Stakeholders' perspectives will be discussed in a broader context of making urban forestry a source of carbon credits that will help make it financially self-sufficient and sustainable.

3. Approach

The project started with the identification of key stakeholders in a potential market for urban forest carbon credits. In order to establish a market, potential buyers and sellers of the urban carbon credits must be identified. Given the nature of ownership, local governments and municipalities were considered as the sellers of urban forest credits. A web-based survey was implemented during 2007-2008, contacting urban foresters, arborists and other officials responsible for overseeing their urban forest. Contact details of those officials were obtained from the Society of Municipal Arborists (SMA). The survey questionnaire focused on cities' current urban forest information and management practices, existing stock and available technical and managerial expertise, and interest in participating in an urban forest carbon offset trading program.

Identifying the potential buyers was challenging given that the United States market for forest urban carbon credits has not been well developed. However, because credit buyers in the United States are voluntarily participating in carbon trading rather than complying with mandatory government regulations, existing credit buyers may have unique preferences for credits sourced from specific locations such as urban forests. Therefore, businesses and organizations that are currently participating in carbon markets were identified as the potential buyers of urban forest credits. While many buyers purchase carbon credits from over-the-counter (OTC) market, surveying them is difficult due to the lack of their contact information. For this reason, primary buyers of carbon credits at the Chicago Climate Exchange (CCX), which was the largest carbon trading platform in North America, were surveyed as the potential buyers.

All CCX members and associate members were invited to complete a survey that covered questions regarding their attitudes and perceptions related to climate change, government regulation of greenhouse gas emissions, and their preferences for credits sourced from a variety of carbon project types, including urban forestry. Some of the questions were related to their willingness to purchase urban forest carbon credits and the price they were willing to pay. This survey was conducted during late 2009.

4. Key observations

This section presents some basic statistics and summary of survey responses from the surveys of both the buyers and sellers.

4.1 Seller's survey

From a total of 277 successfully delivered surveys, an adjusted response rate of 54% was achieved. The group of responding municipalities was highly diverse in terms of population size and regional location. About one-fifth of respondents in the sample represented large cities (with population larger than 100,000) and another one-fifth represented small cities (population less than 20,000). Roughly one-third of the respondents were from mid-size cities (with population between 20,000 and 50,000). Respondents from the Northeast region were slightly underrepresented (6%) while other regions (i.e., Midwest, 37%; South, 27%; and West, 31%) were more uniformly represented. Only one-fifth of the respondents were familiar with the Chicago Climate Exchange which, at the time of survey, was the only actively operating carbon trading platform in the country. Further details on respondent's characteristics can be found in Poudyal et al., (2010).

Local government units that responded to the survey indicated that they were maintaining or managing urban forest resources of some sort within their jurisdiction. The exact form of urban forests varied from urban parks, forest patches within city limits to individual trees along streets, roadside tree plantings and protected vegetation along critical riparian buffer areas. More importantly, a clear majority of responding municipalities (63%) had an official designated to oversee the urban tree care and management activities. Similarly, about 56% of the respondents had at least a portion of their forest resource recently inventoried. A similar survey of U.S. cities recently conducted by the United States Conference of Mayors suggested that as much as 55% of cities had a current inventory of urban tree canopy (Nowak et al., 2010).

When asked if local governments were currently participating in any climate change initiatives, respondents identified a number of projects, including remodeling and construction of energy efficient buildings, using alternative fuel vehicles, capturing landfill methane, and planting trees. More importantly, tree planting was the most common initiative undertaken recently (85% of the respondents) to help mitigate climate change (Figure 1). Similarly, about 50% in the sample indicated either using alternative fuel vehicles or constructing/remodeling energy efficient buildings as a recently undertaken initiative to mitigate climate change. It seems that local governments' tree plantation investments in recent years, and perhaps in the near future, would give them an advantage in initiating active-management-based urban forest offset projects. This is necessitated because the already planted stocks do not meet the 'additionality' criterion, unless they are placed under an intensive management regime to boost their carbon sequestration rate.

Prior to reading the questionnaire, approximately one-third of the respondents were familiar with the idea of carbon storage and offset selling. However, very few of the responding municipalities were familiar with existing market platforms like the Chicago Climate Exchange where they could sell their carbon credits. When asked if their city would be willing to participate in a carbon offset selling scheme, 29 out of 150 (roughly 20%) indicated that they were interested or very interested in such a program. On the other hand, 15 respondents (about 2%) indicated that their city was uninterested or not at all interested in carbon trading at this point. An econometric model was estimated to examine factors that

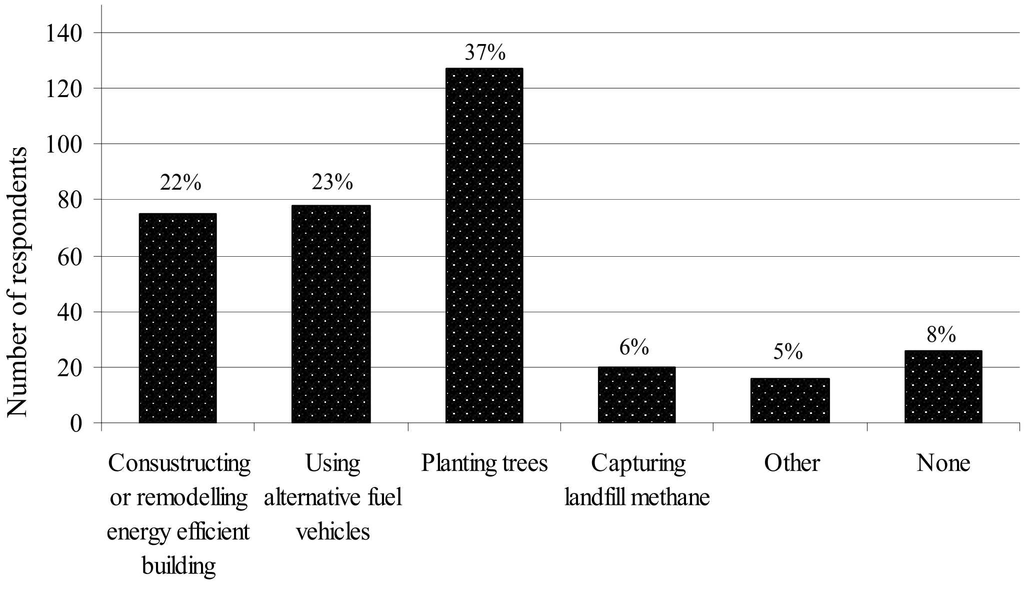

Fig. 1. Number of municipal governments currently participating in various climate change mitigation initiatives

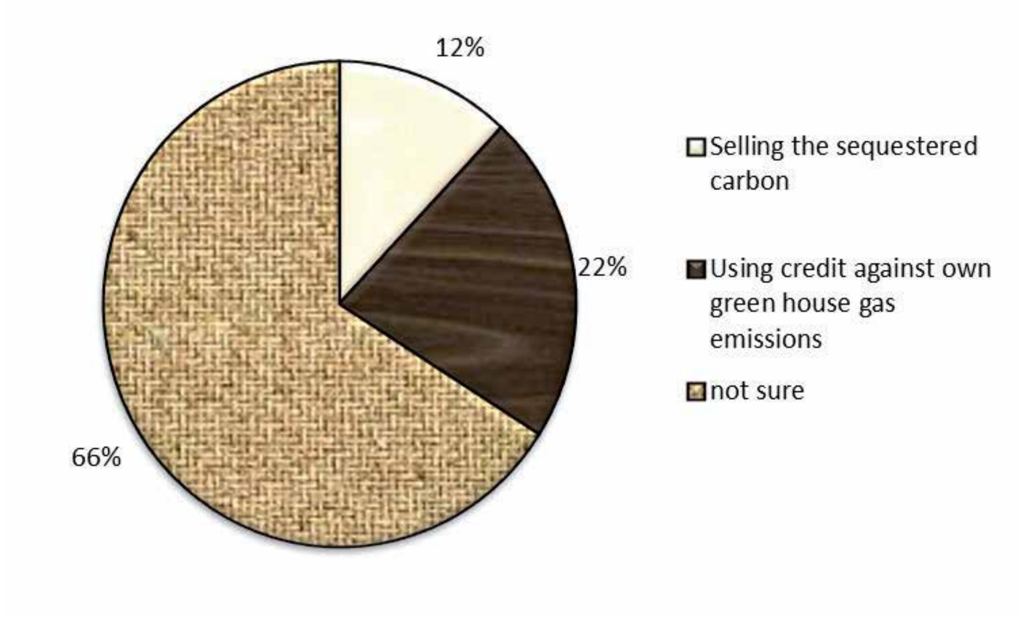

Fig. 2. Local government's plan to utilize the certified carbon credits sourced from their urban forests

influenced respondent's willingness to participate in carbon trading program. Detailed results in Poudyal et al., (2010) indicate that a local government's decision to participate in carbon trading was positively influenced by staff's knowledge of carbon sequestration and familiarity with carbon trading intuitions such as CCX, potential interest of voters, level of urbanization, and a city's need for generating revenue. This observation indicates that along with the increasing need of local governments to generate revenue combined with rising environmental awareness of voters and urban congestion, more local government units will be interested in selling carbon credits though urban forest projects.

Cities which were yet to generate certified offset credits were asked about their plans for using their credits. A majority (66%) were unsure, which is a common response for such a hypothetical question (Figure 2). Among the remaining one-third who had tentative plans regarding the utilization of their certified credits, a significantly higher number of respondents (22%) indicated that they will count the credits against the city government's green house gas emissions rather than selling them to interested buyers (12%). Hence, as the public pressure grows for environmental compliance, and as government units require more credits to offset their own emissions, some local governments may have fewer credits left to sell in the market. How these currently 'unsure' respondents will decide the use of their carbon credits could largely determine whether this may become an issue at all.

4.2 Buyer's survey

From a total of 155 successfully delivered addresses, an adjusted response rate of 41% was achieved. Respondent businesses and organizations (i.e., members and associate members at the CCX) were diverse in terms of their business characteristics such as profit motive, employment and geographical scope of business operations. Slightly more than half in the sample were private or for-profit organizations, whereas just about a quarter of the sample were public or non-governmental organizations. The remaining one–fifth were government institutions. About half of them confined their business operations to the United States. About one-half of all respondents had a target of reducing their greenhouse gas emissions by 5% in the near future. In terms of their carbon trading history, one-half of the sample had been participating in carbon trading for 3 or more years. Respondents, on average, purchased about thirty three thousand metric ton equivalents of carbon dioxide offset credits in the most recent calendar year (i.e., 2008). Further details of respondents' characteristics can be found in Poudyal et al., (2011).

Overall, current buyers of carbon credits in the North American market were found to be pro-environmental and generally supportive of government regulation to control the greenhouse gas emissions. Discussing buyer attributes in detail is beyond the scope of this chapter, but a rigorous analysis of their responses can be found in Poudyal et al. (2011). Buyers were asked to rank credit types by the location of an offset project. Respondents showed much higher preference for credits sourced from local projects than those generated from regional or international projects (Figure 3). Since a number of businesses and organizations interested in offsetting their emissions are located around urban areas, a noticeably higher preference for locally generated credits shows a potentially high value of such credits to buyers.

A more specific question required respondents to rank carbon credits generated from different sources. As Figure 4 shows, buyers clearly placed the highest value on the credits sourced from renewable energy projects. However, their preference for urban forest credits was relatively higher than those sourced from agriculture or methane soil projects. Urban forest credits were found as desirable as rural forestry credits among the credit buyers in the North American market.

Figure 4 suggests that urban forest carbon credits may be fairly competitive in the market. However, whether they will generate more revenue compared to other credit types is a separate question. Buyers' responses in terms of willingness to offer a premium for specific credit types varied substantially among various types of projects. In addition to urban forest credits, respondents were asked to consider offering premiums for credits sourced from three other types of projects: (1) projects promoting nature conservation in developing countries; (2) projects aimed at alleviating poverty in developing countries through carbon payment to forest landowners; and (3) rural forest projects in the United States. While a modest (roughly 15%) number of respondents consistently rejected the idea of paying premium for any kind of carbon credits, many respondents had favored offering a premium for credits sourced from a range of projects. Among the projects listed above, roughly 55% of respondents indicated that they would be willing to pay a premium for urban forest credits. None of the other projects generated a higher level of support or willingness to offer premium. Compared to the current market price of credits for which the source is not generally disclosed, urban forest credits, if known, could draw a significant premium.

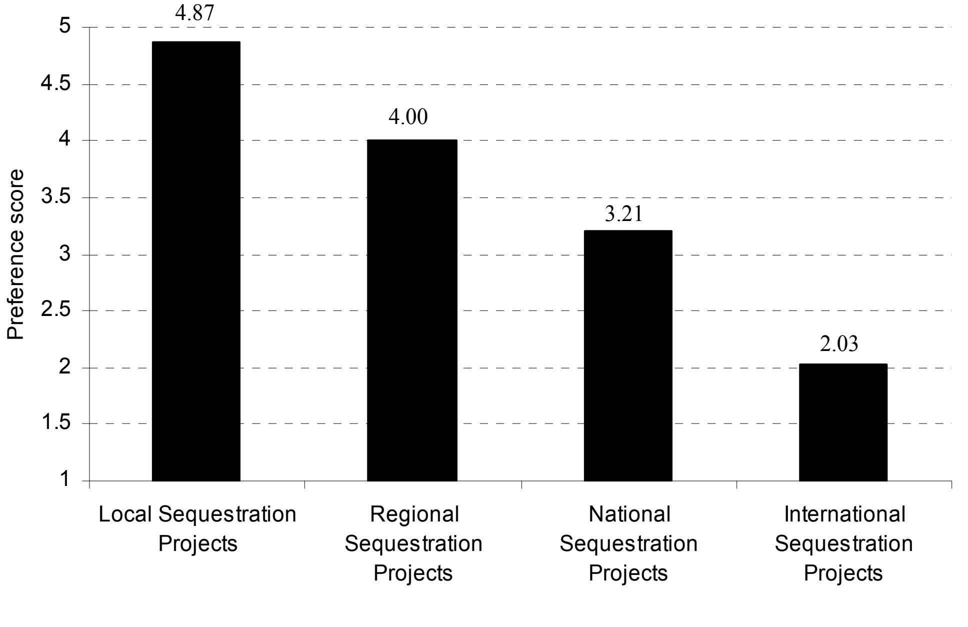

Fig. 3. Buyers' preference for carbon credits by project location

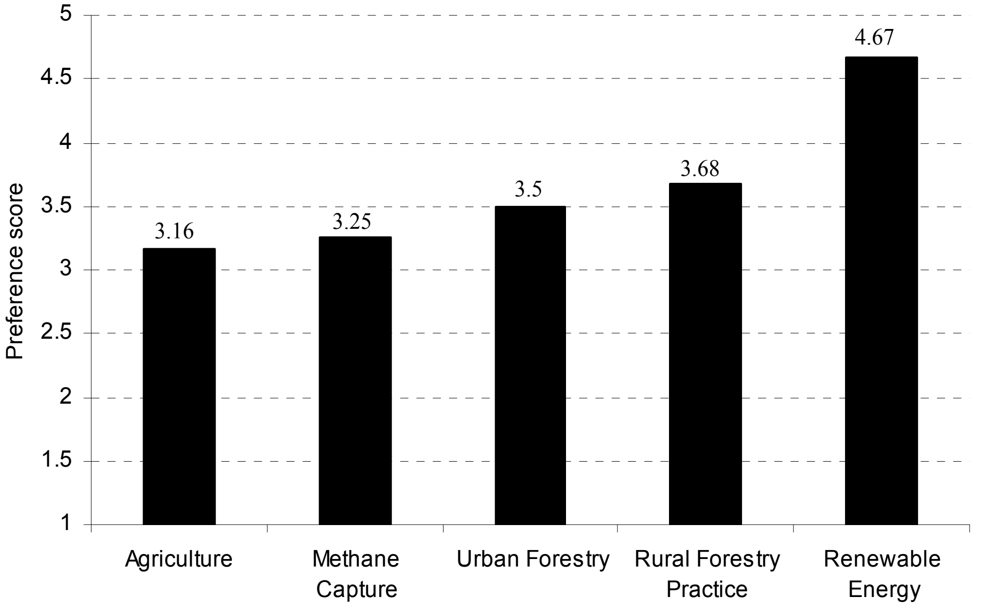

Fig. 4. Buyers' preference for carbon credit by project types

5. Concluding remarks

Buyers and sellers of carbon offsets are interested in this new urban forest output. Urban forest credits are more desirable than other types of credits and buyers are willing to pay a higher price. This will certainly help local governments to be more competitive in the offset market. In fact, this could present an opportunity to be active in localized markets and generate sufficient revenues while preserving urban forests in the long-run and providing a wide range of co-benefits to the society. We argue that promising financial potential provides incentives for local governments to utilize marginal and abandoned industrial lands to increase urban canopy coverage, and to adopt stricter tree management ordinances to boost the carbon storage capacity of public trees. Nowak et al. (2010) noted that about one half of the sample in a recent survey of the United States cities with population of 30,000 or more indicated that expanding tree canopy is their goal and as much as 95% of them have even adopted some sort of tree management ordinance (City Policy Associates, 2008). Current local government initiatives are not necessarily motivated by the need to develop an offset market, but these recent developments when considered together with our results suggest that local governments adopting such policy initiatives may have an advantage with early entrance into the carbon market. Thanks to a number of federal programs that currently offer federal funds to help local communities establish sustainable, clean and green communities, local governments could establish such innovative projects. The Climate Showcase Community Grants of the US Environmental Protection Agency, Sustainable Communities Grants of US Department of Housing and Urban Development, and US Department of Energy's Energy Efficiently and Conservation Block Grants are just a few examples (American Public Works Association, 2007).

However, some research results suggest that the long-term viability of urban forests as a source of carbon credit may be debatable. First, as Nowak et al., (2010) note that increasing tree coverage may increase the potential for storing additional carbon in urban tress, but the maximum tree coverage will entail additional risk and costs, such as wildlife risk along high density residential areas, human-wildlife conflict due to expanded habitat for birds and animal species, and water usage. A long-term strategy for optimizing the social, economic and ecological benefits might be needed to make this effort sustainable. Second, researchers are still debating the net carbon footprint of urban forest projects themselves. Third, our results suggest that more municipalities are likely to use their offset credits against their own emissions targets if they have to comply with a mandatory emission reduction regulations in the future. As more cities sign the Mayors Climate Change Protection Agreement, larger number of carbon offsets will have to be used by cities themselves to improve their green image and meet their constituents' environmental expectations. But again, whether this issue will remain a real concern will largely depend on how the interest and responses of the currently "unsure" group will unfold against increasing demand for carbon credit in future.

Nevertheless, given some of the unique characteristics of urban forest, cities could still produce surplus and market offset credits. Nowak and Crane (2002) argued that by fostering larger trees and by inducing energy savings effects, an urban tree may store four times more carbon than a single tree in a forest stand. However, this assertion should be viewed cautiously as it was derived from a simulation study rather than an empirical measurement of actual sequestration between urban trees and its rural counterparts.

In any case, it seems that there are increasing signs of favorable views and interest among administrators and urban forestry professionals to initiate projects generating carbon offsets. For example, our observations of sellers' motivations and their interests corroborates the findings from a recent survey of members of Society of Municipal Arborists, in which researchers observed that urban forestry professionals are embracing ecosystem services such as climate management, habitat protection, and biodiversity conservation as departmental goals beyond their traditional focus on enhancing property values and protecting utility lines (Young, 2010). It is reasonable to assume that there might be a shift in the way both residents and city managers view the significance and utility of urban forest resources. Part of the enthusiasm and favorable view of professionals probably relies on the availability of practical and user-friendly computer models such as i-Tree or UFore (http://www.itreetools.org) that are useful in quantifying and valuing city forests' offset capacity. All these factors broaden the scope of future urban forest management to include benefits like carbon offset credits.

Key findings highlighted in this chapter provide a holistic view of the market potential and opportunities for making urban forest projects financially self-reliant and more sustainable. Of specific interest to stakeholders are the deeper understanding of the preferences, motivations, and expectations of potential players in the context of establishing markets for urban forest carbon credits. This information could be used to develop new and expand existing market protocols for carbon credits sourced from urban forestry projects.

While this study was based in the United States, the challenge of generating income from urban and community forest projects is likely transferable to other developed countries. Accordingly, many local governments outside the United States are also working to measure and quantify carbon credits generated by their urban forests. While European and Scandinavian countries are already leading in several climate and carbon offset initiatives, some Asian countries (Liu & Li, 2011) and African countries (Stoffberg et al., 2010) have also begun quantification and valuation of carbon sequestration in their urban forests. As more cities and local governments look for ways to make their urban forest projects financially self-sufficient and sustainable, policy implications and recommendations available in this chapter and associated publications should be useful in guiding urban forest management in the United States and beyond.

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16

Implementation of the U.S. Legal, Institutional, and Economic Criterion and Indicators for the 2010 Montreal Process for Sustainable Forest Management

Frederick Cubbage[1], Kathleen McGinley[2], Steverson Moffat[2], Liwei Lin[1] and Guy Robertson[2]
[1]*North Carolina State University Department of Forestry and Environmental Resources*
[2]*U.S. Department of Agriculture, Forest Service*
USA

1. Introduction

At the 1992 United Nations "Earth Summit" in Rio de Janeiro, most of the countries in the world, including the United States, agreed to international accords to protect biodiversity and mitigate climate change. However, they could not agree on a convention for forests, because developing countries wanted to preserve their autonomy and sovereign control of their forest resources, and developed countries would not guarantee them financial support to protect their forests (Humpheys 2006). This failure eventually led to the development of multi-lateral forest agreements and treaties to at least measure and monitor forest sustainability through Sustainable Forest Management Criteria and Indicators (SFM C&I), as well as the movement to create forest certification programs for sustainable forestry.

The creation of multilateral SFM C&I frameworks were a public response to the lack of a binding international agreement on forests; similarly, the development of forest certification systems were a non-state market driven response (Cashore et al. 2004). SFM C&I processes have since been developed to measure and monitor various conditions of forest sustainability at the national or regional level. Forest certification, on the other hand, was developed to also measure SFM, but at the forest management unit level. Many efforts have been made to harmonize national-level SFM C&I with national forest certification efforts, particularly in Europe.

These various efforts at measuring, monitoring, and encouraging SFM address biophysical, economic, and social aspects of forest systems. Many of the C&I efforts have made considerable progress at tracking biophysical characteristics of forests, but the measurement and monitoring of legal and institutional features has developed more slowly. Furthermore, determining whether we are achieving SFM, in general, and if our laws and institutions are helping, in particular, is difficult to ascertain.

In this book chapter, we discuss the development of one criterion of SFM C&I in the United States—the Legal, Institutional, and Economic Criterion and Indicators for the 2010 Montreal Process for Sustainable Forest Management (Criterion 7). This criterion has the greatest number of indicators of the seven Criteria developed by its participating countries, yet most of these are not easily measured or tracked. Thus, this paper describes the approach that we developed in the United States to measure and discuss the legal and institutional indicators for SFM. Criterion 7 and its Indicators have been revised since the U.S. National Report on Sustainable Forests (USDA Forest Service 2011) was issued, and those revisions and suggestions for the next round of C&I reporting also are discussed.

2. International agreements to measure, monitor, and report on SFM

The International Tropical Timber Organization (ITTO) is considered the pioneer of international C&I development, publishing its first framework of C&I for tropical forests in 1992 (Humphreys 2006). That same year, at the United Nations Conference on the Environment and Development (UNCED) in Rio de Janeiro, the non-binding plan of action known as "Agenda 21" and Statement of Forest Principles were signed by more than 178 countries (www.un.org/esa/dsd/agenda21/). These non-binding agreements included a call for the development of international criteria for monitoring national forest resources in all forest types (McDermott et al. 2010).

This combination of the initial ITTO C&I work and the UNCED agreements led to the development of eight regional C&I processes—African Timber Organization, Asia Dry Forests, Dry-Zone Africa, Lepaterique (Central America), Montreal (Non-European Temperate and Boreal), Near East, Pan-European Forest, and Tarapoto (Amazon). The Montreal and Pan-European (now known as the Ministerial Conference on the Protection of Forests in Europe (MCPFE)) Processes were the first to develop C&I frameworks in the mid-1990s, adopting comparable sets of national level C&I for the sustainable management of temperate and boreal forests (The Montreal Process 2009). Today, more than 150 countries are engaged in one or more regional and/or international SFM C&I process (Wijewardana 2008).

As of 2011, the Montreal Process includes 12 member countries—Argentina, Australia, Canada, Chile, China, Japan, Republic of Korea, Mexico, New Zealand, Russian Federation, United States of America, and Uruguay. The multilateral Montreal Process demonstrates that the countries agree on the importance of improving understanding of and measuring progress toward SFM (www.mpci.org). The Montreal Process framework of Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests was adopted initially through the Santiago Declaration in 1995. This covered 7 Criteria and 67 associated specific Indicators. Criteria reflect broad principles or themes that measure forest sustainability; while specific Indicators can be used to determine whether these principles are being achieved. As a whole, the C&I framework serves as a tool for assessing trends in forest condition and management at the national level and as a common framework among countries for describing, monitoring and evaluating progress towards sustainability at both national and international levels (The Montreal Process 1999). This framework has also grown to serve as a standard reference for many national statistics about forests in the U.S., both in the National Report on Sustainable Forests and in separate supplemental reports and web based data bases.

The initial Montreal Process Criteria for forest conservation and management were intended to measure and monitor forest sustainability with the best indicators possible. Sustainability generally refers to the classic 1987 Brundtland Report definition to "provide for the needs of the present generation without compromising the ability of future generations to meet their needs." This definition of sustainability has evolved to include ecological, economic, and social components. The Montreal Process drew from these principles to develop broad criteria that are listed below:

Criterion 1: Conservation of biological diversity
Criterion 2: Maintenance of productive capacity of forest ecosystems
Criterion 3: Maintenance of forest ecosystem health and vitality
Criterion 4: Conservation and maintenance of soil and water resources
Criterion 5: Maintenance of forest contribution to global carbon cycles
Criterion 6: Maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies
Criterion 7: Legal, institutional and economic framework for forest conservation and sustainable management

In general, each Montreal Process member country develops its own approach to measuring and monitoring Indicators, although the Montreal Process Working and Technical Groups facilitate discussions among members and provide technical guidance. In 1997, Montreal Process member countries produced an Approximation Report that provided information on the status of data availability and collection with emphasis on significant implementation issues related to the C&I. The first national reports on the 7 Criteria and 67 Indicators were released in 2003 by participating member countries. These reports varied in the extent and depth to which they covered the suite of C&I. Overall, the 2003 efforts revealed that most countries regularly collected most of the data needed to report conditions with regards to SFM biophysical Indicators, but struggled to address the largely qualitative economic, social, and institutional Indicators in Criterion 7.

Subsequently, the Montreal Process Working Group initiated a process to revise the original C&I, based on experiences with their implementation. At a Montreal Process meeting in Buenos Aires, Argentina in 2007, member countries agreed to revisions of the Indicators associated with the first six Criteria. These Criteria were retained as originally proposed, but some of the Indicators were changed or deleted and new Indicators were added.

In a 2009 meeting in Korea, member countries agreed to revisions of Criterion 7 and its Indicators, including a change to the title of the criterion to “Legal, policy, and institutional framework”, as well as a decrease from 20 to 10 Indicators. For the 2010 reporting cycle, member countries had time to incorporate the revised Indicators for Criteria 1 – 6, but the modified Indicators under Criterion 7 were released too late to be analyzed and integrated with the 2010 country reports. Table 1 summarizes the original and revised Indicators under Criterion 7. The revised 2010 Montreal Process reports had 64 Indicators, and with no further changes, the 2015 reports will have 54 Indicators.

3. Criterion 7 developments

Criterion 7 and its original 20 Indicators are intended to address the crucial question of whether current laws, institutions, and economic structures are adequate to sustainably manage and conserve a nation’s forests. The importance of the legal, institutional and economic framework in forest conservation and sustainable management to the Montreal Process participants is clear given the quantity and breadth of the original Indicators. Most of these Indicators, however, are not amenable to concise quantified measurement. Characterizing national trade policies in terms of their impact on forest sustainability (Indicator 7.3.b), for example, entails an analysis framework and synthesis of information at the level of a full research paper. However, Indicator 7.3.b is but one of 20 Indicators under Criterion 7, and one of 64 within the entire suite of C&I in the 2010 reports.

Table: Initial Criterion 7 Indicators, 1995-2010	**Table: Revised Criterion 7 Indicators, 2011+**
Criterion 7: Legal, Institutional and Economic Framework for Forest Conservation and Sustainable Management	***Criterion 7: Legal, Policy, and Institutional Framework***
7.1 Legal and Policy Framework	
7.1 Extent to which the legal framework (laws, regulations, guidelines) supports the conservation and sustainable management of forests, including the extent to which it:	7.1.a Legislation and policies supporting the sustainable management of forests.
7.1.a Clarifies property rights, provides for appropriate land tenure arrangements, recognizes customary and traditional rights of indigenous people, and provides means of resolving property disputes by due process;	7.1.b. Cross-sectoral policy and programme coordination
7.1.b Provides for periodic forest-related planning, assessment, and policy review that recognizes the range of forest values, including coordination with relevant sectors;	
7.1.c Provides opportunities for public participation in public policy and decision-making related to forests and public access to information;	
7.1.d Encourages best practice codes for forest management;	
7.1.e Provides for the management of forests to conserve special environmental, cultural, social and/or scientific values.	
7.2 Extent to which the institutional framework supports the conservation and sustainable management of forests, including the capacity to:	7.2.a Taxation and other economic strategies that affect the sustainable management of forests.
7.2.a Provide for public involvement activities and public education, awareness and extension programs, and make available forest-related information;	
7.2.b Undertake and implement periodic forest-related planning, assessment, and policy review including cross-sectoral planning and coordination;	
7.2.c Develop and maintain human resource skills across relevant disciplines;	
7.2.d Develop and maintain efficient physical infrastructure to facilitate the supply of forest products and services and support forest management;	
7.2.e Enforce laws, regulations and guidelines	
7.3 Extent to which the economic framework	7.3a Clarity and security of land

(economic policies and measures) supports the conservation and sustainable management of forests through:	and resource tenure and property rights
7.3.a Investment and taxation policies and a regulatory environment which recognize the long-term nature of investments and permit the flow of capital in and out of the forest sector in response to market signals, non-market economic valuations, and public policy decisions in order to meet long-term demands for forest products and services;	7.3.b Enforcement of laws related to forests
7.3.b Non-discriminatory trade policies for forest products	
7.4 Capacity to measure and monitor changes in the conservation and sustainable management of forests, including:	7.4.a Programmes, services, and other resources supporting the sustainable management of forests
7.4.a Availability and extent of up-to-date data, statistics and other information important to measuring or describing indicators associated with criteria 1-7;	7.4.b Development and application of research and technologies for sustainable management of forests
7.4.b Scope, frequency and statistical reliability of forest inventories, assessments, monitoring and other relevant information;	
7.4.c Compatibility with other countries in measuring, monitoring and reporting on indicators	
7.5 Capacity to conduct and apply research and development aimed at improving forest management and delivery of forest goods and services, including:	7.5.a Partnerships to support the sustainable management of forests
7.5.a Development of scientific understanding of forest ecosystem characteristics and functions;	7.5.b Public participation and conflict resolution in forest-related decision making
7.5.b Development of methodologies to measure and integrate environmental and social costs and benefits into markets and public policies, and to reflect forest-related resource depletion or replenishment in national accounting systems;	7.5.c Monitoring, assessment and reporting on progress towards sustainable management of forests
7.5.c New technologies and the capacity to assess the socio-economic consequences associated with the introduction of new technologies;	
7.5.d Enhancement of ability to predict impacts of human intervention on forests;	
7.5.e Ability to predict impacts on forests of possible climate change	

Table 1. Initial and Revised Indicators for Montreal Process Criterion 7: Legal, Institutional and Economic Framework for Forest Conservation and Sustainable Management

The time, financial, and human resources available for the development of each Indicator are limited, as is space for reporting. Moreover, the US National Report on Sustainable Forests is set up to provide concise two-page reports on the importance, status and change in each Indicator, albeit longer technical reports for each Indicator are available in an on-line database. This is not just a matter of limited space for analysis, but also reflects the broad scope for different levels of details and perspectives in the analysis. Comparing the data and numbers in the comparable two-page summaries within and between reports is much easier than comparing two larger associated research papers.

Much of the Indicator development for Criterion 7 in the 2003 National Report relied on separate narrative assessments that identify key concepts and policy components, but which are not regularly collected or monitored and are difficult to update in a consistent fashion. Other Montreal Process Working Group countries had similar results from their efforts to address Criterion 7, largely resulting in revisions of these Indicators to a more qualitative structure. Criterion 7 Indicator assessment and reporting for the 2010 US National Report on Sustainable Forests was seen as an opportunity to bridge between the original and revised Indicators. To achieve this, we developed a new theoretical approach to describe the status and changes in the SFM Indicators under Criterion 7.

In the following sections, we present the approach developed in the U.S. to analyze the original Criterion 7 Indicators and discuss some of the key findings as well as implications for the next assessment of forest sustainability in the U.S. through the Montreal Process.

4. Indicator analytical methods

4.1 Theoretical model

An understanding of the effectiveness of the legal, institutional, and economic framework for forest conservation and sustainable management first requires knowledge of related policy. Policy may be considered a purposive course of action or inaction that an actor or set of actors takes to deal with a problem (Anderson 2010, Hiedenheimer et al. 1983). Policy statements are the formal written outputs of government or private decisions that express the means for implementing policy goals. Laws and regulations are generally the first formal step to policy implementation, which may also include informational, educational, fiscal, market-based and voluntary mechanisms and applications.

In order to understand and analyze the effectiveness of the legal, institutional, and economic framework for forests in the U.S., we drew from theory and research on policy instruments and their analysis (Sterner 2003, Cubbage et al. 2007), "smart regulation" (Gunningham et al. 1998), forest regulatory "rigor" (Cashore and McDermott 2004), and nonstate governance of sustainable forestry (Cashore et al. 2004). Rooted in this literature, McGinley (2008) developed a theoretical model for analyzing the forest policy *structure* and *approach* of government regulation and non-government forest certification in prospective study countries in Latin America. Policy *structure* refers to the level of obligation on the part of individuals and organizations, or government compulsion (voluntary, mandatory) and the policy *approach* refers to the type of policy or practice employed (prescriptive, process-based, performance-based). This model was developed to examine forest policy directives intended for the forest management unit level. Thus, it was modified for use in our analysis of Criterion 7 Indicators for the U.S.

For Criterion 7, the scale of the institutional responses to forest conservation and sustainable management is particularly relevant, since there is wide variation among the 50 U.S. states, not to mention the innumerable local government jurisdictions. Furthermore, many of our U.S. policies and institutions are actually determined by private markets, not government, so this must be considered as part of the analysis of the Criterion 7 Indicators. Therefore, modifications to McGinley's (2008) model included the expansion of policy structure to account for higher level policy mechanisms (non-discretionary/command-and-control; informational/educational; discretionary/voluntary; fiscal/economic; market-based), and adding an approach component for the role of private enterprise in setting institutional policy (Figure 1).

The model displayed in Figure 1 illustrates the range and variation in forest policy mechanisms, approaches, and scales, as characterized by Gunningham et al. (1998); Cashore and McDermott (2004); Cashore et al. (2004); Sterner (2003), and Cubbage et al. (2007). Note that the schema summarized in Figure 1 varies by policy mechanism (often referred to as policy instruments) from command-and-control to market-based, and by approach from prescriptive to private enterprise. To some extent these are continuous scales, not categorical, but we used the categories to make classification and discussion clearer.

We operationalized the theoretical concepts presented in Figure 1 into a "Forest Policy and Governance Matrix" by converting the model into a two-sided classification schema, which we used to classify U.S. SFM laws, institutions, and economic programs under Criterion 7 (Table 2), and to provide comparisons and a meaningful basis for the discussion of each Indicator. This classification schema also fits nicely within the more detailed schema of policy instruments for multi-functional forestry developed by Cubbage et al. (2007), which is presented in Appendix A.

<table>
<tr><th colspan="20">I. Scale</th></tr>
<tr><td colspan="5">National</td><td colspan="5">Regional</td><td colspan="5">State</td><td colspan="5">Local</td></tr>
<tr><th colspan="20">II. Mechanism</th></tr>
<tr><td colspan="4">(A) Non-Discretionary/ Command and Control</td><td colspan="4">(B) Informational/ Educational</td><td colspan="4">(C) Voluntary/ Discretionary</td><td colspan="4">(D) Fiscal/ Economic</td><td colspan="4">(E) Market Based</td></tr>
<tr><th colspan="20">III. Approach</th></tr>
<tr><td colspan="5">Prescriptive</td><td colspan="5">Process or Systems Based</td><td colspan="5">Performance or Outcome Based</td><td colspan="5">Private Enterprise</td></tr>
</table>

Fig. 1. Forest Policy and Governance Matrix by Geographic Scale, Mechanism, and Approach for the United States

In its application, we added specificity to the Matrix by detailing the types of policy instruments that may be employed through the legal, institutional, and economic framework for forest conservation and sustainable management. These include government ownership, Best Management Practices, payments for environmental services, and forest certification, among many others. The typology of specific policy instruments that we reviewed is listed at the bottom of Table 2 and described in detail in the next section.

Mechanism	Scale: National (N), Regional (R), State (S), Local (L)	Approach			
		Prescrip-tive	Process or Systems Based	Perfor-mance or Outcome Based	Private Enter-prise
Non-Discretionary/ Mandatory[a]					
Informational/ Educational[b]					
Discretionary/ Voluntary[c]					
Fiscal/Economic[d]					
Market Based[e]					

Policy Instruments Possible that Could Be Entered in Each Row of the Table 2 Above:
[a] Laws (L), Regulations or Rules (R), International Agreements (I), Government Ownership or Production (G)
[b] Education (E), Technical Assistance (T), Research (R), Protection (P), Analysis and Planning (A)
[c] Best Management Practices (B), Self-regulation (S)
[d] Incentives (I), Subsidies (S), Taxes (T), Payments for Environmental Service (P)
[e] Free enterprise, private market allocation of forest resources (M), or market based instruments and payments, including forest certification (C) wetland banks (W), cap-and-trade (T), conservation easement or transfer of development rights (E)

Table 2. U.S. Forest Policy and Governance Matrix by Geographic Scale, Mechanism, and Approach

The Forest Policy and Governance Matrix developed for the U.S. corresponds well with the general qualitative indicators developed by the Ministerial Conference on the Protection of Forests in Europe (MCPFE 2003). That Process also categorized forest policy instruments into three similar classes: legal/regulatory, financial/economic, or informational. In addition, the MCPFE schema identifies the main policy area, objectives, and relevant institutions. We include most of these factors in our matrix in similar categories, which we termed policy *mechanisms*.

4.2 Using the Matrix model

In our Matrix (Table 2), *approaches* to forest policy and governance include prescriptive, process- or systems based, performance or outcome based, and private enterprise. A *prescriptive* policy identifies a preventive action or prescribes an approved technology to be used in a specific situation. It generally requires little interpretation on part of the duty holder, offers administrative simplicity and ease of enforcement, and is most appropriate for problems where effective solutions are known and where alternative courses of action are undesirable. However, a prescriptive policy may also inhibit innovation or discourage adaptive management (Gunningham et al. 1998). An example of non-discretionary prescriptive standard is: "Cutting intensity does not exceed 60% of the number of trees per species with a diameter at breast height greater than or equal to 60 cm."

A *process-based* policy identifies a particular process or series of steps to be followed in pursuit of a management goal, such as conservation of endangered species habitat or public involvement in National Forest management planning. It typically promotes a more proactive, holistic approach than prescriptive-based policies. Challenges associated with process-based policies include complicated oversight, compliance 'on-paper' rather than on the ground, and an over-reliance on management systems (Gunningham et al. 1998). An example of discretionary process-based is: "Measures should exist to control hunting, capture and collection of plant and animal species." The fact that there is a process developed also has an embedded assumption that a good process leads to good outcomes, which is often but not always the case.

Performance-based policy specifies the management outcome or level of performance that must be met, but does not prescribe the measures for attainment. It allows the duty holder to determine the means to comply, permits innovation, and accommodates changes in technology or organization. Performance-based policies neither specifically promote nor preclude continuous improvement, and enforcement may require intensive monitoring, analysis, and related resources (Gunningham et al. 1998). An example of non-discretionary performance-based policy is: "The rate of forest products harvested does not exceed the rate of resource growth."

Private enterprise relies on voluntary market exchange to allocate many of the forest resources in the world, both in private markets and for allocation of goods and services on public lands. Many new market-based conservation incentives are being developed as well (Cubbage et al. 2007). Market mechanisms represent both a broad philosophical policy approach—letting the private sector develop policies—and a number of mechanisms or instruments, often supported by government. Markets provide flexibility in individual and firm responses and promote innovation, but outcomes are not directly measured or guaranteed. Furthermore, markets do not ensure or even yield equitable outcomes. In many cases in the U.S. and elsewhere, markets for private goods are deemed best to achieve SFM. In addition, many public policy mechanisms, such as the regulation of no net loss of wetlands or payments for permanent easements to protect forest lands, have involved public-private partnerships to achieve SFM.

In addition to the various *approaches* to policy implementation, there are various *mechanisms or policy instruments* that have been employed to protect and sustainably manage forests. These range from mandatory command-and-control regulations or government ownership to reliance on market-based certification or cap-and-trade to allocate forest resources. Intermediate steps between these approaches include information and education, voluntary, and fiscal or incentive mechanisms. Cubbage et al. (2007) outline these approaches in detail (Table 3), and we relied on that schema to identify specific policy *mechanisms* relevant to each SFM Criterion 7 Indicator.

In using the Forest Policy and Governance Matrix displayed in Table 2, the first column identifies the *mechanism* or instrument through which policies and programs are implemented. The second column denotes the *scale* at which policy is developed and applied. The final four columns show the policy *approach* (prescriptive, process-based, performance-based, private enterprise). Specific policy *instruments* are listed in further detail at the bottom of the table. These are used to add further detail to the *approach* columns, with the most prescriptive policies appearing in the upper left of the matrix and the most voluntary appearing in the lower right.

In the matrix, non-discretionary approaches and instruments would include, laws (L), regulations and rules (R), international agreements (I), and government ownership (G). Informational or educational approaches include education (E), technical assistance (T), research (R), protection (P), and analysis and planning (A). Voluntary approaches include best management practices (B), or self-regulation (S), such as forest certification. Fiscal and economic approaches include incentives (I), subsidies (S), taxes (T), or payments for environmental services (P). Last, free market mechanisms include private markets (P), market based systems such as forest certification (C), wetland banks (W), cap-and-trade (T), and conservation easements (E).

The Criterion 7 analysis for the 2010 US National Report on Sustainable Forests (USDA Forest Service 2011) was seen as an opportunity to bridge between past, current, and future assessments of forest laws, institutions, and policies. The Forest Policy and Governance Matrix that we developed for the 2010 National Report can be utilized, along with the in-depth analysis of previous reporting, to track changes in the status of the Criterion 7 Indicators in future assessments.

For the 2003 National Report on Sustainable Forests, Ellefson et al. (2003) performed detailed analyses and summaries of most Criterion 7 Indicators (USDA Forest Service 2004). We utilized these analyses as the basis for the 2010 Criterion 7 update, examining them through the lens of the Forest Policy and Governance Matrix, and identifying and analyzing any changes in the associated legal, institutional, or economic framework. These combined analyses served to generate the 2010 C7 Indicator reports. The Matrix can be used in future assessments to analyze revisions in Criterion 7, and to assess trends in a systematic manner. This approach also provides a framework for comparing U.S. and other Montreal Process countries at a given point in time.

We used the Forest Policy and Governance Matrix to classify the U.S. legal, policy, and economic approaches to forest conservation and management as described by the Indicators under Criterion 7 of the Montreal Process. We first prepared an initial draft characterizing the U.S. approach to each Indicator according to the relevant variables and cells in the matrix. These draft analyses were reviewed by experts in a set of three public workshops on the U.S. SFM C&I, and well as through an extensive open public comment process.

Based on the political science theory, the draft Forest Policy and Governance matrix, and the public meetings and written reviews, we revised the approach slightly, and the application to various indicators moderately. Then we re-analyzed and applied the matrix to each of the 20 legal, institutional, and economic indicators used for the 2010 report.

To illustrate the application of the Forest Policy and Governance Matrix, Appendix B shows the relevant matrix and associated text published in the U.S. National Report on Sustainable Forests 2010 for Indicator 7.1.d - Extent to which the legal framework (laws, regulations, guidelines) supports the conservation and sustainable management of forests, including the extent to which it encourages best practice codes for forest management. A similar set of matrices and text was published for each of the 20 C7 Indicators in the National Report (Moffat et al. 2011). In using the Matrix, note that each Indicator in Criterion 7, and in the National Report, with a couple of exceptions, had a standard two-page write-up. The Criterion 7 template for each Indicator included a description of what the indicator is and why it is important; the Policy and Governance Matrix with the Relevant Approach, Mechanism and Scale cells completed; a statement of what the indicator shows; and what has changed since 2003.

5. Discussion

The summaries from the 2003 National Report and the Forest Policy and Governance Matrix were used as a framework to discuss each Indicator in Criterion 7 and to make more general observations about the U.S. legal and institutional approach to SFM in the 2010 National Report on Sustainable Forests. Conclusions from this theory-based analysis verify that there is a wide variety of legal, institutional, and economic approaches that encourage sustainable forest management in the United States, at all levels of government. Public laws govern public lands, which comprise about one-third of the nation's forests. They dictate management and public involvement through various detailed approaches and mechanisms. Federal and state laws also provide for technical and financial assistance, research, education and planning on private forest lands, but do not prescribe specific actions or standards. However, at the state and local level, in many cases, laws do prescribe specific management actions or standards, such as state forest practice acts, prescribed burning laws, water quality standards, and local zoning regulations.

Federal and state environmental laws protect wildlife and endangered species in forests on all public and private lands. They regulate or promote (best) forest practices to protect water quality, air quality, or other public goods, varying significantly by state. Private markets allocate forest resources on most private forest lands, and even governments use markets for making timber sales, leasing lands for minerals, contracting with private concessionaires for tree planting, or providing recreation services. Many new market based mechanisms, including forest certification, wetland banks, payments for environmental services, conservation easements, and environmental incentives are also being developed to implement sustainable forest management and conservation on private and public lands in the United States.

The effectiveness of the Criteria and Indicators in achieving SFM does rely ultimately on value-based politics, which determine the effectiveness of policies and institutions. The Matrix can enhance the rigor and clarity of this discussion and analysis, help clarify gaps and weaknesses in our institutions, and identify opportunities for improvement in the pursuit of sustainable forest management. Note that the Matrix and associated discussion are intended to summarize the institutional context, not to make policy recommendations. Other parts of the National Report and related subsequent implementation efforts such as that by the Pinchot Institute (Sample et al. 2006) can provide appropriate means of identifying policy responses.

The 2009 Montreal Process modifications to Criterion 7 and its Indicators are expected to better facilitate assessments of the current status and trends in forest laws, institutions, and policies. The revised 10 C7 Indicators to be used in the next round of reporting and beyond stem from the original 20 Indicators, but are more succinct and objective. While they are still more apt to be described qualitatively than measured quantitatively, they are expected to improve measurement and reporting.

Based on the revised 2011 Criterion 7 Indicators, future analysts will be able to summarize existing laws and polices supporting SFM; effects of taxation or incentives; the relative strength of tenure rights; programs and cooperative efforts; public participation; and monitoring and reporting. The Policy and Governance Matrix developed for the 2010 US

National Report on Sustainable Forests can be used to categorize these efforts and subsequent data summaries and legal or policy analyses can add depth to the theoretical framework.

6. Applications

The usefulness of the original 2010 Criterion 7 Indicators and the Forest Policy and Governance Matrix rests on their abilities to condense and convey national, regional, and state information about the policies, laws, and institutions promoting the conservation and sustainable management of U.S. forests. Like the other C&I, Criterion 7 and its Indicators represent an attempt to track the status and trends of forest sustainability for the nation. However, as documented here, the social and legal bases for sustainability are difficult to quantify. We tried to at least make the analysis of this Criterion and its Indicators more consistent and objective through a theoretically-based approach.

Many of the Montreal Process C&I are being used beyond the mere reporting of status and trends, and indeed are leading to program or policy changes and development. Examples include the identification of forest health problems or tracking of fire occurrences and conditions, which then lead to new programmatic responses. The C&I reports for several countries also form the basis for national program development and monitoring, such as for implementation of programs to achieve Reduced Emissions from Degradation and Deforestation (REDD). Comprehensive C&I assessments provide the data and structural platform to design and implement national REDD programs, and in some cases even the structure for forest management level measurement and monitoring.

Description, monitoring, and tracking of the C7 Indicators can also assist in identifying and improving national or state programs for SFM. For example, bilateral trade agreements often require demonstration of sustainable forest practices, which can be evidenced by laws, institutions and policies tracked in Criterion 7, by the U.S. and by our Montreal Process trading partners. Questions about environmental laws and illegal logging addressed in Criterion 7 have become key issues in trade of forest products. These Indicators also are relevant for cross-country comparisons. As the 10 new simplified Criterion Indicators are implemented, the comparison within and among countries will become even more useful. Similarly, so will our Forest Policy and Governance Matrix, or some adaptation of that conceptual framework.

In general, the characterization/categorization of legal and institutional aspects related to SFM as required by Criterion 7 is not a measure of their adequacy for forest conservation and management. Though this same tact (i.e., 'just the data') is taken for the Indicators associated with Criteria 1 through 6, for many of those Indicators the linkage between the data and sustainability can be surmised or, at least, considered. This link is more difficult to make with characterizations of forest policies, laws, and institutions. Perhaps the best use of the C7 analysis is a more explicit and comprehensive categorization of the legal and institutional framework for forests that leads to a better understanding of related policy, law, and institutions, and thereby provides a more complete and transparent basis for assessing the overall framework in regards to actual outcomes and, ultimately, to forest sustainability.

The Criterion 7 indicators do not measure sustainability directly, but address the social components of sustainable development. To some extent, they are the tools used to achieve sustainable forest management. The ecological and even social SFM C&I help directly measure and monitor the status of SFM. Thus in the Montreal Process C&I construct, Criteria 1 to 6 are mostly objective measures of forest sustainability, and Criterion 7 is the assessment of the institutions that help achieve sustainability. The implementation and effectiveness of these laws and institutions will determine how well sustainable forest management is achieved.

Consistently using an analytical tool like the Forest Policy and Governance Matrix in future assessments would facilitate measurements of changes in policy over time, as well as cross-country comparisons, and would potentially permit assessments of related results. The key will be in detecting variance both in terms of matrix coding and in terms of forest impacts and outcomes. These sorts of comparisons (i.e., over time, cross-country) would permit a more substantive characterization of forest policy approaches, to determine, for example, if the U.S. relies more/less on economic incentives to promote SFM than in the past, or more/less than other countries, and given links to other forest measures, may permit associations with changes in forest conservation and management.

7. Conclusion

In the 2010 U.S. National Report on Sustainable Forests, we developed a theory-driven classification scheme to discuss each of the Indicators of SFM in Criterion 7. This approach relied on existing available data and information that was examined through the lens of the Forest Policy and Governance Matrix to measure and monitor legal, institutional, and policy trends related to SFM in the U.S.. The effectiveness of these C&I in achieving SFM does rely ultimately on normative measures about the effectiveness of policies and institutions. Moreover, there is significant debate regarding which forest policies are "best" for achieving SFM, particularly in different countries and biophysical and social contexts. Our analytical approach can enhance the rigor and clarity of this discussion and analysis, help clarify gaps and weaknesses in the legal and institutional framework, and identify opportunities for improvement to achieve SFM.

It is important to note that the intent of Criterion 7 is to provide an objective measurement of the status of laws, policies, and institutions that support forest conservation and management in each country, and perhaps allow comparisons among countries. This is nominally a "positive" or value-free analysis, not a normative assessment designed to make policy recommendations. This is a subtle distinction, since each Indicator reflects specific elements of the value-laden policies that governments choose to enact. Criterion 7 and its Indicators are meant to reveal the status of public policies related to forest conservation and management. Decisions on the adequacy of these public policies in promoting SFM are left to high-level government policy-makers and the relevant legislatures and related interest groups.

In most countries, agency personnel are charged with implementing legislative, executive, and judicial policy decisions, not advocating for changes, even through analytical assessments like those derived from SFM C&I applications. This requires that the C&I be analyzed and reported judiciously in each country report. In fact, the U.S. report primarily focused on the technical findings of the seven Criteria and 64 Indicators, such as forest area trends, forest health issues, carbon storage, forest fragmentation, timber and nontimber market values. And, though the report identifies the "implications of the findings for policy and action", it purposefully does not make policy judgments or recommendations. Nonetheless, an assessment of the status and change in forest policy, law, and institutions through the Criterion 7 Indicators provides information to decision- and policy-makers, who are then authorized to determine if the legal and institutional framework at various levels is adequately addressing forest conservation and sustainability, or if changes should be made, and whether that can be afforded in the current and probably enduring times of budget austerity.

Overall, this new approach to analyzing the 2010 and perhaps future Criterion 7 Indicators provides a better understanding over time of the ways in which policy, legal, and institutional capacity affects forest sustainability. The outcome of this process will determine the extent to which the work on Criterion 7 presented in this document becomes a foundation for future reporting. In any case, the analysis presented here provides a consistent and useful way of characterizing and understanding a broad and complex topic area.

Appendix A. Selected Policy Instruments for Multi-Functional Forestry (Cubbage et al. 2007)

Government Ownership and Planning	Government Regulation	Subsidies & Protection	Education & Research	Private Markets	Private/ Public Project Financing	Private/Public Market Development
Land ownership	Best practices	Plantations	*Education*	*Land Ownership/ Management*	*Financing and grants*	Tradable development rights
National	Harvesting, roads	Timber stand improvement	Professional	Small private	International bank Loans	Conservation easements
Community	Illegal logging	Income tax reduction	Continuing	Industrial	Debt-for-nature swaps	Concession/ extraction quotas
Native/indigenous	Water quality and quantity	Property tax reduction	Public	Timber investment organizations	Venture capital funds	Tradable protection rights
Production	Wildlife, biodiversity	Forest industry & manufacturing	Landowner	Environmental organizations	National forestry funds	Water resource use charges
Timber products	Endangered species	Ecosystem management	Logger and worker	Cooperatives	Policy/ business guarantees	Bioprospecting fees
Nontimber products	Landscape effects	Environmental services	*Research*	*Goods and Services*	Conservation trust funds	Payments for environmental services
Final products	Aesthetics	Fire protection	Federal	Products	Environmental protection funds	Payments for environmental degradation
Services &Amenities	Conversion	Insect & disease protection	State	Services	Securitization	Carbon offset payments
Recreation	Workers/safety/ pay	Invasive species	Forestry schools	Amenities	Grants by philanthropies, NGOs	Clean Development Mechanism
Environmental Services	Community benefits/impacts	Trespass, theft, illegal logging	Other academic disciplines	*Financing*	*Joint management arrangements*	
International Fora and SFM Processes	International trade agreements	Forest law enforcement & governance	Private industry	Banks/loans/ credit	Contracting, leasing, joint	
SFM Criteria & Indicators			Non-government organizations	Foreign direct investment	Build Operate Transfer	
UN Forum on Forests				Forest certification	Build Own Operate	

Appendix B. Verbatim Text of Indicator 7.48 from The National Report on Sustainable Forests, 2010

Indicator 7.48 - Extent to which the legal framework (laws, regulations, guidelines) supports the conservation and sustainable management of forests, including the extent to which it encourages best practice codes for forest management

What is the indicator and why is it important?

Forest management practices that are well designed are fundamental to the sustainability of forest resources. At all levels (stand, landscape, local, regional, national, global), forests depend on the application of forest practices that are capable of ensuring sustained use, management, and protection of important social, economic, and biological values. Well-founded best practice codes, and the forest management practices that comprise them, can ensure sustained forest productivity for market goods; protection of ecological values; and protection of the various social, cultural, and spiritual values offered by forests. They can be among the most important tools for responding to national trends and conditions involving forests.

Policy and Governance Classification

Mechanism	**Scale:** National, Regional, State, Local	**Approach**			
		Prescriptive	**Process or Systems Based**	**Performance or Outcome Based**	**Private Enterprise**
Non-Discretionary/ Mandatory[a]	N,S,L	L,R,G	L,R,G	L,R	
Informational/Educational[b]	N,S,L	P,T,R	E,T,R	E,T,R	
Discretionary/Voluntary[c]	N,S	B	B	B	B,S
Fiscal/Economic[d]					
Market Based[e]	N,S,L				C

[a]Laws (L), Regulations or Rules (R), International Agreements (I), Government Ownership or Production (G)
[b] Education (E), Technical Assistance (T), Research (R), Protection (P), Analysis and Planning (A)
[c] Best Management Practices (B), Self-regulation (S)
[d] Incentives (I), Subsidies (S), Taxes (T), Payments for Environmental Service (P)
[e] Free enterprise, private market allocation of forest resources (M), or market based instruments and payments, including forest certification (C) wetland banks (W), cap-and-trade (T), conservation easement or transfer of development rights (E)

What does the indicator show?

National, state, and local government landowners, as well as all private landowners, have various levels of recommended or required forest best management practices (BMPs). BMPs may be implemented through educational, voluntary guidelines, technical assistance, tax incentives, fiscal incentives, or regulatory approaches.

Ellefson et al. (2005) provide detailed summary of BMPs, albeit for 1992, but it can provide a guide for types of programs now. More than 25 states have regulatory forestry BMPs to protect water quality and to protect landowners from wildfire, insects, and diseases. Almost all states (≥ 45) have educational and technical assistance programs for BMPs about water quality, timber harvesting methods, protecting wildlife and endangered species; and more than 40 have such programs to enhance recreation and aesthetic qualities.

Even states that do not have legally required BMPs often have water quality laws intended to control surface erosion into water bodies of the state, and can be used to enforce BMP compliance. Local governments also implement BMPs for private forest lands, along with other land use controls on development, agriculture, or mining.

BMPs may be prescriptive and mandatory, as required in the state forest practice laws of all the states on the West Coast and many in the Northeast; may require that forest managers and loggers follow specific processes, such as in Virginia; or may be performance or outcome based, ensuring that water quality is protected, such as in North Carolina.

BMPs may cover a variety of practices, such as timber harvest, road construction, fire, site preparation and planting, and insect and disease protection. They also may cover diverse natural resources to be protected, such as water quality, air quality, wildlife, endangered species, or visual impacts.

While BMPs are pervasive, differences of opinion exist about their effectiveness. Almost all forestry compliance surveys have found a high overall rate of compliance for most landowners, but environmental groups contend that many individual practices, such as road-building or wildlife habitat impacts, remain problematical.

The federal government and most states provide detailed technical assistance for information and education about BMPs, as well as research about efficacy, benefits, and costs. The private sector including forest industry, large timberland investors, nonindustrial private forest owners, and forest consultants have been actively involved in development and promotion of BMPs. BMP compliance also is required as part of the standards of all three major forest certification standards in the U.S.—the Sustainable Forestry Initiative, Forest Stewardship Council, and American Tree Farm System.

What has changed since 2003?

Voluntary and regulatory state best management practices for forestry have continued to evolve and improve since 2003. They have been evaluated periodically through on-the-ground effectiveness surveys, and periodically revised. Their scope has been extended in some states to cover more than just timber harvesting and roads to include wildlife, landscape level effects, or aesthetics. Enforcement has increased through inspections, even in states with voluntary BMPs. Several states also have issued separate BMPs for biomass fuel harvesting. And BMPs are now explicitly required under all forest certification systems in the United States.

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Section 7

Decision Making Tools

17

How Timber Harvesting and Biodiversity Are Managed in Uneven-Aged Forests: A Cluster-Sample Econometric Approach

Max Bruciamacchie, Serge Garcia and Anne Stenger
Laboratoire d'Economie Forestière, INRA/AgroParisTech-ENGREF
France

1. Introduction

Nonindustrial private forest (NIPF) landowners have been shown to be more multi-objective by nature than industrial landowners: they give more importance to standing timber and forestland for the amenity values they provide (Newman & Wear, 1993). Among analyses of forest landowner behaviour, the household production framework recognises the benefits associated with forest amenities, as first applied by Binkley (1981). These non-market services are jointly produced with timber and are a determinant in the landowner's utility function.

NIPF landowners comprise close to 70% of land ownership in many U.S. states and significant land holdings throughout Europe (Amacher et al., 2003). In France, almost 75% of the total forestland is privately owned, and 96% of private landowners are nonindustrial. In this article, we investigate the joint production of timber and biodiversity for NIPF landowners using a micro-econometric household production model.

Even though our model is situated within a standard framework where a non-marketed good is jointly produced with timber products, we consider here that biodiversity is not totally disconnected from market strategies. Biodiversity is measured by the diversity of tree species. This assumption is based on the theory of coevolution introduced by Ehrlich & Raven (1964). Coevolution acts as an evolutionary engine and a vehicle for biological diversification. Thus, the diversity of trees or plants may not only tend to increase the diversity of insects and animals, but the converse may also be true. In our model, tree diversity is a determinant of consumer satisfaction and a joint product in the profit-maximisation problem. Tree diversity has an additional impact: it is closely related to some market aspects since the different species have different monetary values. The forest landowner can decide to favour one tree species over another, depending on its value on the market. Conversely, he can make the choice of species diversification to cope with the volatility of timber prices.

We focus on a complete set of forest landowners' decisions in uneven-aged forests where landowners are assumed to value the tree diversity of their forests, as well as timber harvesting. Our economic model is based on the maximisation of their utility that depends on the revenues from harvesting and tree diversity with respect to technological and budgetary constraints. The global objective of the paper is to explain the links between some of the harvest strategies of forest owners, unit price variability and the observed diversity of trees.

More precisely, we analyse: (1) their demand for species diversity and their timber supply; and (2) the joint production of timber and species diversity. Timber supply and amenity demand functions are derived using first-order conditions of the maximisation problem for the landowner.
The behaviour of the forest owner is also strongly dependent on the characteristics of the forest blocks in question. Moreover, his/her harvesting strategy should differ according to the tree species and its value (depending itself on the quality and the diameter of the trees). The issue of heterogeneity in this case is crucial and its omission may result in consequent biases in the estimation stage. The estimation of timber supply and diversity demand is made using a database on uneven-aged forests in France for which several economic and ecological variables are regularly collected. This database typically concerns several forest blocks within which different tree species cohabit. This makes it possible to consider the forest owner within a multi-product framework where each product corresponds to a particular tree species.

2. Methods

2.1 Biodiversity and the economic model

In the literature on NIPF landowners, recent models of timber supply have included non-monetary returns or amenities (Binkley, 1981; Hyberg & Holthausen, 1989; Max & Lehman, 1988; Pattanayak et al., 2003; 2002). The idea is to better understand the trade-off between timber harvesting and amenity benefits.
In this study, we attempt to understand forest owners' decisions concerning timber harvesting and biodiversity. Indeed, different tree species have different monetary values, and the forest landowner has several alternatives: to favour one tree species over another, depending on its market value, or to diversify the tree species in order to cope with the volatility of timber prices.
Our definition of biological diversity may appear to be restrictive due to the sole inclusion of trees (instead of global biodiversity). Nevertheless, tree diversity accounts for a large part of biodiversity: it is generally accepted that the mixture of species is the guarantee of a certain degree of diversity of other living communities (for invertebrates, see Greatorex-Davies et al. (1993), and for bats, see Mayle (1990)). This is the principle of coevolution (Ehrlich & Raven, 1964). The diversity of trees or plants may not only tend to increase the diversity of insects and animals, but the converse may also be true.[1] Even if the extrapolation of tree diversity to global biological diversity is still in debate, this makes it possible to take both biodiversity and strategies on the timber market into account with only one indicator. Furthermore, there is no consensus about the choice of the diversity indicator. This is why several measures were tested in our model.
We, in fact, used two notions, richness and diversity, the latter being the Shannon diversity index computed as $H = -\sum_h p_h \ln p_h$, where h represent a species. Three diversity indices were calculated:

1. Tree richness, designated by $RICH$, is computed as the number of species in the forest compartment. This is the simplest and the most intuitive index used to measure biodiversity. However, this measure strongly depends on the area surveyed.

[1] Many references exist on this topic, see Lähde et al. (1999), Barbier et al. (2008), Schuldt et al. (2008), McDermott & Wood (2009), among others.

2. The Shannon diversity index on the basis of number, designated by $SHANN$, is computed from the number of stems (n_h) with $p_h = \frac{n_h}{\sum_h n_h}$.
3. The Shannon diversity index on the basis of volume, referred to as $SHANV$, is expressed in volume v_h: $p_h = \frac{v_h}{\sum_h v_h}$. The Shannon diversity index based on number is often used by ecologists, but the Shannon diversity index based on volume is more effect for characterising the crown size of different species.

In our model, tree diversity is a determinant of consumer satisfaction and a joint product in the profit-maximisation problem. The landowner i is represented in the framework of the household production function by a utility function that depends on the total income and non-pecuniary attributes:

$$U_i = U(I_i, z_i), \tag{1}$$

where I_i represents the total income of the landowner i and z_i is the forest biodiversity.
The forest landowner faces a budget constraint where the total income is the sum of timber production profit π and exogenous income E:

$$I_i = \pi_{ij} + E_i. \tag{2}$$

The timber profit π_{ij} depends on timber production y_{ij} sold at the price p_{ij}, where the subscript j designates the tree species. The profit function is the difference between the timber revenue and the multi-product cost function related to the production of the (marketable) timber output y_{ij} and the tree diversity z_i conditional on some exogenous variables x_{ij} (including forest capital and ecological variables). It can be written as:

$$\pi_{ij} = p_{ij} \times y_{ij} - C(y_{ij}, z_i, x_{ij}). \tag{3}$$

Timber production y_{ij} and tree diversity z_i are linked by the following transformation function:

$$T(y_{ij}, z_i, x_{ij}) = 0. \tag{4}$$

The forest landowner has to choose the level of decision variables (i.e., y, z and I) that maximizes the utility function (1) subject to constraints (2) and (3). This utility maximisation problem can be solved by substituting these constraints into the utility function. The resolution is done in two steps: the household first selects the optimal level of I and z and then chooses the level of production y. In order to obtain explicit solutions to this problem, we have imposed some simple functional forms on our model. We chose a Cobb-Douglas form for the utility and cost functions. With these particular functional forms and by deriving with respect to y, we obtain the timber supply function that depends on timber price p, non-timber product z and other variables x. Expressing the first-order condition in log-linear form, we find the following timber supply function:

$$\ln y_{ij} = \alpha_0 + \alpha_1 \ln p_{ij} + \alpha_2 \ln z_i + \alpha_3 \ln x_{ij}, \tag{5}$$

where the unknown parameters α are to be estimated. Note that α_1 represents the price elasticity of supply. If α_1 is respectively $<, =$ or $>$ 1 then the supply is price inelastic, unit-elastic or price-elastic. α_2 measures the trade-off between tree harvesting and diversity

in terms of elasticity. If α_2 is negative, there is a substitution effect, whereas a positive sign is synonymous with complementarity.
Entering the equation (5) in the utility function and deriving it with respect to z give us the diversity demand. Transforming it into log-linear form, we have:

$$\ln z_i = \beta_0 + \beta_1 \ln p_{ij} + \beta_2 \ln x_{ij}, \tag{6}$$

where β are the unknown parameters of the demand function to be estimated. β_1 represents the elasticity of diversity demand with respect to timber price. If β_1 is respectively $<, =$ or > 1 then the diversity is inelastic relative to the timber price, unit-elastic or price-elastic.

2.2 The econometric approach

A two-step estimation procedure is implemented by first estimating the diversity demand equation (at the forest level), followed by the timber supply equation in which the predicted value of diversity is entered as a regressor.
Harvest observations collected for different tree species in different forests lead to the use of methods specific to cluster sampling (Wooldridge, 2003). However, the diversity of tree species is observed at the forest compartment level and is therefore cluster-invariant. Supposing that all variables are exogenous, the tree diversity demand equation (6) is estimated by the Ordinary Least Squares (OLS) method.
Cluster specificity is taken into account in the estimation of the timber supply equation (5). The units within each cluster (or forest) may be correlated, whereas independence across clusters is assumed. Specific methods applied to Fixed Effect (FE) and Random Effect (RE) models make it possible to control for unobserved forest heterogeneity while studying the effects of factors that vary across species and forests (e.g., price), and others specific to forests (e.g., tree species diversity). Moulton (1986) shows the consequences of inappropriately using OLS estimation in the presence of random group effects. In particular, he demonstrates that the OLS standard errors that are not adjusted in this case are biased.
Consider the following timber supply cluster-sample equation:

$$y_{ij} = \alpha + X_{ij}\beta + Z_i\gamma + u_{ij}, \qquad i = 1, \ldots, N, \quad j = 1, \ldots, J_i, \tag{7}$$

where i indexes the "cluster" (or forest), j indexes individual observations within the cluster (or tree species). There is a total number of N clusters. The number of species is not the same throughout the different forests i, so that J (i.e., the number of species in the case of balanced data) is indexed by i. The total number of observations is $n = \sum J_i$. Harvest in the forest i of the tree species j is designated by y_{ij}. X_{ij} is a $(1 \times K)$ vector of explanatory variables that vary with respect to i and j. Z_i contains L explanatory variables that only depend on the cluster i. u_{ij} is the error term. α is the constant, and β and γ are the parameter vectors associated with the X and Z to be estimated, respectively.
We consider the following unbalanced one-way error component:[2]

$$u_{ij} = \mu_i + \epsilon_{ij}, \qquad i = 1, \ldots, N, \quad j = 1, \ldots, J_i, \tag{8}$$

[2] Only five species are observed and not within all forests. We can therefore not implement a two-way error component regression model. Moreover, each forest is observed only once since we only have cross-section data.

where μ_i is the cluster specific effect, and ϵ_{ij} represents the remaining unobservables. μ_i and ϵ_{ij} are assumed to be independent and respectively i.i.d. $(0, \sigma_\mu^2)$ and $(0, \sigma_\epsilon^2)$. In matrix form, the one-way cluster model can be written as:

$$\begin{aligned} y &= \alpha\iota_n + X\beta + Z\gamma + u \\ &= R\delta + u, \end{aligned} \tag{9}$$

where $u = R_\mu \mu + \epsilon$, with $R = (\iota_n, X, Z)$ and ι_n a vector of n ones. y and R are of dimensions $n \times 1$ and $n \times (1 + K + L)$. $\delta' = (\alpha', \beta', \gamma')$ is the vector of parameters to be estimated. Finally, $R_\mu = diag(\iota_{J_i})$ with ι_{J_i} is a vector of ones of dimension J_i.
Supposing that all variables are exogenous, the equation can first be estimated by pooled OLS from the unbalanced data. The OLS estimator is trivially given by $\hat{\delta}_{OLS} = (R'R)^{-1}R'y$. It is unbiased and consistent. However, according to the method proposed by Pepper (2002), we use an estimate of the asymptotic variance matrix that is robust to heteroscedasticity and within-cluster correlation of arbitrary forms: $Var(\hat{\delta}_{OLS}) = (R'R)^{-1} \left(\sum_{i=1}^{N} R_i'\hat{u}_i\hat{u}_i'R_i\right)(R'R)^{-1}$, where $\hat{u}_i$ is the $N \times 1$ vector of OLS residuals $(Y_i - \hat{\delta}_{OLS}R_i)$.
Other consistent methods exist (some of which are more efficient), which make it possible to take the presence of unobserved effects in the error term into account. Cluster samples and panel data sets (where i represents individuals and j time periods) can be treated with similar methods (FE and RE models). In our case, the database has the same structure as an unbalanced panel data set. This is why we based our estimation method on the work of Baltagi & Chang (1994).
We can first consider that μ_i represents the unobserved heterogeneity related to the forest, and treat it as a constant parameter to be estimated for each cluster i. If the fixed effects are correlated with the explanatory variables, there is an endogeneity problem that implies a biased estimator of parameters α, β and γ. We can obtain a consistent estimator of β by removing these effects with a suitable transformation (within-group transformation). However, an important drawback is that the parameters (γ) associated with cluster-invariant variables cannot be identified. The within-group transformation matrix for the (unbalanced) cluster-sample case is $Q = diag(E_{J_i})$. $E_{J_i} = I_{J_i} - \frac{\iota_{J_i}\iota_{J_i}'}{J_i}$, where I_{J_i} is an identity matrix of dimension J_i. The Within (or FE) estimator of β is:

$$\hat{\beta}_{FE} = (X'QX)^{-1}X'Qy, \tag{10}$$

under the assumption of non correlation between ϵ and X. A drawback of this method is that γ cannot be identified because the variables Z disappear after within transformation.
In order to take any possible autocorrelation or heteroscedasticity into account, Arellano (1987) proposes the following variance-matrix estimator:

$$Var(\hat{\beta}_{FE}) = (X'QX)^{-1} \left(\sum_{i=1}^{N} QX_i'e_ie_i'QX_i\right)(X'QX)^{-1},$$

with $e_i = Qy - QX\hat{\beta}_{FE}$, which is fully robust.
If the specific effects are assumed to be non-correlated with the explanatory variables, then a random effects (Generalised Least Squares, GLS) estimation can be used. Even if OLS

estimators provide consistent parameters, a heteroscedasticity-consistent variance matrix is necessary. The effect μ_i is now treated as a (cluster-specific) error term and assumed to be i.i.d. $(0, \sigma^2_\mu)$. In this model, we can identify all coefficients related to all variables (including those that are cluster-invariant). Hence, the matrix of explanatory variables is now $R = (\iota_n, X, Z)$. The vector of parameters $\delta' = (\alpha', \beta', \gamma')$ and the variance components $(\sigma^2_\mu, \sigma^2_\epsilon)$ are estimated. The variance-covariance matrix of error terms u is $\Omega \equiv E(uu') = \sigma^2_\epsilon \Sigma$, where $\Sigma = I_n + \rho Z_\mu Z'_\mu$, with I_n an identity matrix of dimension n and $\rho = \frac{\sigma^2_\mu}{\sigma^2_\epsilon}$. The GLS (or RE) estimator is:

$$\hat{\delta}_{RE} = (R'\Omega^{-1}R)^{-1}(R'\Omega^{-1}y). \tag{11}$$

The variance of the RE estimator is: $Var(\hat{\delta}_{RE}) = \sigma^2_\epsilon(R'\Omega^{-1}R)^{-1}$. Several methods of estimation of variance components $(\sigma^2_\mu, \sigma^2_\epsilon)$ exist. However, the solution the most often chosen is the method of Swamy & Arora (1972) by using the Within and Between residuals.

The RE estimator is asymptotically more efficient than pooled OLS under the usual RE assumptions. However, if the cluster effects are correlated with μ_i are correlated with X or Z, this estimator is not consistent. This possible endogeneity can be tested for by performing a Hausman test. The Hausman test statistic is: $(\hat{\beta}_{FE} - \hat{\beta}_{RE})'[Var(\hat{\beta}_{FE}) - Var(\hat{\beta}_{RE})]^{-1}(\hat{\beta}_{FE} - \hat{\beta}_{RE})$. Under the null hypothesis, this statistic has an asymptotic chi-square distribution with a number of degrees of freedom equal to the number of cluster-variant variables (K).

3. Results and discussion

3.1 Data sources and characteristics

The database of the AFI network (*Association Futaie Irrégulière* - Uneven-aged forest network) was used. Uneven-aged forest management is characterised by two fundamental principles: the use of natural dynamics of the ecosystem and the individual treatment of each tree. The first principle implies the use of all tree species on the site: forests are always mixed-species (with variations depending on the acidity of the soils). The second principle means that each tree is examined in order to assess its different functions (e.g., value-added wood, aesthetic aspect). Hence, the decision of tree harvesting or conservation does not result from the stand age but rather from its functionality: Does this tree "pay" for its place? (Bruciamacchie & de Turckheim, 2005). Uneven-aged forest management is practised in numerous forests worldwide with a multitude of variations in terms of species composition and stand structures under local ecological, social and economic constraints.

The AFI network consists of 68 compartments in the northern part of France. The compartment is the management unit for uneven-aged forests (whereas the whole forest is the unit considered for even-aged forests) and corresponds to a block that varies from 5 to 15 ha. One compartment is made up of ten permanent plots that make it possible to monitor the individual growth of approximately 200 trees per compartment. These compartments also make it possible to monitor poles, coppice and regeneration. Some of them are good examples of successful transitions between even-aged and uneven-aged stands. Our sample is made up of forests whose stands are well-balanced in terms of forestry (consistent harvesting), which makes it possible to handle economic data that are uniform on the long term.

As mentioned above, we consider a forest owner who maximises his utility that is a function of total income and diversity. The forest owner decides on the main orientations of his/her forest management (e.g., level of revenues, distribution of species, risks concerning species management). However, we wanted to introduce an important characteristic of forest management into the empirical model: in practice, forests are managed by the "owner/forest manager" pair. Indeed, the owner often delegates the management to a forest manager who implements the owner's choices and can thus have an influence on the harvesting decision and the distribution of species. This is why we include dummies that proxy the identity of the manager (see below).

Among the 68 compartments, 39 were selected because all of the information in all of the categories of variables was available. We classified tree species into five classes: oak, beech, precious broad-leaved trees, other broad-leaved trees and conifers. These five classes of species are not observed in all of the compartments, so that the total number of observations in our sample is 102.[3]

However, the number of species is greater and we compute the diversity for each compartment from the total number of species (varying from 2 to 14 in our sample). As presented above, we calculate three diversity indices. The first index used is tree richness, designated by $RICH$, simply computed as the number of species in the forest compartment. The last two are Shannon diversity indices computed as $H = -\sum_h p_h \ln p_h$, where h represent a species.[4] We compute a Shannon diversity index on the basis of number ($SHANN$) and a Shannon diversity index on the basis of volume ($SHANV$), already defined above.

The variables used in the model are the following:

- Variables observed per compartment and broken down by species: harvested volume (y), unit price (p),[5] stock inventory (INV), volume increment ($VOLINCR$).
- Percentage of quality ($QUAL\%$) and average diameter ($DIAM$) are measured for standing timber.
- At the compartment level, seven dummy variables (from $ST1$ to $ST7$)) are built for seven different ecological conditions ranging from the more basic to the more acid soils. In fact, this set of dummies represents an ecological indicator built from the variables, pH and moisture.[6]
- The type of owners is represented by four dummy variables: institution ($DUMO1$), individual ($DUMO2$), group of owners ($DUMO3$) or joint ownership ($DUMO4$).
- The owner often delegates the management to a forest manager. He/she implements the owner's decisions but can have an influence on the distribution of species. Dummies $DUME1$ to $DUME10$ are used for the manager. $DUME10$ is the remaining sum of managers that are in charge of only one forest compartment.

Descriptive statistics are reported in Table 1.

[3] In a complete data cluster, the number of observations would be 195.

[4] We use two different subscripts in our article. Subscript j refers to the (five) classes of species, whereas h refers to the species alone (the total number of species varies from 2 to 14 in our sample).

[5] Unit price refers to the market price depending on species, diameter and quality. In the empirical model, we use the average unit price, i.e., the unit price for one species in one compartment.

[6] In reality, a more in-depth ecological study would take pH, moisture and altitude into account. There is actually no significant variation in altitude since all forests observed in our sample are located at altitudes below 500 meters.

Variable	Definition	Unit	Mean	Standard deviation	Minimum	Maximum
Dependent						
RICH	Richness index		7.80	2.84	2.00	14.00
SHANV	Shannon index (in volume)		1.17	0.42	0.07	2.16
SHANN	Shannon index (in number)		1.39	0.41	0.00	1.98
Y	Timber harvest	m^3/ha/year	1.02	1.86	0.03	15.03
Independent						
P	Timber price	euros/m^3	31.34	31.53	3.00	170.20
DIAM	Tree diameter	centimeters	30.40	12.17	9.50	58.93
QUAL%	Percentage of quality		0.25	0.16	0.00	0.62
INV	Stock inventory per species	m^3/ha	48.23	72.14	0.75	667.52
INVD	Stock inventory (sum of species)	m^3/ha	131.55	81.71	59.00	677.00
VOLINCR	Volume increment (of stock)		4.31	2.27	1.90	17.90
	Type of owners (Dummies)					
*DUMO*1	Institution		0.1078			
*DUMO*2	Forest owner		0.3333			
*DUMO*3	Group of owners		0.4216			
*DUMO*4	Joint ownership		0.1373			
	Manager (Dummies)					
*DUME*1			0.0686			
*DUME*2			0.1667			
*DUME*3			0.2157			
*DUME*4			0.0980			
*DUME*5			0.1176			
*DUME*6			0.0490			
*DUME*7			0.0490			
*DUME*8			0.0392			
*DUME*9			0.0392			
*DUME*10			0.1568			
	Ecological conditions (Dummies)					
*ST*1	Calcareous		0.2059			
*ST*2	Calcareous clay		0.0490			
*ST*3	Silt and clay		0.1961			
*ST*4	Hydromorphic		0.3333			
*ST*5	Sand		0.1176			
*ST*6	Sandstone		0.0784			
*ST*7	Acid		0.0196			

Table 1. Descriptive statistics, 102 observations

3.2 Estimation results

We first estimate the tree diversity demand equation (6). The diversity of tree species is observed for each forest compartment and is cluster-invariant. Since some explanatory variables vary according to the forest compartment as well as to the species (such as the price), the diversity equation is estimated by a (between-type) OLS method. All variables can be considered as exogenous in this estimation (at least, on the short term). In particular, the price is determined by the market. Hence, there can be no doubt about the direction of the cause-effect relationship. For example, it is the timber price that explains the tree diversity in a compartment and not vice versa.

As mentioned above, there are three different indices to proxy diversity. Three regressions were successfully run with the three different indices as dependent variables. The estimated coefficients are similar. However, the goodness of fit as well as the significance of parameters are better with the logarithm of the number of species (i.e., the richness index). The richness varies as soon as an individual of a new species is added or removed. Shannon indices are preferred by ecologists because they take the richness as well as the distribution of species into account at the same time. However, according to the managers, taking biodiversity into account tends to favour minority species. Estimation results are presented in Table 2.

Variable	Coefficient	s.e	Variable	Coefficient	s.e.
Dependent variable: $\ln z = \ln RICH$					
$Constant$	-4.1585**	1.7642	$DUME6$	-0.3070***	0.0837
$\ln p$	0.3110*	0.1781	$DUME8$	1.2389***	0.2417
$\ln DIAM$	0.7357**	0.2985	$ST1$	2.4252***	0.4362
$QUAL\%$	-2.4432***	0.6085	$ST2$	2.3281***	0.3816
$\ln INVD$	0.0645	0.2351	$ST3$	2.2615***	0.3882
$VOLINCR$	0.4216***	0.1398	$ST4$	1.8120***	0.4080
$DUME2$	0.3974***	0.0970	$ST5$	1.7468***	0.4016
$DUME3$	0.1129	0.1259	$ST6$	1.8774***	0.4155

Notes: $n = 102, N = 39$. Adjusted $R^2 = 0.602$. Heteroscedasticity-consistent s.e.
***: significant at 1%, **: significant at 5%, *: significant at 10%.

Table 2. Demand estimation - OLS method

The overall performance of the demand equation is good since the adjusted R^2 is equal to 0.602. The estimated parameters are all significantly different from zero, except for the stock inventory (i.e., the standing timber per ha) and a dummy variable that proxies a forest expert. However, other variables related to the state or trend of forest capital such as the average diameter of trees, the share of qualitative stand wood and the volume increment of forest are significant in our model. In particular, the negative sign for the coefficient associated with the percentage of quality ($QUAL\%$) has an interesting interpretation. Forests with the highest percentage of quality correspond to ones with the lowest diversity. Since a high percentage of quality increases the revenues over time, this result would mean that in this case, species

diversity is less favoured, showing a trade-off between quality and diversity. Moreover, the coefficient associated with the variable $DIAM$ is significantly positive. In order to favour diversity, some trees were harvested early to diminish natural competition between species.
As expected, the site context has a significant impact on the diversity. Coefficients associated with dummies from $ST1$ to $ST6$ are all significant with positive signs (decreasing from 2.43 to 1.88, respectively) with respect to acid soils ($ST7$), confirming a decrease in richness when the context is acid. Furthermore, the estimated coefficients allow a classification of the site conditions that is in agreement with the observed ecological link between the chemical characteristics of the soil and tree (and flora) diversity.
Some forest managers have a significant positive impact on tree diversity, while other ones have a negative impact that supports a short-term view. The variables for the type of forest owner have been removed because their coefficients were not significantly different from zero. The unit price has a significant and positive influence on the diversity. Its value (0.3110) means that a 10% decrease in timber price implies a 3.11% decrease in tree diversity. This result highlights the effect of timber price on the abandonment of species. For example, in the ecological context where diversity is the highest (14 species in our sample), a 23% decrease in price could lead to the loss of one species. Unit prices for timber are exogenous. However, average unit price (for one species in one compartment) can vary according to the distribution in the stand with respect to its quality and its size. The forest owner can therefore adjust his/her revenues by acting on these variables. One of the principles in uneven-aged forest management is to concentrate the volume increment on the high-quality trees. Hence, low-quality trees are progressively cut and, at the same time, the unit price of standing timber as well as that of harvested timber increase. Once this unit price has increased, forest managers and owners are more inclined to maintain the minority species. The objective is to reduce economic risks by finding an optimal distribution among the different species.
Using the estimates of the demand equation, the fitted value of diversity was computed and used as an explanatory variable in the timber supply equation (5). The use of generated regressors may produce non-consistent estimated standard errors. This is why a vector of regressors was used that includes some or all exogenous variables already in the first regression (Pagan, 1984). This second-step OLS leads, in fact, to a two-stage least squares procedure since the regressors are variables used in the first-step estimation (of the demand equation), and gives correct standard errors. Because the predicted diversity $\widehat{\ln RICH}$ can be approximated by a linear function of the explanatory variables in the demand equation and leads to a problem of collinearity, several exclusion restrictions were used in the supply equation. Some variables that do not appear to be significant to explain harvesting have thus been excluded, including the volume increment of stock ($VOLINCR$) and some dummies that proxy the forest manager. Finally, this estimation procedure is implemented with a robust variance-covariance matrix.
Within and GLS methods (for FE and RE models, respectively) are implemented as described in the econometric method section. They are also conducted in two steps like the OLS method. A Hausman test was then computed to check for the exogeneity of explanatory variables. The value of the statistic is 3.217 (with a P-value of 0.5222) and is below the $\chi^2(4)$ critical value at the 1% level. This result confirms the exogeneity of variables. Hence, the GLS method is the best adapted here for dealing with the cluster feature of our sample. R^2 is equal to 0.606

and indicates a good fitting of our model. Estimation results of (second-step) OLS, Within and GLS methods are reported in Table 3.

	OLS (Pooled)		Within (FE - Fixed Effects)		GLS (RE - Random Effects)	
Variable	Coef.	Robust s.e.	Coef.	Robust s.e.	Coef.	s.e.
Dependent variable: $\ln y$						
$Constant$	-1.2893*	0.7816	-3.3204***	0.2433	-1.3713**	0.5866
$\ln p$	0.5735***	0.0955	0.3467***	0.1089	0.4659***	0.0699
$\ln DIAM$	-0.0368***	0.0089	-0.0424***	0.0072	-0.0387***	0.0066
$QUAL\%$	-1.4822**	0.6484	-1.1813	0.7142	-1.3775***	0.4515
$\ln INV$	0.7178***	0.0771	0.9054***	0.0756	0.8085***	0.0630
$\ln \widehat{RICH}$	-0.8651**	0.3456	–		-0.7797***	0.2610
$DUME2$	-0.8210*	0.4226	–		-1.0129***	0.2387
$DUME3$	-0.3628	0.2651	–		-0.4820**	0.2119
$ST4$	-0.5238**	0.2433	–		-0.5008**	0.1980
$ST5$	-0.8125**	0.3761	–		-0.7456**	0.3018
$\hat{\sigma}^2_\epsilon$					0.5065	
$\hat{\sigma}^2_\mu$					0.2861	
Hausman test (P-value)					3.217 (0.5222)	
R^2	0.613		0.460		0.606	

Notes : n=102, N=39. ***: significant at 1%, **: significant at 5%, *: significant at 10%. Robust s.e. for OLS and FE estimation are respectively computed following Pepper (2002) and Arellano (1987).

Table 3. Supply estimation - Cluster-sample econometric methods

As explained above, OLS is less efficient than GLS since it does not fully take the cluster feature of our sample into account, even if a robust variance-covariance matrix makes it possible to alleviate this problem. OLS coefficients are rather similar to those estimated by specific cluster methods. However, some interest coefficients such as those associated with price and diversity are slightly overestimated. For example, the coefficient associated with the price is 0.57 with OLS, compared to 0.43-0.47 with GLS. For the diversity, it is equal to 0.87, compared to 0.75-0.78 (in absolute value).

The coefficients associated with the variables $QUAL\%$ and $DIAM$ are significantly negative (with estimates of -1.38 and -0.04, respectively). This means that high-quality trees with big diameters are harvested to be sold. Hence, the actual standing timber is characterised by a lower percentage of quality and a lower average diameter. Moreover, it is not surprising to

see that whereas forest managers have a positive impact on tree diversity, this is not the case for timber harvest.
Results also show a positive and significant impact of both timber inventory and unit price. As expected, timber harvest increases with the standing volume of trees. The coefficient associated with the price (or price elasticity of timber supply) is estimated at 0.47, meaning that a 10% increase in price implies a 4.7% increase in harvesting.
The diversity is negatively and significantly correlated to the timber harvest, all things being equal. The estimated coefficient can be directly interpreted as a measure of substitution between tree diversity and the volume of timber harvested. The point estimate is equal to -0.78. This value is rather high. However, based on the standard error estimate, we can reject the hypothesis of a unitary elasticity substitution. An explanation for this negative sign is that when the site context is acid, the forest manager cannot influence the unit timber price interval per species. In this case, under acid soil conditions, the forest manager can only act on timber volume. On the contrary, the basic context allows for a greater variety of species. However, in order to favour all species, the forest manager cannot increase the standing volume and in some cases, may be forced to reduce it. Hence, the forest stock is low on the long term and this trend leads to a lower timber harvest.

4. Conclusion

In this study, a household production approach was used to model the behaviour of the NIPF owner in order to derive the structural econometric equations of timber supply and diversity demand and to estimate substitution and price elasticities. In the empirical application, a definition of diversity was chosen solely on the basis of the number of tree species. This diversity is simple to calculate and positively correlated with the diversity in flora and fauna. Moreover, the richness of data related to harvested species and the cluster-sample methods used in this context make it possible to deal with heterogeneity and variability within clusters. In addition, Within and GLS estimation methods make it possible to test for the possible endogeneity problem of some variables.
This study revealed that diversity demand and timber supply are negatively linked, meaning that an increase in tree diversity will lead to a decrease in timber harvesting. This result confirms that these two forest outputs are substitutes. Estimation also shows that timber price and tree diversity evolve in the same direction: the positive and significant coefficient associated with the timber price in the demand equation indicates that a price decrease has a negative effect on diversity. This result is certainly the consequence of the characteristics of uneven-aged forests and the strategies used to manage them. This could be explained by the fact that a part of the diversity not only procures some satisfaction for the forest owner, but that the price paid for this diversity is a decrease in timber production. Management strategies should therefore be aimed at finding a trade-off between timber production and tree diversity in a given ecological context.

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18

Models to Implement a Sustainable Forest Management – An Overview of the ModisPinaster Model

Teresa Fonseca[1], Bernard Parresol[2],
Carlos Marques[1] and François de Coligny[3]
[1]Department of Forest Sciences and Landscape Architecture (CIFAP), University of Trás-os-Montes e Alto Douro
[2]USDA Forest Service, Southern Research Station
[3]Institut National de la Recherche Agronomique - botAnique et bioInforMatique de l'Architecture des Plantes, INRA-AMAP
[1]Portugal
[2]USA
[3]France

1. Introduction

For a long period, practical recommendations for forest management were based upon experience gained through trial and error experimentation, observation and an understanding of density effects on tree growth within the stand. As stated by Zeide (2008), the limitations of the traditional empirical approach coupled with improvements on modelling efforts led to a change of procedures from forestry to forest science, this being defined by the author, as a new development relying on reasoning to produce the optimal system of forest management aimed at satisfying human needs and preserving nature at the same time (though not at the same place).

Nowadays, the use of mathematical models for tree and stand growth dynamics is the recommended scientific approach to test for alternative management options under a Sustainable Forest Management (SFM) concept and to help solve practical problems such as the appropriate range of stand densities, the thinning prescriptions and rotation ages that allow for a given goal. Assessment of volume and biomass growth, for a given period, or of yield and carbon stock at a point in time becomes a straightforward procedure as long as there are proper equations available, for the species and region of study.

Central to the successful implementation of research findings of sustainable forest management is their efficient transfer from the researcher to the manager (Farrell et al., 2000). In this context, there is a strong need for easily accessible programs to run various and numerous simulations in a convenient and flexible way. There are different possible approaches to build a simulation system, each having advantages and drawbacks. One is to build a specific tool for each model. Development can be fast when the objectives of the model are well defined, its structure remains simple and there is no need for complex outputs and interfaces. This approach nevertheless results in building many prototypes which are generally not very flexible and are difficult to reuse. A second approach is to build one tool around a reusable model and adapt it to different species and situations by changing model parameters. The main drawback remains the limitation to one model with little possible modification. A more interesting option is given by Capsis (Computer-Aided Projection of Strategies In Silviculture, http://www.inra.fr/capsis). The Capsis software is a domain specific tool with a common methodology, but accepting models with different data structures, simulation steps and evolution methods.

This chapter will focus on a forest model developed for maritime pine (*Pinus pinaster* Ait.), the ModisPinaster model (Fonseca, 2004), as a supporting tool for Sustainable Forest Management that is freely available for use in the user-friendly Capsis platform. ModisPinaster (Model with Diameter Distribution for *P. Pinaster*) is a dynamic growth and yield (G&Y) model that applies to pure maritime pine stands. It is constituted by several components allowing the simulation of stand evolution through the rotation period and the simulation of interventions such as thinning and clear-cut. In the mortality component, abiotic and biotic variables are used to determine forest vulnerability to damages from wind and snow. This feature is invaluable in a climate change adaptation scenario. The level of detail of the output is the diameter class, with the diameter distributions being recovered by the 4-parameters Johnson S_B distribution (Johnson, 1949; Fonseca et al., 2009; Parresol et al., 2010). The model can be downloaded from the CAPSIS simulation platform web site.

This chapter has the following structure. Section 2, gives an overview of forest models that have been proposed for maritime pine (2.1), followed by a description of the Capsis platform (2.2) and its current uses. Section 3 is devoted to the description of the structure (3.1) and subcomponent models of the ModisPinaster model (3.2). A portrayal of ModisPinaster simulation capabilities within the Capsis interface is depicted in (3.3). Section 4 presents an example of simulation of three management scenarios for the species, using the CAPSIS environment. One scenario follows the traditional management guidelines in the study area. The second scenario follows density management criteria according to the self-thinning line theory. A third scenario provides a simulation that is compatible with the biodiversity promotion, under a SFM policy. Concluding remarks are presented in Section 5.

2. Use of forest models as a supporting tool for SFM

The sustainable management of the forests has been seen, for a long time, as a sustained yield of wood supply. Thus, it is not unexpected that until recently the great emphasis in the forest research domain has been towards stand volume predictions. Estimates on timber volume production originally come from spacing and thinning trials. The field experiments led to the creation of the first generation of yield tables, by German scientists, in the late 18th to the middle of the 19th century. From the experimental tables to the present G&Y models, different types were developed; although their main uses still are for timber management purposes.

The onset of the multi-functional forest paradigm caused the development of models for other purposes such as: management of non-wood products, the promotion of biodiversity, increasing the social benefits and aesthetic demands. For instance, according to the EU commission study (Nieto & Alexander, 2010), in Europe, 11% of the saproxylic beetle species are currently threatened. The main threat, relates to the loss and decline of their habitat either in relation to logging and wood harvesting in forests or due to a general decline in veteran trees throughout the landscape. Management can help to conserve the biodiversity if done in a sustainable way (e.g. leaving dead wood material in the forest) and promoting the existence of older trees. Attention has also been focused on modelling natural disturbances as they can seriously affect timber production and other forest benefits. It is worthwhile to say that although simulations help to provide management guidelines for the forests, nature is not a virtual forest. Unexpected results might occur in a real forest under a real management process. Critical evaluation of results and adaptive management procedures that take risk into account are therefore advocated when using models for forest growth simulation purposes.

Contemporary studies in modelling extend to the use of the physiologic process based models. Nevertheless, process based models are generally not considered feasible for predicting G&Y under a SFM, as they require a great number of variables and parameters. Some of these variables (e.g. daily meteorological data, radiation absorption, transpiration rate) are hardly accessible or cannot be measured at all and their values have to be guessed, and the models are not capable of providing adequate predictions of tree growth (Zeide, 2008).

An overview of the model approaches for management of European forests is presented by Pretzsch et al. (2008), while a review of models employed to deal with the complexities associated with natural disturbance processes can be seen in Seidl et al. (2011).

Pretzsch et al. (2008), state that the objectives and structure of a model reflect the state of the art of the respective research area at the time, and document the contemporary approach to forest growth prediction. This does not mean that a new model is a better model than the previous existing ones. The selection of one model instead of a past one is strongly dependent on the reliability of the model and on the accuracy of the estimates, which are both dependent on the quality of the supporting data. From the model user's point of view, other useful features are the minimum input requirements; the ability for allowing simulations for diverse combinations of the state variables; and the ease of use. With the development of modelling software, this has become straightforward.

2.1 Available models to help for a SFM of *Pinus pinaster*

The maritime pine (*Pinus pinaster* Ait.), originally from the Mediterranean Basin, is an important conifer species in Portugal, Spain and France occupying an area greater than 3 500 thousand hectares (885 000 ha in Portugal, 1 684 000 ha in Spain and 1 100 000 ha in France). The first evidence of the species in Portugal, dates from the Pleistocenic, about 33 000 years ago. Now it is the leading softwood species in the country covering 27% of the mainland forested area. The major continuous cover is located in the central part of the country, in Mata Nacional de Leiria (MNL) and in the north of the country, in the Tâmega Valley region (TVR). The main uses of the species are related to wood for timber and pulp and to a lesser extent in resin production. On the poor sites it is used in afforestation programs for soil protection. The rotation age usually ranges from 40 to 50 years, although higher rotation ages do occur namely when aiming at high target diameters. To help with the management of the species, several G&Y models have been proposed. The earliest refers to the stand tables by Santos Hall (1931) for the even-aged stands in the MNL in Portugal. The tables by Echeverría & de Pedro (1948) for the pine stands in Pontevedra (Galicia), and the tables by Décourt & Lemoine (1969), for the pine stands in the South-West region, are the first references in Spain and in France, respectively.

Traditionally, the stand tables were based on a reference stand, originally of normal density or fully stocked, and site index, following predetermined average silviculture guidelines. The development of improved analysis methods has allowed for new types of models, which are not restricted to tabular forms or to fixed densities. For Portugal, the most recent include the diameter distribution models PBRAVO (Páscoa, 1987) and ModisPinaster (Fonseca, 2004), Dryads (Gonçalves, 2003) for mixed stands, and PBIRROL for uneven-aged structures (Alegria, 2003). For Spain, there are PINASTER (Soalleiro et al., 1994; Soalleiro, 1995) for even-aged stands in Galicia and the model developed by Diéguez-Aranda et al. (2009), included in the GesMO platform. Orois & Soalleiro (2002) proposed a model that applies to mixed stands. The platform SIMANFOR (Bravo et al., 2010), not being a model, integrates a set of modules for simulating and projecting stand conditions in Central Spain.

Available G&Y models for maritime pine in France, are one whole-stand model named PP1 (Lemoine, 1991) and two distance independent models, Afocelpp (Najar, 1999), and PP3 by B. Lemoine, P. Dreyfus and C. Meredieu (derived from Lemoine, 1991; Salas-Gonzalez et al., 2001) (http://www.inra.fr/capsis/models). The latter model presents several functionalities such as a dead wood estimation (Brin et al., 2008), windthrow risk through a connection to ForestGales (Cucchi et al., 2005), and wood quality assessment (Bouffier et al., 2009). The three French models and ModisPinaster are currently integrated in the Capsis platform.

2.2 The CAPSIS 4 platform

Software development can be very time-consuming and expensive. The Capsis project has undergone continuous development in France since 1994 with the aim to simulate the consequences of silvicultural treatments based on scientific knowledge, and to build an integration platform for forestry growth and yield models. One of the objectives of Capsis is to share this effort by organising the work around a small number of software developers, who concentrate on the technical aspects of the software and common tools, and modellers who concentrate on specific modules related to the scientific core of their models. Capsis relies on the JAVA environment. This choice of an object-oriented language promotes easier adaptation of common ancestor objects by the modellers, as well as modularity. Capsis has an open software architecture around a stable kernel, augmented with applicative and technical libraries. Different models are integrated in it as many modules, and various tools can be added at any time within flexible extensions. This "platform" runs either in interactive context to explore possibilities or in batch mode to run long or repetitive simulations (Dufour-Kowalski et al., 2012). Initially developed for forest modellers, the range of Capsis end-users very quickly expanded to a large number of stakeholders (Figure 1). It is used in an increasing number of applications for forest management and training. Stakeholder aims are now: to contribute to the development of models and test their sensitivity to model parameters by simulating managers' actions, to share tools and methods, to compare results of different models, to transfer models to managers and to develop training material.

Every component developed inside Capsis, except the Capsis modules (i.e. the model implementations) can be freely distributed under a free license (Lesser General Public Licence), meaning that the core application, including all the extensions, can be used by anyone.

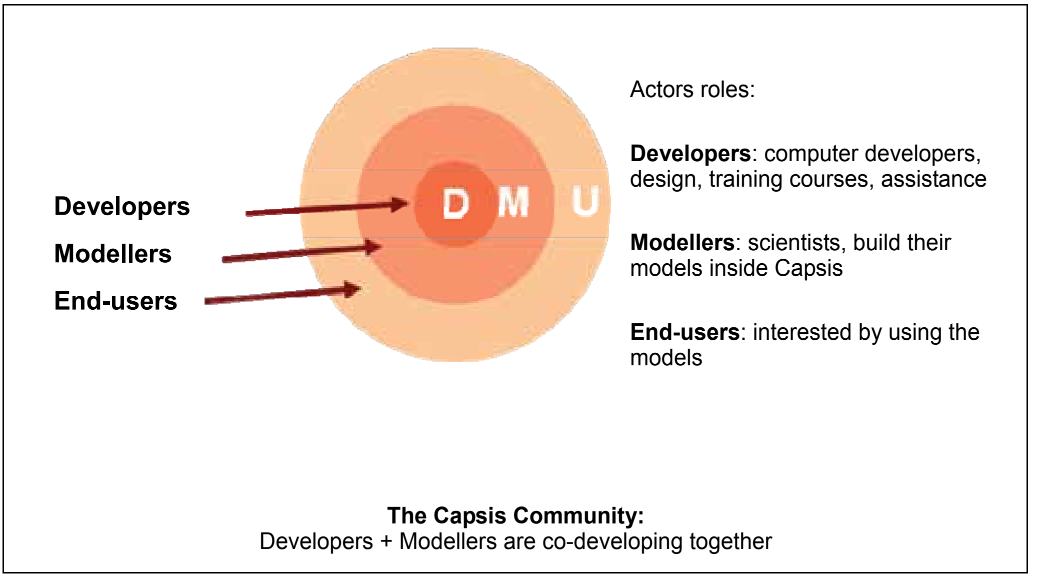

Fig. 1. The Capsis project organization.

Concerning the modules (i.e. the models), the authors decide on the license they wish, free or not, and choose the way to distribute them outside the community. This framework relies on mutual confidence and favours multiple public and private partnerships. The current release of Capsis (Capsis 4) now contains more than 50 forest growth or dynamics models of different types: distance-independent tree models and individual tree models, as well as mixed models, developed by modellers worldwide (http://www.inra.fr/capsis/models). In addition, models within Capsis can be connected with other software (GIS, visualisation, architectural models, de Coligny, 2007). The potentialities of Capsis enhance the use of forest models for SFM through the ease of sharing the models with forest managers without charges and permitting the analysis of different scenarios. Simulations are easy to run and users can utilize and test different silvicultural scenarios for a sustainable forest management. It is used in an increasing number of applications for forest management and training. Capsis is particularly useful in situations where observation and experimentation is difficult. Many local and regional French National Forest Service offices have used the Capsis software to help define management operations for implementation by the field services (Meredieu et al., 2009). For example the silviculture handbook for the French northern Alps, applied to both public and private forests, is based on Capsis simulations, especially for mixed fir-spruce forests (Gauquelin & Courbaud, 2006).

3. A case study: the ModisPinaster model

Data used in the model development come from a large database on maritime pine (Data_Pinaster) created and maintained over the last two decades at the Department of Forest Sciences and Landscape Architecture of the University of Trás-os-Montes e Alto Douro. Data come from temporary and permanent plots and were collected in northern Portugal, more precisely in stands located in the Tâmega Valley (latitude range: 41° 15′N –

41º52′N; longitude range: 7º 20′ W – 8º 00′ W). The model addresses forest growth and yield, risks (wind related) and management procedures such as thinning and harvesting. Since its development, efforts have been made to promote its dissemination to potential users and to allow for a more effective use under a SFM vision. The implementation of ModisPinaster within the Capsis platform has improved its use as a tool for sustainable management of maritime pine forests. Details are given in the following sections.

3.1 Description of Modispinaster

ModisPinaster is constituted by six components: (i) dominant height growth; (ii) basal area growth; (iii) tree mortality; (iv) diameter distribution; (v) thinning algorithm and (vi) output functions for volume, biomass and carbon content assessment by diameter classes. The relationship among the components is shown in Figure 2.

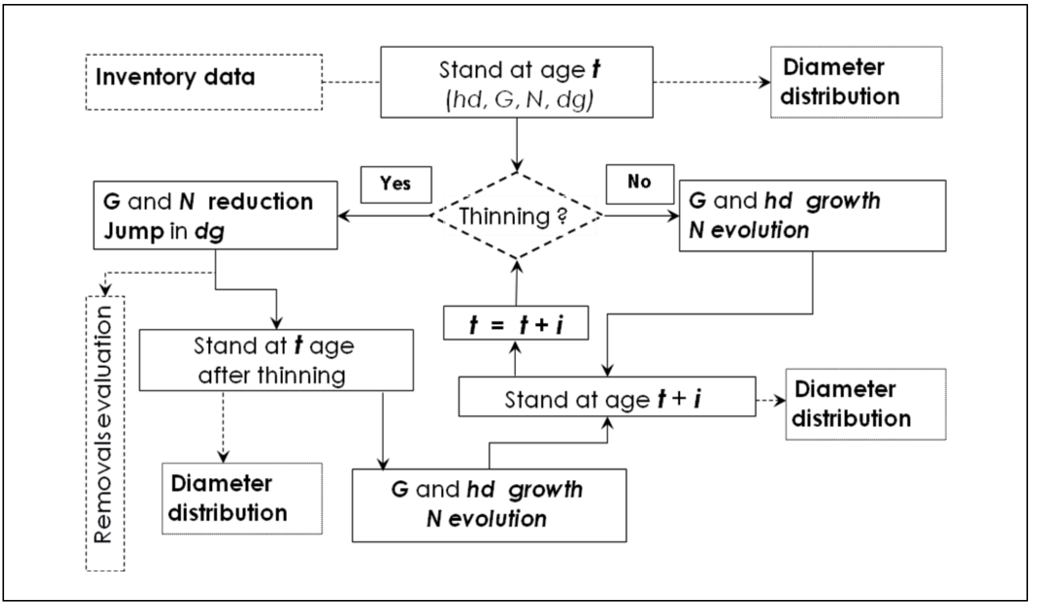

Fig. 2. Simplified structure of ModisPinaster.

The model initiates from a calibration point that requires data variables easily obtained from current inventories:

- Stand variables: stand age (*t*, yrs), the average height (m) of the 100 thickest trees per ha (*hd*, m) or the site index value (*SI*, m, base age 35 years), basal area (*G*, m^2 ha^{-1}), number of trees per hectare (*N*, trees ha^{-1}) and the average diameter of the dominant trees (*dd*, cm).
- Site variables: terrain slope (*Inc*, º.) and terrain direction (*Exp*, º.).
- Stand nature: specified as a qualitative variable (homogeneous or heterogeneous in terms of the uniformity of trees' age) or assessed through the diameter distribution in terms of the number of classes (5-cm wide) and the standard deviation of the diameters (*sd*, cm).
- Management variables (optional): number of trees recently cut (if any) (N_t, trees ha^{-1}).
- Historical details on tree mortality (optional): presence or absence of dead trees (0/1).

Other optional variables include: the median ($d_{0.50}$, cm), the average ($\bar{d}$, cm) and the minimum value of the diameter distribution (d_{min}, cm).

The input data coupled with the model components allow representing the stand growth and the management practices that are typical to the species, including the simulation of mechanical and selective thinnings and clear-felling. The maximum age allowed for the rotation term is 65 years. The minimum scale level admitted for prediction is the year.

Dominant height growth, for a given site index, is estimated using Marques (1987) model (equation 1). Site index value is calculated from Marques (1987, 1991) *SI* model (equation 2).

$$\widehat{hd} = e^{4.04764 - 8.75819t^{-0.56087}} + 1.19874\left(1 - e^{-0.081t}\right)^{2.99578}\left(SI - 17.38\right) \tag{1}$$

$$\begin{aligned} SI = 17.38 - &\left(e^{4.04764-8.75819t^{-0.56087}} - hd\right)\times \\ &\left(0.865685 - 0.00804747t + 0.000994305t^2 - 0.0000187066t^3\right) \end{aligned} \tag{2}$$

Basal area at the projection age is estimated with equation 3. The growth model was originally presented in Svetz & Zeide (1996), and was refitted by T. Fonseca, after Fonseca (2004), using the Data_Pinaster dataset.

$$\begin{aligned} \hat{G}_2 = &\left[G_1^{\,0.4090} + 7.4949e^{-0.0333t_1}\left(1 - e^{-0.0333(t_2-t_1)}\right)\right]^{1/0.4090} \\ &e^{-0.8427(N_1-N_2)/N_2} \end{aligned} \tag{3}$$

In equation (1), *SI* refers to site index, defined as the stand dominant height (*hd*) at the reference age of 35 years whereas, in equation (3), G_i and N_i refer to the stand basal area and to the number of trees at age t_i, respectively (i = 1, 2 for actual and projection age, respectively). The other variables in equations 1-3 were already defined.

Evaluation of tree mortality is a two-phase process. In the first phase the model estimates the probability of mortality to occur during the projection period. In a second phase the number of survival trees is calculated for the projection age and then it is adjusted by the probability of mortality to occur.

Probability of mortality is predicted by two equations developed by Fonseca (2004), according to the major influences: wind (equation 4) and other causes (equation 5), these being mainly related to competition effects.

$$\hat{p}_1 = \left[1+\exp\left(-\begin{pmatrix} -30.4753 + 0.3725Qhdc + 21.1705RS_b \times BRS \\ +4.2303BSExp + 0.1758Incl \times BF_1 + 5.5347BF_2 \end{pmatrix}\right)\right]^{-1} \tag{4}$$

$$\hat{p}_{OC} = \left[1 + \exp\left(-\begin{pmatrix} -8.5235 \; - 0.3822N_t / 100 \; + \\ (t_2 - t_1)\left(2.1449 \; + 1.5768BMA \times C \; + 39.2942BAMPD / dg_{MAX}\right) \end{pmatrix}\right)\right]^{-1} \tag{5}$$

In equation 4 and equation 5, the variable *Qhdc* and the binaries BF_1 and BF_2 are related to the stability of the stand; the variable RS_b refer to relative spacing before thinning with *BRS* being a binary that makes a distinction of the average space conditions between the trees for the current stand; dg_{MAX} is the maximum values of tree diameter allowed according to the self-thinning line for the species (Luis & Fonseca, 2004) and *BAMPD* is a binary variable used to differentiate the stands according to the proximity to the self-thinning line. These variables and the ones related to the occurrence of recent mortality (*BMA*) and to the intrinsic risk of damages occurrence, due to the terrain direction (*BSExp*) are defined as follows:

$Qhdc$ = $100 \times (hd - 1.30\text{m})/dd$
BF_1 = 1 if $Qhdc > 48$; $BF_1 = 0$, otherwise.
BF_2 = 1 if $100 \times hd/dd \leq 54$; $BF_2 = 0$, otherwise.
RS = $100/(\sqrt{N} \times hd)$
RS_b = $100/(\sqrt{N_b} \times hd)$
BRS = 1 if $RS \leq 0.20$; $BRS = 0$, otherwise.
C = 1 for recent thinning; $C = 0$, otherwise.
dg_{MAX} = $25\,(1859/N)^{1/1.897}$
$BAMPD$ = 1 if $(dg_{MAX} - dg) \leq 17.5$cm; $BAMPD = 0$, otherwise.
BMA = 1 for recent mortality; $BMA = 0$, otherwise.
$BSExp$ = 1 if direction (º) belongs to]60, 120] ∪]180, 240] ∪]300, 360]; $BSExp = 0$, otherwise.

A join probability of mortality for the period t_2-t_1 is estimated as

$$\hat{p} = \hat{p}_{\mathrm{I}} + \hat{p}_{\mathrm{OC}} - \hat{p}_{\mathrm{I}}\hat{p}_{\mathrm{OC}} \tag{6}$$

An initial assessment of the living trees at projection age t_2 is given by the survival model (equation 7), developed after Huang et al. (2001).

$$\hat{N}_2 = N_1 \left(\frac{1 + \exp\left[-5.2560293 + 1.81990161 \ln\left(1 + t_1\right) - 0.1532847SI + 0.86246466BE\right]}{1 + \exp\left[-5.2560293 + 1.81990161 \ln\left(1 + t_2\right) - 0.1532847SI + 0.86246466BE\right]} \right) \tag{7}$$

In equation 7 *BE* is a binary variable that characterizes the stand horizontal structure based on the heterogeneity of tree diameters. The stand is homogeneous in composition ($BE = 0$) for diameter distributions not exceeding 25 cm in range and 5.5 cm in standard deviation; otherwise it is heterogeneous ($BE = 1$).

The number of trees is adjusted according to equation 8.

$$\hat{N}_{2aj} = N_1 - \hat{p}(N_1 - \hat{N}_2) \tag{8}$$

ModisPinaster is a distribution model that presents output information at the detailed level of the diameter class. The diameter distribution is modelled by Johnson's S_B (Johnson, 1949), (equation 9).

$$f(d) = \frac{\delta\lambda}{\sqrt{2\pi}\left(d-\xi\right)\left(\xi+\lambda-d\right)} \exp\left(-\frac{1}{2}\left[\gamma + \delta \ln\left(\frac{d-\xi}{\xi+\lambda-d}\right)\right]^2\right) \quad \xi < d < \xi + \lambda \tag{9}$$

$$= 0, \text{ otherwise}$$

where λ, $\delta > 0$, $-\infty < \xi < \infty$, $-\infty < \gamma < \infty$; λ is a range; ξ is a location parameter (lower bound), δ and γ are shape parameters, $\gamma = 0$ indicating symmetry.

The algorithm used to incorporate the S_B distribution in ModisPinaster was based on the parameter recovery method in combination with the parameter prediction proposed by Parresol (2003). Briefly, in his approach, Parresol assumed the minimum location parameter was pre-specified (set to 0.8 of minimum diameter in ModisPinaster). The range and two shape parameters were then recovered from the median and the first two noncentral moments of the diameter distribution (average diameter and quadratic mean diameter). A complete SAS code for the procedure is available in Parresol et al. (2010). The evolution of the stand structure in terms of diameter class distribution is provided for each year of the simulation and whenever an intervention is simulated.

The procedure to represent the stand structure after a thinning requires previous information on diameter distribution (actual or simulated using the S_B distribution). Trees to be removed from the diameter distribution are identified with a thinning algorithm (Alder, 1979). The procedure assumes a probability of survival to cut proportional to a tree's size, $l(F) = Fc$, with c given by N_t / N_a. The number of trees that remain in the diameter class j (N_{ja}) is then calculated as:

$$N_{ja} = N_b L\left[F(d_j)^{1/L} - F(d_{j-1})^{1/L} \right] \tag{10}$$

In equation 10, F represents the initial probability density function (PDF) and L corresponds to the proportion of the standing trees, comparing to the number of trees before thinning ($L = N_a/N_b$). The diameter distribution for the removed trees is obtained by subtraction.

The stand basal area after thinning is calculated with equation 11. This equation was refitted by T. Fonseca, after Fonseca (2004), using the Data_Pinaster dataset.

$$G_a = G_b \left(N_a / N_b \right)^{2.4979 N_b^{-0.1951}} \tag{11}$$

Stand variables after thinning are used to recalibrate stand variables at age t. That is, N becomes equal to N_a and G becomes equal to G_a.

Auxiliary functions, not presented here, were developed to allow for the estimation of the optional input variables. These include functions for the median, the average, the minimum and the maximum values of the diameter distribution, all required for the S_B recovery procedure. Additional functions appended to ModisPinaster refer to the height-diameter relationships, and to tree equations to calculate the volume and biomass content. At present, the model uses the height-diameter relationships by Almeida (1999), the volume equations from Fonte (2000), and the biomass equations from Lopes (2005).

3.2 ModisPinaster within the Capsis interface

The potentialities of Capsis enhance the use of ModisPinaster for SFM through the ease of permitting the analysis of different scenarios in a friendly environment. The extended outputs provide diverse information of stand growth and structure as well as of thinning intervention.

The simulations are easy to run and the users can utilize and test different silvicultural scenarios for a better choice under a sustainable forest management policy.

Figure 3 presents the ModisPinaster initializing scenario for a sampled stand (Stand_1) with the minimum input variables required for simulation purposes. By default, the stand is assumed to be of homogeneous structure (even-aged). For the example, a merchantability limit of a top diameter of 7 cm is specified for volume estimates. The state variables of the stands, age (t, yrs), dominant height (hd, m), number of trees (N, trees ha^{-1}), and basal area (G, m^2 ha^{-1}), are projected on an annual basis until the end of the growth period, using the set of equations 1-8. A portrait of the Evolution dialog and of the appended output options is depicted in Figure 4. An automatic management procedure, based on the self-thinning theory is available in the Management/Self-thinning dialog. By default, the limits specified for the Stand Density Index (*SDI*, Reineke, 1933; Luis & Fonseca, 2004) are with respect to a stand that grows under high values of density. The occurrence of mortality by competition is expected whenever *SDI* reaches the threshold of 60%. The limits can be modified by the user, according to pre-determined management guidelines.

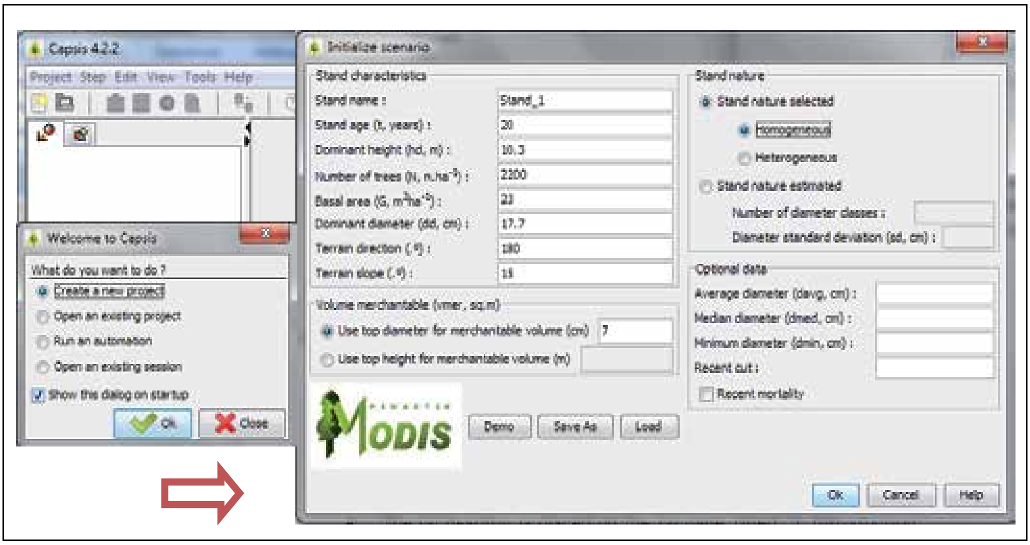

Fig. 3. The ModisPinaster input dialog in the Capsis platform.

Selected outputs shown in Figure 4 refer to a stand table for the 5-year evolution period (bottom left) and to the number of trees, per diameter class, at the initial stand age (20 yrs) and at the end of the simulation (25 yrs) (bottom right). The diameter distribution is presented for each selected scene (year) according to the methodology described in section 3.1 and detailed in Parresol et al. (2010).

Major improvements for the thinning simulation ability of ModisPinaster are available in the Capsis thinning interface (see Figure 5, to the left). Users can decide the thinning prescription based on the total number of trees to remove, or in density regulation rules, according to the stand density index (*SDI*) criteria based on the self-thinning theory, or according to the Wilson spacing factor, $Fw = 100N^{-0.5}hd$, where the variables N and hd had already been defined. In each of these cases, the selection of the trees to remove from the stand is made according to the algorithm of Alder, described in section 3.1, which assures a probability of a tree to survive to cut being proportional to the tree´s size. Alternatively, the user can perform a selective thinning using the Capsis interactive diagram. With this option, the trees are cut by action on an interactive diameter distribution diagram. Figure 5 presents the intervention dialog with a simulation of a thinning at 20 years of age. In the example, a Wilson spacing factor of 0.23 was specified for the thinning criteria. At the bottom of the dialog, a summary of the variables N, G and dg, is presented for the initial and for the residual stand, and for the thinned material. Results obtained for a growth series of 10 years, after the thinning, are presented to the right of the figure: a stand table in an annual basis, and the disaggregation by diameter classes of the total and merchantable volumes and of carbon content, per component, at the end age.

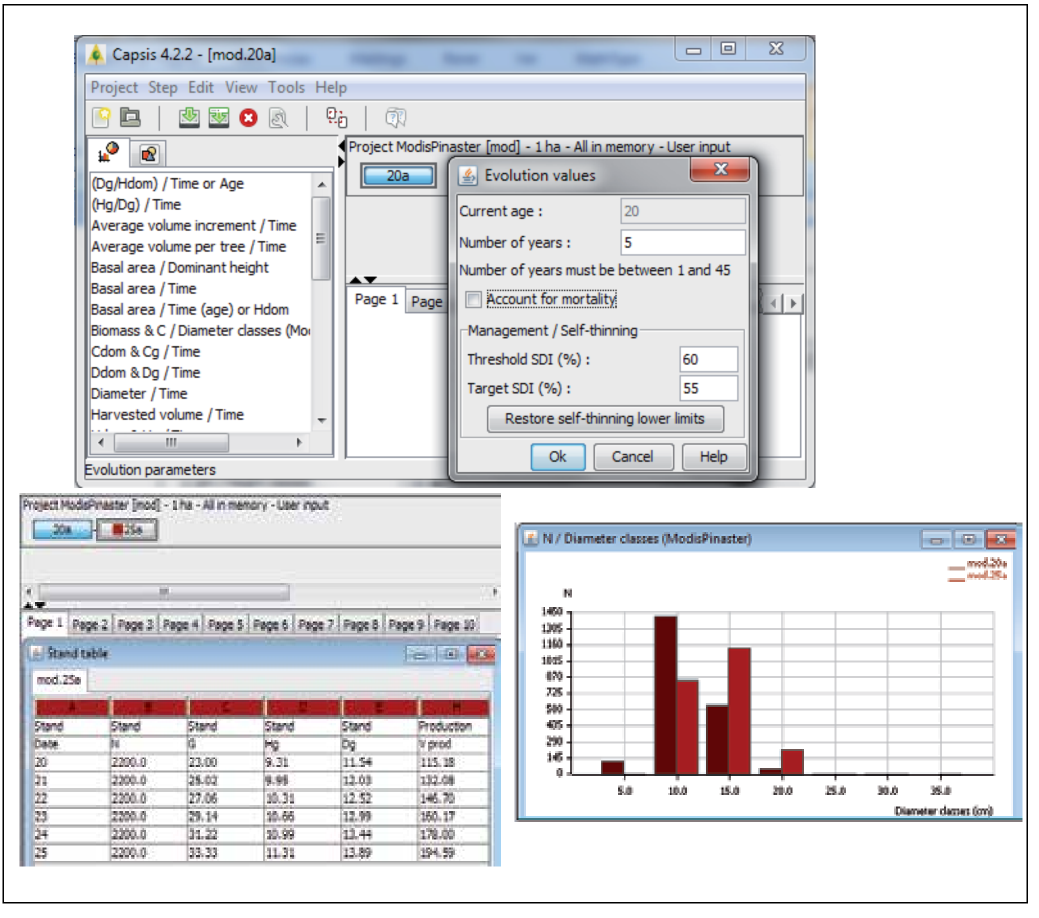

Fig. 4. The ModisPinaster evolution dialog and examples of the output information.

3.3 Analysis of different scenarios for a SFM

The majority of the maritime pine stands in Portugal are even-aged and are handled in a thinned managed regime. Density regulation is usually based on the Wilson spacing index. A typical value of Fw = 0.23 has been assumed in the maritime pine stands of the Tâmega Valley region (Moreira & Fonseca, 2002). The rotation age is defined by the age at which occurs the maximum annual increment of tree stem volume. Depending on site quality, stands attain their absolute explorability term for the volume variable when the stand reaches 35 (high quality) years to 45 (low quality) years (Marques, 1987; Moreira & Fonseca, 2002). In the area studied it is usual to set the lowest limit of stand age to harvest to 40-45 years.

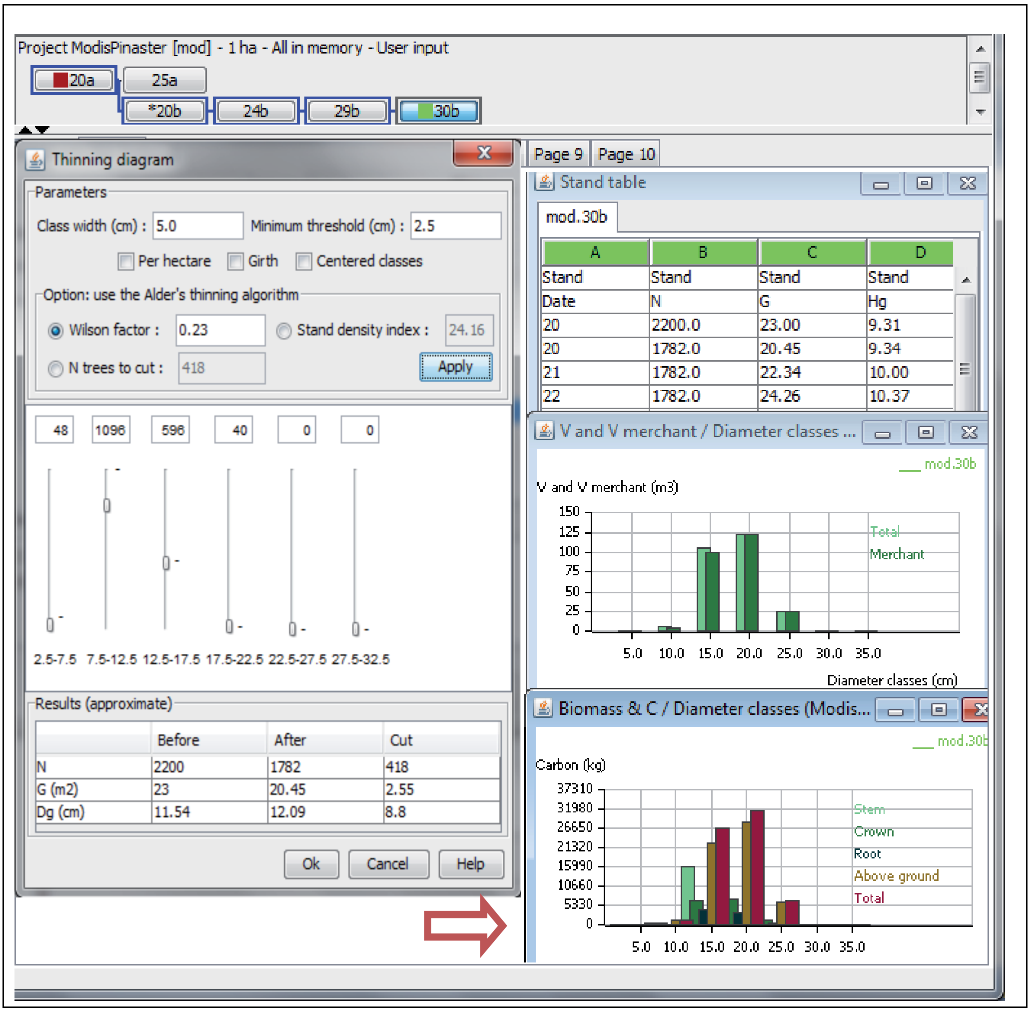

Fig. 5. The thinning dialog and some of the possible results available for ModisPinaster in the Capsis environment.

Three scenarios are proposed for comparison:

- Typical forestry guidelines (**TYF**). This scenario focuses on timber production, according to the traditional silviculture guidelines followed in the Tâmega Valley region. That is, cyclic thinning with an average silviculture compatible with a strong to moderate grade with *Fw* = 0.23; and a rotation age of 45 years.
- Low competition forestry (**LCF**). This scenario also focuses on the single purpose of timber. The management is made in accordance to the self-thinning line theory. The stand has a window for density between 25-35%, as measured by the stand density index, to keep the inter-tree competition at lower levels. The rotation age is maintained at 45 years.
- Combined objectives forestry (**COF**). Here, emphasis is done on maximizing the total volume yield and on promotion of the biodiversity. This scenario can be viewed as landowner absence where natural mortality is expected to occur. An old-stand situation is promoted. Rotation age is extended to 65 years.

The data used for the simulations refer to a 20 year old stand with the characteristics depicted in Figure 3.
For each scenario the following indicators were selected for comparison: yield in volume and in carbon in the aboveground component; products obtainable from the thinning practices (volume and average tree size). Dead wood was quantified in scenarios **TYF** and **LCF** as downed woody debris produced by thinning, considering that the tip of the trees (top diameter of 7 cm) are kept in the stand. For scenario **COF**, dead wood was quantified as the total volume of the stem for the trees that die during the rotation period. Table 1 presents the results obtained by following the current silviculture guidelines and the strategy of allowing the stand to grow in lower levels of competition. In the first case, a total of four thinnings, with a cycle of 5 years, starting at age 20, were considered until the stand reaches 35 years. Simulations of the thinning were made using the thinning dialog with specification of the target value for the Wilson factor (see Figure 5). After each thinning, the stand growth was simulated with the evolution dialog. For the **LCF** scenario, the simulation was made in an automatic mode using the facilities given in the evolution dialog (see Figure 4), setting a threshold of *SDI* equal to 35% and a target value of *SDI* equal to 25%. The first thinning occurs at the age of 24 years, a second at 30 years and a third and last one when the stand age is 37 years.

Sim.	t (yrs)	hd (m)	dg_b (cm)	N_b (trees ha^{-1})	C_b (ton ha^{-1})	V_b (m^3 ha^{-1})	V_t (m^3 ha^{-1})	Debris (m^3 ha^{-1})
TYF	20	10.3	11.5	2200	28.5	115.2	14.6	3.2
	25	12.9	14.7	1782	41.1	176.9	45.6	4.2
	30	14.8	18.9	1136	49.0	212.6	37.9	1.1
	35	16.5	22.6	859	58.1	255.8	37.6	0.5
	45	19.4	28.2	693	82.2	374.1		
LCF	20	10.3	11.5	2200	28.5	115.2	-	-
	24	12.5	13.4	2200	41.0	178.0	61.8	7.6
	30	14.8	18.9	1146	49.4	214.2	71.4	1.9
	37	17.1	25.2	648	57.4	252.5	79.3	0.6
	45	19.4	31.8	401	65.7	293.4		

Table 1. Characteristics of the stand, at age t (years), according to the simulation results of the management scenarios focused on timber production: typical forestry guidelines (TYF) and low competition forestry (LCF). The presented variables refer to dominant height (hd), quadratic mean diameter (dg), number of trees (N), carbon (C) and volume (V) of the standing trees before (b) the thinning practice. The products of the thinning operation refer to the volume of the removed trees (V_t) and to the tip debris.

The current silvicultural guidelines with a 4 thinning management regime through the rotation period produce a 25% greater yield, in terms of volume and carbon content, than the yield achieved with the low competition management. In terms of total volume (with the thinning removals included) the difference reduces to just 3.9 m^3 ha^{-1}, because the **TYF** scenario presents thinning removals of 135.7 m^3 ha^{-1}, while for the **LCF** scenario 212.5 m^3 ha^{-1} are harvested by thinning. These results could lead to an undifferentiated selection between both management options, or even to the preference of the **LCF** scenario as it provides timber of great size (31.8 cm of diameter, for the mean tree, at age 45). Nevertheless, under a

SFM other issues, such as the assessment of risk, need to be taken into consideration. For the example, the ratios of the mean height to mean diameter of the dominant trees vary between 0.57 and 0.60, during the rotation. This indicates a potential problem of stability of the trees under windy conditions and a high risk of wind damages. Therefore, to perform a thinning at age 24, with a density of 1054 trees ha^{-1}, might not be a secure option.

Also shown in Table 1 are the dead wood estimates from downed woody debris, ranging from 9 to 10.1 m^3 ha^{-1}. This is a part of the material obtained by thinning. Other values could be estimated depending on the specification of the merchantability limits and of accounting, for instance, additionally for the mass of the branches. For a complete assessment of the debris produced in the stand, it is suggested to evaluate the debris material using as an indicator the biomass (or carbon content) of the entire crown component (branches and leaves, instead of restricting the evaluation to the tip volume of the trees). Independently of the indicator chosen, the ModisPinaster features allow for the quantification of debris by diameter classes (not shown in Table 1). This might be of importance in some studies, such as when evaluating for the wood decay.

Although dead wood and decaying trees were considered for a long time as being of less or null commercial value, they do have considerable ecological value. The dead wood has a major influence on biodiversity. Many forest species, such as forest floor vertebrates and insects benefit or depend on dead wood material for habitat or resources. The scenario **COF** is presented as an example of a potential scenario to promote habitat and resources for the conservation of biodiversity. A comparison between the three scenarios is shown in Table 2.

Sim.	*t* (yrs)	d classes range (cm)	*dg* (cm)	Total volume (m^3 ha^{-1})	Harvested volume (m^3 ha^{-1})	Deadwood volume (m^3 ha^{-1})
TYF	45	20 - 35	28.2	509.8	135.7	9.0
LCF	45	20 - 35	31.8	505.9	212.5	10.1
COF	45	15 - 35	24.0	582.4	-	183.8
COF	65	25 - 40	31.5	920.1	-	350.7

Table 2. Characteristics of the stand at the end of the rotation age according to the simulation results achieved for the management scenarios of typical forestry guidelines (TYF), low competition forestry (LCF) and of combined objectives forestry (COF). In the TYF and LCF scenarios, the rotation age is fixed at 45 years while in the COF scenario an extended rotation age of 65 years is promoted. The characteristics refer to the diameter distribution (range and quadratic mean diameter, *dg*) and to the total volume of the standing trees at age *t*; to the removals obtained by thinning for the management options TYF and LCF; and to the accumulated volume of the trees that die during the simulation period for the three scenarios.

In Table 2, the total volume includes the volume from thinning practices, for the scenarios **TYF** and **LCF**, and the volume of the dead trees, for the **COF** scenario. As expected, a maximum value is achieved when there is no interference in the stand growth. This is consistent with the current consensus about the effect of stand density on growth (Zeide, 2001). The objective of thinning is to anticipate mortality and to provide better growth conditions for the remaining trees. Other goals might be added, such as, to obtain a target value for diameter at the thinning ages and at the final rotation. For the examples presented, the management according to a window for density between 25-35%, presents material distributed by 4 diameter classes (5 cm of amplitude) with average dimension slightly higher than the material obtained with the typical silviculture. When combined objectives are required, other guidelines need to apply. The growth under high densities (55-60% of *SDI*) and the extension of the rotation age, as presented in the **COF** scenario, allow for exploitation of wood, although of minor size, and guarantees better habitat conditions for the promotion of the fauna biodiversity.

4. Conclusion

The use of forest models has undoubtedly enhanced the scientific knowledge about forest dynamics and about the effects of alternative silvicultural options in the stand evolution. Taking as example the ModisPinaster model, it was shown how essential the models are for management decisions and planning purposes. The managers are facing challenges in terms of selecting the most appropriate management guidelines that assure the management goals, which might combine timber and other forest benefits, and increasingly of accounting for risk. Different scenarios are permitted for simulation, leading to better-quality choices under a Sustainable Forest Management guiding principle. From a user's point of view, other needs, such as an easy and free use of the models, are additionally mandatory. Software simulators of forest growth and stand dynamics should favour re-use and share methods and algorithms, promote integration and encourage partnerships. Capsis was delineated to follow these criteria. The examples provided here for ModisPinaster prove how an efficient software simulator can improve capabilities of models and encourage their use by the stakeholders for guidance in decision making.

The involvement in Capsis of different actors, developers, modellers and end-users, brings to the top the desirable features of "easy-to-use" models, allowing for a prompt search of guiding principles while securing the scientific validity of the simulation estimates.

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19

The Effect of Harvesting on Mangrove Forest Structure and the Use of Matrix Modelling to Determine Sustainable Harvesting Practices in South Africa

Anusha Rajkaran and Janine B. Adams
Nelson Mandela Metropolitan University
South Africa

1. Introduction

Mangrove forests exist along a transitional boundary between land and sea. They represent a continuum of biotic communities between terrestrial and marine environments (Hogarth, 1999; Kathiresan and Bingham, 2001; Alongi, 2008). These forests are globally distributed between the subtropical and tropical latitudes, restricted by major ocean currents and the 20°C isotherm of seawater in winter (Hogarth, 1999; Alongi, 2009). On a global scale, temperature is an important limiting factor but on regional and local scales variations in rainfall, tides, waves and river flow have a substantial effect on distribution and biomass of mangrove forests (Alongi, 2009). Erosion and depositional rates are also important as these affect the physical habitat that mangroves occupy. Generally the habitat of mangroves begins at mean sea level and extends to the spring high tide mark i.e. they exist in tidal areas (Hogarth, 1999; Spalding et al., 2010) while in South Africa mangroves are confined to estuaries that either may be permanently open to the sea or have an intermitted connection to the sea (Rajkaran, 2011). Estuaries are defined as "a partially enclosed body of water which is either permanently or periodically open to the sea and within which there is a measurable variation of salinity due to the mixture of seawater with freshwater derived from land drainage" (Day, 1980) as being; river mouths, estuarine bays, permanently open estuaries, temporarily open closed and estuarine lakes. There are five types of estuaries and these are defined by Whitfield (1992). The ecosystem services provided by mangroves include; shoreline protection from sea storms and excessive wave energy, nursery and areas of refugia for faunal populations (BOX 1), input of organic carbon into the food webs and filtration of silt and other compounds from the water column thereby protecting other nearshore ecosystems such as coastal reefs (Gilbert & Janssen, 1998; Fondo & Martens, 1998; Laegdsgaard & Johnson, 2001; Mumby et al., 2003). Mangrove forests are known to have survived for approximately 65 million years and therefore are resilient to large scale disturbances (Alongi, 2009). Key mangrove features that have assisted in their resilient nature include; the presence of a large reservoir of below-ground nutrients so that if a disturbance takes place the remaining nutrients will assist with the re-establishment of new seedlings to replace those that have been lost encouraging re-population of the disturbed area. Rapid biotic turnover has been recorded in mangrove forests and is facilitated by rapid rates of nutrient flux and microbial decomposition. Internal recovery after a disturbance is accelerated by complex and efficient biotic controls such as nutrient-use efficiency (Alongi, 2008, 2009). Frequent, small scale disturbances such as harvesting disrupts the flow of nutrients from the living biomass to the sediment environment via the roots, it also facilitates changes to the microenvironment which will reduce the capacity of the mangrove forests to recover.

BOX 1.

Faunal diversity in mangrove forests is high including organisms from sponges to elasmobranchs and bony fish as well as bird species such as the Mangrove Kingfisher (Nagelkerken et al., 2008). Crabs are the most abundant macrofauna (numbers and biomass) in mangrove forests (Smith et al., 1991). They consume or hide 30 to 80 % of leaves, propagules and other litter on the floor of mangrove forests (Dahdouh-Guebas et al., 1997; Machiwa & Hallberg, 2002; Skov et al., 2002). Crabs enhance degradation of leaves and make the leaves available to meiofauna (Dahdouh-Guebas et al., 1999). The diversity of crabs found in a mangrove forest may vary. At Mngazana Estuary the following species were found *Neosarmatium meinerti* de Man, *Sesarma eulimene* de Man, *Sesarma catenata* Ortmann, *Uca lacteal annulipes* H. Milne Edwards, *Uca chlorophthalmus chlorophthalmus* (H. Milne Edwards), as well as *Parasesarma leptosome* (Hilgendorf) (Plate 1). The latter is a tree climbing crab that spends most of its life in the mangrove trees and is therefore totally dependent on mangrove forests for their existence (Emmerson et al., 2003; Emmerson & Ndenze, 2007). More recently the species *Perisesarma samawati* Gillikin & Schubart, which was only described to occur in East Africa was spotted at Mngazana Estuary in South Africa in 2011 for the first time (Plate 2).


Plate 1 and 2: Images of crab species only associated with mangrove forests. Photos taken by Anusha Rajkaran

2. Mangrove forests: Utilization and destruction

In 2003, the global estimate of mangrove forest cover was 14 650 000 ha and accounted for approximately 0.7% of the total global area of tropical forests (Wilkie & Fortuna, 2003; Giri et al., 2011). Each hectare is valued at between 200 000 – 900 000 USD (Wilkie & Fortuna, 2003; Giri et al., 2011). Human disturbances has resulted in more than 50% of the world's mangrove forests being destroyed (Spalding et al., 2010). This huge loss of mangrove forests globally, has been attributed to urban development, aquaculture, mining along coastal zones and overexploitation of fauna and flora of mangrove forests (Walters, 2005; Walter et al., 2008; Kairo et al., 2008; Alongi, 2009). The connection between coastal developments, water level fluctuations and mangrove loss or transformation has been recorded by a number of authors in South Africa and other parts of the world (Moll et al., 1971; Begg, 1984; Bruton, 1980;

Dahdouh-Guebas et al., 2005) (BOX 2). Worldwide, mangrove forests are harvested for a variety of purposes. The products are particularly important to subsistence economies, providing firewood, building supplies and other wood products (Bandaranyake, 1998; Ewel et al., 1998; Cole et al., 1999; Kairo et al., 2002; Dahdouh-Guebas et al., 2004, Walters et al., 2008). The subsequent effects on the ecosystem ranges from loss of habitat for fauna such as arboreal crabs (Emmerson and Ndenze, 2007), decreases in organic carbon export to the food webs and nearshore environments (Rajkaran & Adams, 2007), coastal erosion (Thampanya et al., 2006) and in the long term, loss of nursery functions (Laegdsgaard & Johnson, 2001).

BOX 2

Freshwater abstraction and poor bridge design has caused the mouths of some South African estuaries with mangroves to close to the sea more frequently, leading to long term inundation of roots and subsequent death of the mangroves (Breen & Hill, 1969; Bruton, 1980; Begg, 1984). Rising water levels have been one of the main factors that have lead to localised mangrove disturbances and mortalities in Kosi Bay (1965-1966) and Mgobezeleni Estuary (74 km south of Kosi Bay) (Bruton, 1980). Past data shows that 78% of the 1084 trees died in the Mgobezeleni Estuary due to submergence of the root structures when the water level rose for an extended period of time. This was a result of water being impounded behind a bridge constructed in 1971. Dead mangrove trees ranged from 40 cm to 15 m in height showing that all height classes are susceptible to death due to water level increases. The living mangrove stand became infested by the mangrove fern. In 2007, 77 *Brugueira gymnorrhiza* trees were still living, these have all since died (2011). The water level was ~ 30 cm of water above the sediment. Less than five seedlings were seen in areas where the sediment was not submerged. This estuary is a prime example of how poor coastal planning and developments can have a negative effect on surrounding coastal habitats such as mangrove forests.

Plate 3 and 4: Images taken at Mgobezeleni Estuary in 2007 by Dr. Ricky Taylor showing the submergence of the root structures of the *Bruguiera* trees and the extent of the mangrove fern.

2.1 Effect of harvesting on mangrove forests

Gaps created during the harvesting of either individual or groups of trees provide opportunities for seedling recruitment and growth (Rabinowitz 1978; Ewel et al., 1998; Sherman et al., 2000). The size class structure of mangrove forests in localities that experience harvesting show under-representation in large size classes, which is the result of selective harvesting (Saifullah et al., 1994; Walters 2005). Because mangrove wood is used for building, the size of the mangrove poles determines the role they play in the built structure. A comparison of height classes of the non-harvested and harvested sites in the Mngazana Estuary ($31^{o}42'$S, $29^{o}25'$ E) in South Africa showed that the height class 2.3 – 3.3 m was dominant in non-harvested sites while in harvested sites smaller trees were dominant. All the harvested poles were approximately 3 m (Rajkaran & Adams, 2010). Traynor & Hill (2008) interviewed harvesters with regard to harvesting preferences at Mngazana Estuary; they stated that any tree greater than 2 m in height with a desired diameter at breast height (DBH) would be harvested. They also stated that the required length of the wall poles used for building homesteads was 3 m for wall poles while roof poles were usually 4 m. This explained the differences found for mangrove height between harvested and non-harvested sites. Traynor & Hill (2008) recorded that the preferred species for building was *Rhizophora mucronata* (41% of participants preferred this species) and *Bruguiera gymnorrhiza* (21%) while *Avicennia marina* was used for firewood.

3. The use of matrix modelling to determine sustainable harvesting practices

With the use of population models one can predict the quantitative changes in population structure and thus add value to any management plan established for a particular mangrove forest. Mathematical models are popular conservation and management tools used to predict changes to plant and animal populations that are at risk due to activities such as harvesting (Raimondo & Donaldson, 2003; López-Hoffman et al., 2006; Owen-Smith, 2007; Ajonina, 2008). Matrix models are age or stage structured models used in cases when harvesting of particular size classes is the main risk. One takes into account the probability of an individual plant moving from one size class to the next i.e. transition probabilities as well as the possibility of the individuals persisting in the size class or dying (Caswell, 2001; Porte & Bartelink, 2002; Boyce et al., 2006; Owen-Smith, 2007; Caswell, 2009). In the case of plants, the model usually uses plant size (height or DBH) as the basis for the model. Model parameters include recruitment (the portion of propagules that is produced by a specific size class that is added to Size Class 1), mortality (M), transition rates (T) and persistence rates (P) for each size class, these are known as the vital rates (Caswell, 2001; Porte & Bartelink, 2002; Owen-Smith, 2007) (Figure 1).

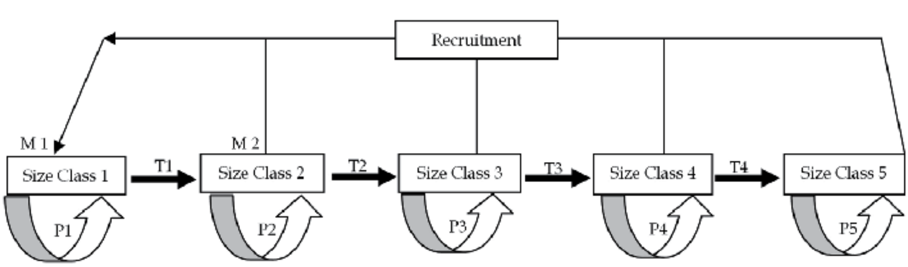

Fig. 1. The layout of the matrix model illustrating the vital rates mortality (M), transition rates (T) and persistence rates (P) for each size class.

The objective of this study was to develop a matrix model to determine the effect of different harvesting intensity scenarios, on the population structure of three mangrove species: *Avicennia marina*, *Bruguiera gymnorrhiza* and *Rhizophora mucronata*. The model results were compared to the observed population structure measured in the field at the end of the study in 2009 to determine the accuracy of the model and used to determine the most sensitive size classes to changes in vital rates within the population. Some data are presented here but more detailed results can be found in Rajkaran (2011).

3.1 Model development and accuracy

Nine sites at Mngazana Estuary were studied to collect data for the population model. This estuary is located in the Eastern Cape Province of South Africa, (Figure 2). In each site the following information was recorded, number of saplings (no hypocotyl less than 1 m), number of adults (over 1 m), the height of saplings and DBH and height of adults were measured. Subsequent measurements took place in November 2005, June 2006, November 2006, June 2007, November 2007, November 2008 and November 2009.

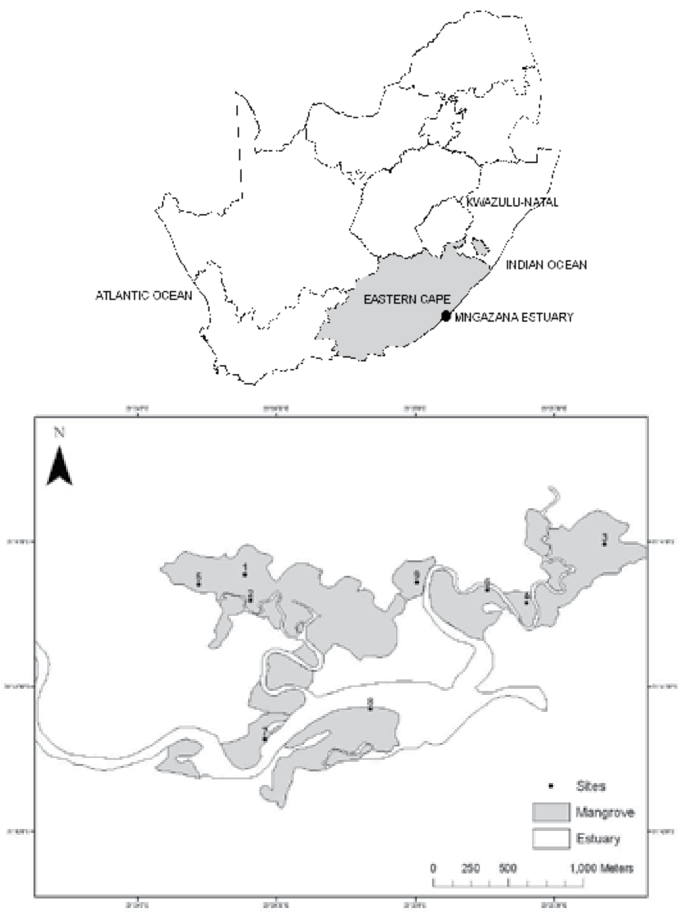

Fig. 2. The location of Mngazana Estuary in the Eastern Cape of the Republic of South Africa and the location of Sites 1-9 where growth was monitored from 2005-2009.

The population of each species, as calculated from nine sites around the estuary, was summarised and divided into a number of size classes based on mangrove height (Table 1). **Transition rates** were determined by counting the number of individuals in each size class over a period of five years (2005-2009). The **persistence rate** was the percentage of individuals that were in the same size class between two successive years (2005 compared to 2006). The transition rate was the percentage of individuals that were still alive but were now in the next successive size class therefore they had grown taller. Mortality rates were determined for the first two size class i.e. <50 cm and 50.5-150 cm height. The natural mortality of the other size classes could not be determined as none of the taller trees died unless they were harvested by the local community. In the model, natural mortality was included within the persistence rate i.e. the persistence rate was lowered by the appropriate percentage determined for each species based on the five year dataset. On two sampling trips (November 2005 and June 2006) the number of propagules on each tree was counted and the height of the tree was recorded. These data were used to determine the **fecundity** of each size class and were used as input on the proportion of propagules added by each size class to the total number of propagules.

Natural recruitment which was the number of new seedlings (hypocotyls present - <50 cm) added to the population was calculated for the five year period. Not all propagules that are produced establish themselves due to crab predation and removal by tidal movement. The number of individuals in each size class was converted from trees. m^{-2} (calculated from site data) to trees.ha^{-1}. The number of individuals that an area is able to support (carrying capacity) was assumed to be the total number of individuals in the population. The model was formulated to be density dependant, therefore the greater the number of individuals in the total population the stronger the effect of competition on the smaller individuals resulting in a lower survival rate. The time span for each population model was determined by how long the population size would take to stabilise. N_t is the size of the population at the start of the study. N_{t+1} is the sum of all the size classes calculated for each year after the start of the study (t+1). The ratio between N_{t+1} / N_t is the finite rate of increase and summarises the dynamics of a population. This ratio is symbolised by lambda (λ-the dominant eigenvalue of the matrix). When λ=1 then the population is in balance and remains stable ($N_{t+1} = N_t$), if λ>1 the population is increasing ($N_{t+1} > N_t$) and if λ<1 then the population is decreasing ($N_{t+1} < N_t$) (Slivertown & Charlesworth, 2001; Rockwood, 2006). Initial model results were compared to the observed population structure measured in the field at the end of the study in 2009 to determine the accuracy of the model.

3.2 Harvesting intensity scenarios

Harvesting scenarios represented a static harvesting rate of 1, 5, 10, 15, 20 and 100% of individuals for the three different species present at Mngazana Estuary. To determine the effect of harvesting on the total population (N) as well as different size classes a number of harvesting scenarios were added to the model. Population monitoring showed that harvesting of trees taller than 250 cm was common, therefore the model assumed that a percentage of Size Class 4 (250-350 cm) and 5 (>351 cm) would be harvested each year. The following harvesting intensities were used; 1, 5, 10, 15, 20 and 100% of a particular size class.ha^{-1}.$year^{-1}$. These scenarios would show how much of the population could be harvested and what the limit was for harvesting. The scenarios also showed how each size class changed in abundance in response to the different harvesting intensities.

3.3 Results

The *Avicennia marina* trees at Mngazana Estuary are either completely harvested or portions of the tree are cut for firewood. The assumptions for this model were 1) a tree, or portion of a tree, used for firewood is taken as a completely harvested tree and 2) that harvesting only affects the tallest trees in the forest (S5). The second assumption was based on field observations from Mngazana and Mhlathuze estuaries, where the tallest trees were the ones that were targeted. A hundred percent harvesting of individuals in the tallest size class decreased the total population to below 10 000 trees.ha^{-1} (Figure 3) and λ to 0.994 (Table 2). Restricting harvesting to just one size class that has reached reproductive maturity will ensure that other trees will still be present to produce propagules and subsequently seedlings. For this reason λ values as shown in Table 2 for *Avicennia* remain just below 1 for all harvesting scenarios. The number of individuals in Size Class 2 under 0% harvesting stabilised at less than 10 000 per ha (Figure 4). This decreased when the harvesting intensity increased as did the number of individuals in all size classes. To ensure more than 5 000 individuals were present in Size Class 1, which represents the main class for natural regeneration, harvesting must not exceed 20% of the trees taller than 350 cm per year. This is equivalent to 238 $\pm$ 4.5 harvested trees.ha^{-1}.yr^{-1}.

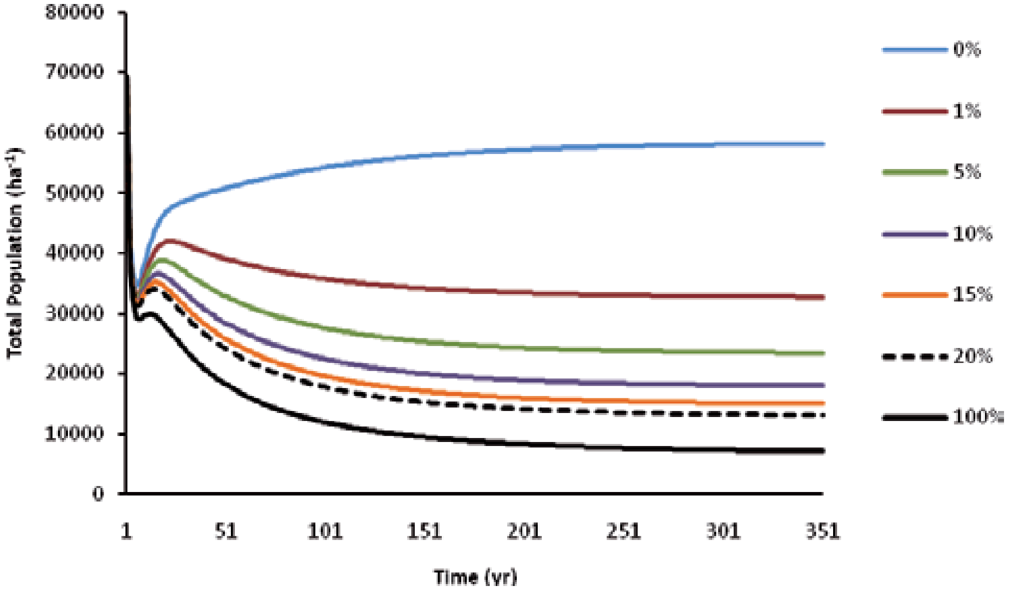

Fig. 3. Changes in total population size for the species *Avicennia marina* over time in response to different harvesting scenarios.

The assumption was that harvesting of two size classes would take place at Mngazana Estuary for *Bruguiera*. All trees greater than 251 cm would be removed. Harvesting of this species had a dramatic effect on the total population size. The total population of this species decreased by 63% when harvesting intensity was set at 1%. This allowed the population to stabilise at 15 000 trees.ha^{-1} (Figure 5). A further scenario was run using a harvesting intensity of 2%, this reduced the total population to approximately 5 000 trees.ha^{-1}. The mean λ for this species dropped from 0.999 to 0.834 at 100% harvesting intensity showing that the population was decreasing and natural regeneration was not taking place (Table 3).

Species	Size class (Height)	S1 <50 cm	S2 50-150 cm	S3 151-250 cm	S4 251-350 cm	S5 >351 cm
Avicennia marina	$N_{(t0)}$ (per ha^{-1})	16 786	40 536	8 036	2 500	1 339
	T	0.2	0.1	0.1	0.1	0
	P	0.6	0.8	0.9	0.9	0.9
	F	0	0	0	0.5	0.5
	MR (%)	21.0 $\pm$ 6.8	6.9 $\pm$ 2.0	ND	ND	ND
Bruguiera gymnorrhiza	$N_{(t0)}$ (per ha^{-1})	12 831	10 703	2 109	2 188	2 266
	TR	0.08	0.08	0.12	0.02	0
	PR	0.79	0.8	0.88	0.98	0.9
	F	0	0.16	0.16	0.33	0.33
	MR (%)	12.2 $\pm$ 4.6	7.2 $\pm$ 7.6	ND	ND	ND
Rhizophora mucronata	$N_{(t0)}$ (per ha^{-1})	11 979	43 750	10 104	8 125	2 917
	TR	0.3	0.03	0.1	0.03	0.1
	PR	0.6	0.88	0.9	0.97	0.9
	F	0	0.16	0.16	0.33	0.33
	MR (%)	15.6 $\pm$ 3.6	8.5 $\pm$ 2.3	ND	ND	ND

Table 1. Summary of data for each species and size class (S1-S5) used to populate the matrix models. (Transition rates (T) and persistence rates (P), fecundity rate (F), mortality rate (MR)).

		Size class (Height (cm)				
Harvesting intensity	**Total Population (N)**	**0-49**	**50-150**	**151-250**	**251-350**	**>350**
0%	1.000	1.001	0.997	1.001	1.004	1.006
1%	0.998	0.999	0.996	0.999	1.002	1.004
5%	0.997	0.998	0.995	0.998	1.002	1.002
10%	0.996	0.997	0.994	0.998	1.001	1.001
15%	0.996	0.997	0.994	0.997	1.001	0.999
20%	0.996	0.996	0.994	0.997	1.000	0.998
100%	0.994	0.995	0.992	0.996	0.999	0.999

Table 2. Mean λ values for *Avicenna marina* under different harvesting scenarios after 350 years.

Harvesting intensities of 15% and 100% were omitted from the graphs as the curves were similar to the 20% harvesting intensity and were not visible. Harvesting 1% of the adult trees maintained the density of size class 1 to < 5 000 individuals.ha^{-1} (Figure 6).

The same assumption regarding harvesting was used for *Rhizophora mucronata* that harvesting of two size classes would take place at Mngazana Estuary. Documented data showed that the average length for harvested poles was 3.4 m. Harvesting scenarios in the model were restricted to the last two size classes (>251 cm). Total population size decreased from ~ 80 000 to 28 000 individuals.ha^{-1} when harvesting intensity was 1%, this represented a 65 % reduction (Figure 7). λ values decreased to less than 1.000 showing that the population was decreasing as a result of the harvesting (Table 4). Harvesting intensity greater than 15% decreased the density of Size class 1 to ~3 500 individuals.ha^{-1} (Figure 8). Harvesting between 5-10% of trees per year would amount to 183 – 283 harvested trees.ha^{-1}.yr^{-1}.

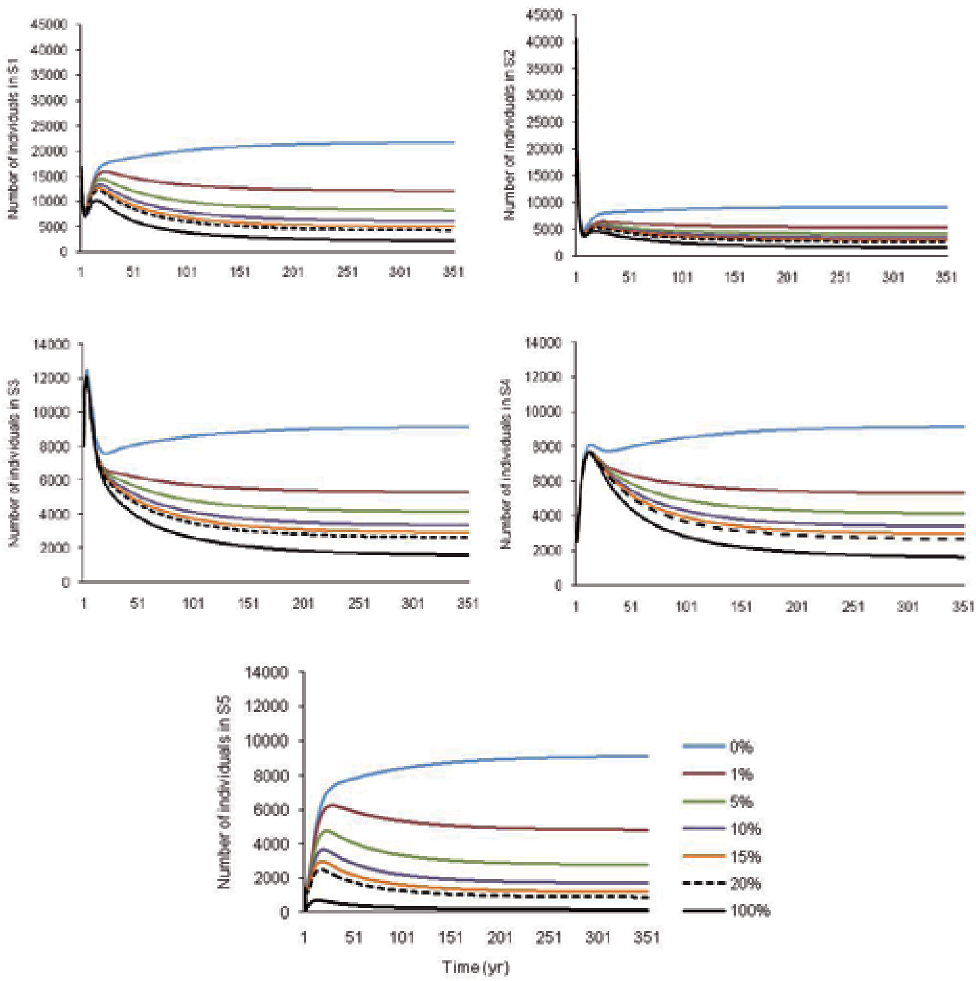

Fig. 4. The impact of harvesting on the number of individuals.ha^{-1} in each size class of the *Avicennia marina* population over time. (Y-axis was not standardised for all graphs so that curves would be visible)

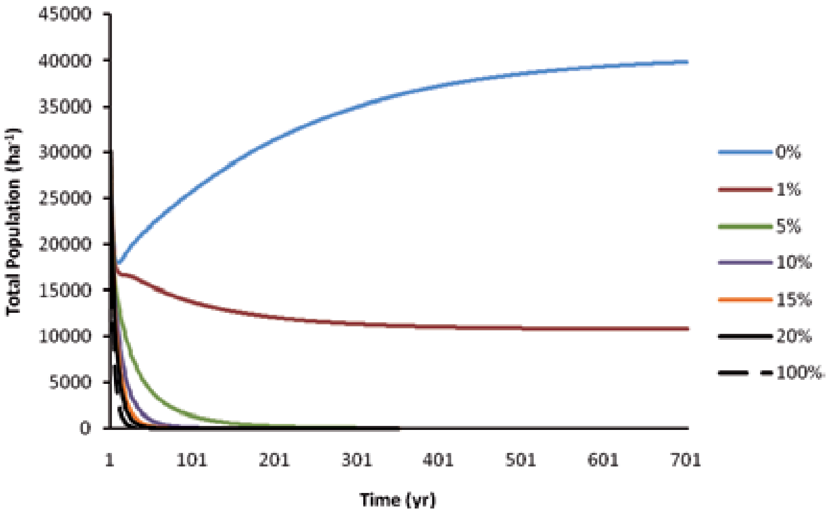

Fig. 5. Changes in total population number for the species *Bruguiera gymnorrhiza* over time in response to different harvesting scenarios.

		Size class (Height (cm)				
Harvesting intensity	**Total Population (N)**	**0-49**	**50-150**	**151-250**	**251-350**	**>350**
0%	1.000	1.001	0.999	1.000	1.002	1.000
1%	0.999	0.999	1.000	1.000	0.999	0.999
5%	0.980	0.980	0.979	0.981	0.983	0.977
10%	0.949	0.949	0.948	0.950	0.951	0.946
15%	0.922	0.922	0.922	0.924	0.924	0.918
20%	0.901	0.901	0.901	0.902	0.902	0.896
100%	0.834	0.832	0.835	0.834	0.832	0.822

Table 3. Mean λ values for *Bruguiera gymnorrhiza* for different harvesting scenarios after 701 years, the number of years required for the population to reach equilibrium was greater than for the other two species.

3.4 Discussion

Small scale disturbances such as harvesting, depending on the timing, frequency and intensity, which result in the loss of some of the mangrove population, may lead to natural regeneration if there are existing seedlings, saplings and mother trees (standard) around the disturbed area, if there is potential for water-borne propagules to travel to the area via tidal flow and if the propagules from disturbed trees are still present (FAO, 1994). A "standard" is defined as a seed bearing tree that can withstand exposure to strong winds and light and, in fringe areas, high tidal action (FAO, 1994). Regeneration will be restricted if the number of standards is reduced, if dead trees and branches reduce the light on the forest floor, if damage occurs to surrounding seedlings/saplings due to trampling and if a substantial change in soil conditions occurs (FAO, 1994; Harun-or-Rahsid et al., 2009).

Clarke et al., (2001) noted that the lack of diaspore dormancy in most mangrove species translates into a small or non-existent seed bank. The lack of a persistent soil seed bank of true mangrove species decreases the probability of a full recovery by mangrove populations after large scale disturbances and increases the chances of invasions of mangrove-associate species (Dahdouh-Guebas et al., 2005; Harun-or-Rahsid et al., 2009). Populations are reliant on regular cohorts of diaspores for regeneration so their continuous production by adults is vital. Rajkaran & Adams (2007) recorded movement of propagules out of the creeks and main channel of Mngazana Estuary, dispersed propagules were found on the adjacent beach near the mouth of the estuary. At Mngazana Estuary the presence of propagules on the forest floor is dependent on that produced by the adults in that specific area and not on the propagules brought in by tides. So at this estuary the continuous production by adults remaining after harvesting is vital for natural regeneration.

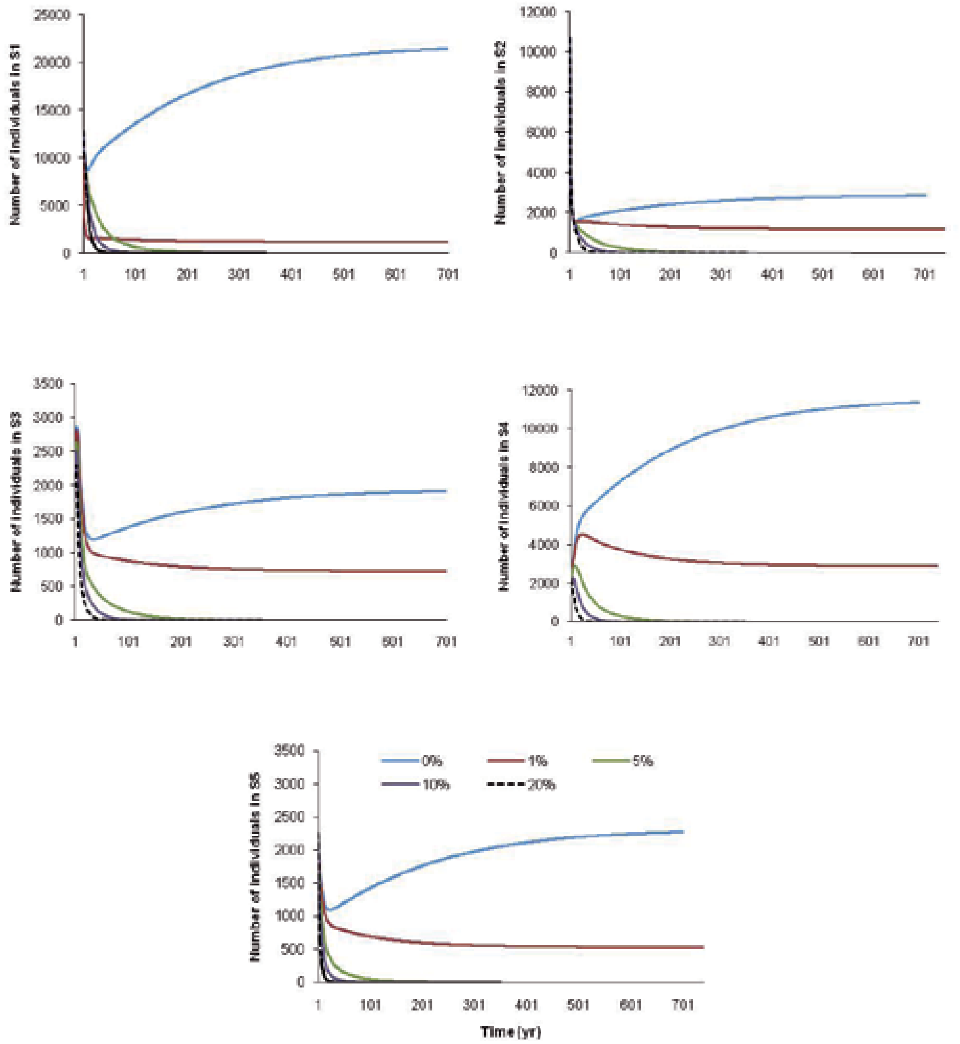

Fig. 6. The impact of harvesting on the number of individuals.ha^{-1} in each size class of the *Bruguiera gymnorrhiza* population over time. (Y-axis was not standardised for all graphs so that curves would be visible).

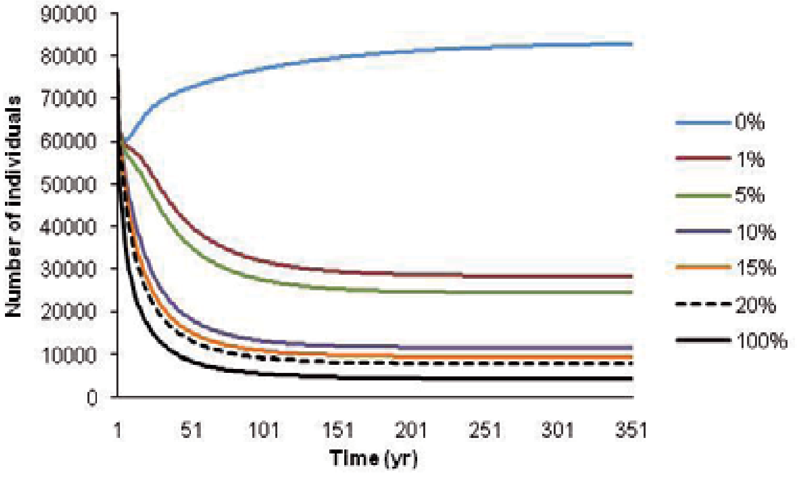

Fig. 7. Changes in total population size for the species *Rhizophora mucronata* over time in response to different harvesting scenarios.

		Size class (Height (cm)				
Harvesting intensity	**Total Population (N)**	**0-49**	**50-150**	**151-250**	**251-350**	**>350**
0%	1.000	1.003	0.998	0.999	1.003	1.002
1%	0.997	1.000	0.996	0.997	0.997	1.000
5%	0.997	0.999	0.996	0.996	0.997	0.999
10%	0.995	0.997	0.994	0.995	0.993	0.994
15%	0.994	0.996	0.994	0.994	0.992	0.993
20%	0.994	0.996	0.994	0.994	0.992	0.991
100%	0.992	0.994	0.992	0.992	1.001	0.974

Table 4. Mean λ values for *Rhizophora muronata* for different harvesting scenarios after 350 years.

Size classes in this study were based on height as previous studies have shown that harvesters targeted specific heights within the population (Rajkaran & Adams, 2009; Traynor & Hill, 2008). A density dependent model was used to simulate population structure and growth over time and the results conformed well to the logistical equation. The average λ value for each species in the absence of harvesting scenarios was 1.000, which shows that the populations are not increasing under the current harvesting rates for each size class. This may be a consequence of the continuous past harvesting in the Mngazana mangrove forest that has influenced vital rates. This was not taken into account in this model. López -Hoffman et al. (2006) recorded λ values of 1.050 when no harvesting was taking place. Vital rates for *Rhizophora mucronata* were comparable to those measured by López -Hoffman et al., (2006). Persistent rates ranged from 0.909 to 0.983, while transition rates ranged from 0.026-0.034 for adult size classes in that study, which is similar to the current study for this species. Similar studies for *Bruguiera gymnorrhiza* were not found. Clarke, (1995) used a matrix model to predict the population dynamics of *Avicennia marina* in New Zealand. Persistence rates for seedlings were 0.825, saplings - 0.909, young tree -

0.963 and older trees 0.999, while transition rates were 0.010, 0.073, 0.008, 0.012, 0.000 respectively. Sizes of each life stage were not stated in the study. The persistence rate in this study for *Avicennia* seedlings was much lower at 0.6 and transition was higher at 0.2, while all other rates were comparable with other studies. This implies that *A. marina* seedlings in South Africa grow faster and more seedlings survive to the next population size class within one year but the overall survival of the seedlings is similar between the two studies. Faster growth rates are dependant on site specific environmental conditions such as sediment characteristics and interspecific competition (Rajkaran, 2011).

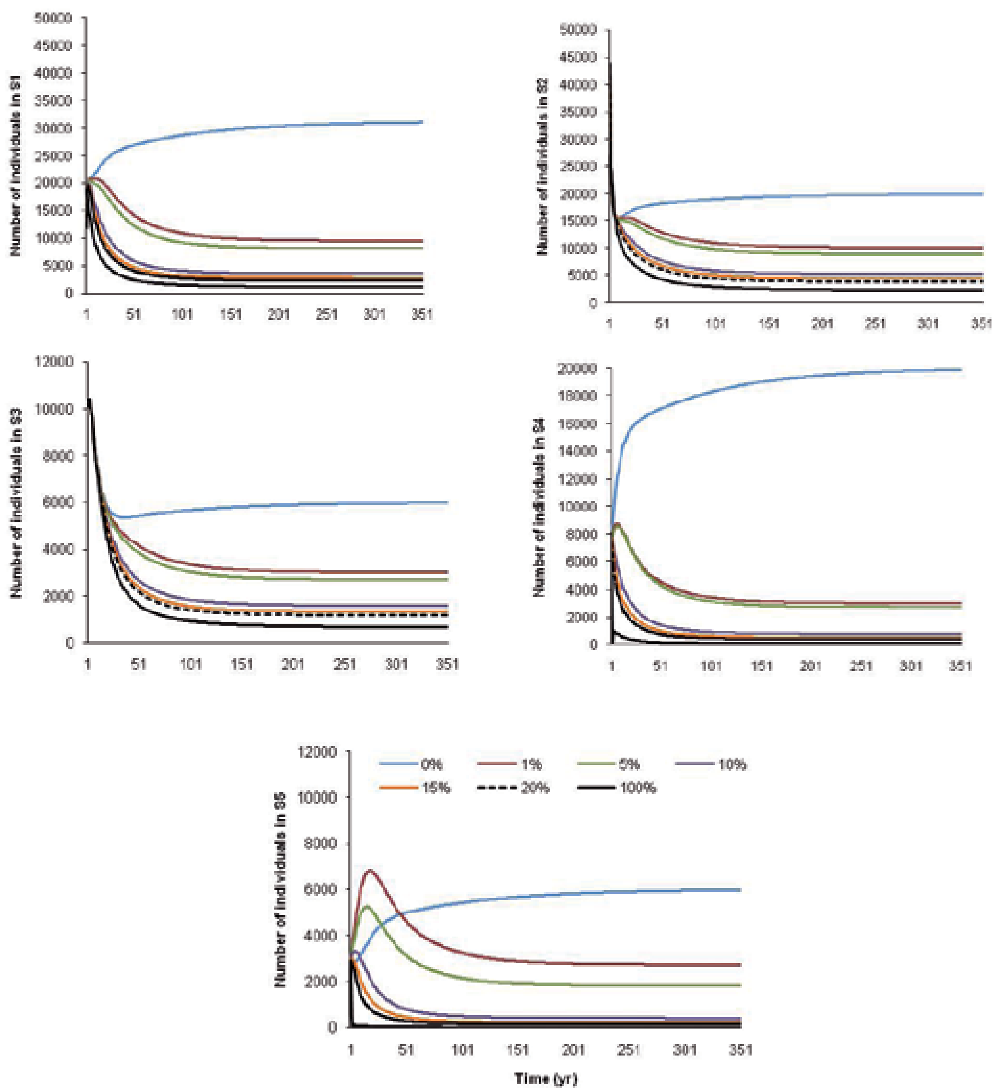

Fig. 8. The impact of harvesting on the number of individuals.ha^{-1} in each size class of the *Rhizophora mucronata* population over time. (Y-axis was not standardised for all graphs so that curves would be visible for S5 and S3)

All harvesting scenarios decreased λ to less than 1.000, showing that the populations were decreasing in size. A sustainable harvesting rate would be one where λ is greater than 1. This would indicate that harvesting would be increasing the population growth by increasing space and decreasing competition between individuals. A λ value of 1.000 would mean that the population is unchanging (López -Hoffman et al., 2006) and disturbance would be detrimental to the population. FAO (1994) have set minimum limits for the number of "mangrove" seedlings that must be present to facilitate natural regeneration once adults have been removed from the population. The harvesting intensity that leads to a seedling density of less than 5000.ha^{-1} were 100% intensity for *Avicennia marina* all intensities greater than 1% for *Bruguiera gymnorrhiza* and 15, 20 and 100% for *Rhizophora mucronata*. The limits of harvesting in the Mngazana mangrove forest should not approach these levels. López -Hoffman et al., (2006) set sustainable harvesting in the Rio Limón mangrove forests of Lake Maracaibo in Venezuela at 7.7% per year for *Rhizophora mangle*, the current study has set harvesting limits at 5% per year for *Rhizophora mucronata* and *Avicennia marina*. Harvesting of *Bruguiera gymnorrhiza* should be stopped as the density of this species is lower than the other two species. Preferably there should be no harvesting of this species. Harvesting intensity must ensure that seedling density is maintained within acceptable limits as set out in the published literature (FAO, 1994; Bosire *et al.* 2008; Ashton & Macintosh, 2002). A density of 2 500 – 3 200 seedlings ha^{-1} has been suggested as a minimum number required for natural regeneration to take place after a disturbance (FAO, 1994; Bosire et al. 2008). Ashton and Macintosh (2002) recommended 5 000-10 000 seedlings ha^{-1} for adequate regeneration in a cleared area in the Matang Mangrove forest in Peninsular Malaysia. Density of individuals of the three species were measured at Mngazana Estuary in 2005 and were found to be 17 000, 13 000 and 12 000 seedlings.ha^{-1} for *Avicennia*, *Bruguiera* and *Rhizophora* respectively. To set the minimum number of seedlings to 5 000 individuals.ha^{-1} would mean that this size class would be more than half the original density. Increasing the limit to 10 000 seedlings.ha^{-1} would be more acceptable at the Mngazana Estuary for all species. The harvesting limits for each species will be different but managers must ensure that the seedling densities are maintained.

Mangrove management regimes may also suggest different densities for standards, i.e. the reproductively active trees producing propagules; these range from 7 (Malaysia) to 20.ha^{-1} (Phillipines) (Choudhury, 1997). This depends on the species; FAO (1994) suggested 12 standards.ha^{-1} for the genus *Rhizophora*. These levels are recommended for forests where clear-felling takes place in tropical countries where growth rates are high. Clear felling should be avoided in the Mngazana mangrove forest as this will significantly change sediment characteristics. Sediment conditions are significantly affected by changes in vegetation cover and plant density in a mangrove forest (Rajkaran and Adams, 2010). Mangrove forests are made up of species that are able to attain slow growth under a wide variety of conditions (Krauss et al., 2008) but Rajkaran and Adams (under review) recorded that growth and mortality of different size classes within a population were related to certain sediment parameters i.e. seedling growth was negatively related to high sediment pH (*Rhizophora* upper limit for pH in this study was 7.1) while seedling mortality for *Bruguiera* was negatively affected by an increase in sediment moisture.

A harvesting intensity of 5 % would maintain the number of individuals for *Rhizophora mucronata* at greater than 3 000.ha^{-1} in Size Class 3 and Size Class 4 while Size Class 5 would be reduced to approximately 2 000 individuals.ha^{-1}. Traynor & Hill (2008) estimated the annual demand for mangroves at 18 400 stems.yr^{-1} at Mngazana. These were mainly used by the local communities to build homesteads. The suggested harvesting intensity of between 5 and 10% per year would provide this required number of stems and indeed yield more harvested stems than those required at the time of the 2008 study. A more detailed study about the increase in the demand over time due to increases in the human population is required, but in the meanwhile an alternative wood resource must also be provided to the communities to replace the mangroves. The full effects of harvesting have not been measured in this study because, for example, the effects of trampling on seedling survival and its influence on population growth and structure were not addressed. Recruitment was extremely low in this study which may have been the influence of physical disturbance from harvesters. Other management recommendations include reducing harvesting within the 10 - 20 m strip from the estuary channel. The purpose would be to sustain trees that form a barrier between the energy of the water flowing in on a high tide and the young seedlings.

4. Management of mangrove systems in South Africa

The management of ecosystems calls for the interaction between researchers and society to ensure that environmental and socio-economic issues are integrated with government policies. For this to take place a number of conceptual frameworks exist as tools for communication between researchers and end users of environmental information such as government departments (Maxim et al., 2009). The Drivers-Pressures-Status-Impact-Response (DPSIR) framework focuses on the connecting relationships between the **Driving** forces that are usually societal and economic developments that place the environment under **Pressure** which alters the **State** of the environment, and **Impacts** on the ecosystems. The **Response** from society is usually in the form of regulatory laws or rehabilitation plans depending on the situation (Bidone & Lacerda, 2004; Maxim et al., 2009; Omann et al., 2009; Atkins et al., 2011). The DPSIR framework allows managers and scientists to highlight issues that must be prioritised with regard to management of natural systems. The DPSIR framework was applied to the results from this research and identifies the issues associated with the management of mangroves in South Africa (Figure 9).

Overall interventions for the conservation of mangroves in South Africa include directly protecting pristine mangroves, protecting the hydrological regimes supporting these ecosystems (particularly freshwater quantities flowing into the estuaries-which would be dependent on the base-flows required to maintain mouth conditions in the optimal state), promoting natural regeneration for self renewal, enforcing mangrove buffer zones and the continued capacity development and education of those communities that use the forests (Macintosh & Ashton, 2004). Mangrove buffer zones provide protection to any habitat or human areas behind them. Vietnam maintains a 100 m – 500 m wide belt of mangroves to protect the Mekong Delta coastline against storm and flood protection, while the Philippines maintain a 20 m wide zone for protection of shorelines (Macintosh & Ashton, 2004). All mangroves in South Africa are found within estuarine ecosystems so their capacity to protect the coastline is limited. However in many cases coastal developments have occurred along the banks of estuaries behind mangrove and salt marsh communities. In these cases it is recommended that a mangrove buffer zone of 25 m be maintained and in the case of creeks, a 10 m buffer zone should be created. No activities, such as harvesting, should take place within these zones. In addition to these measures the identification and promotion of alternative resources for building is required.

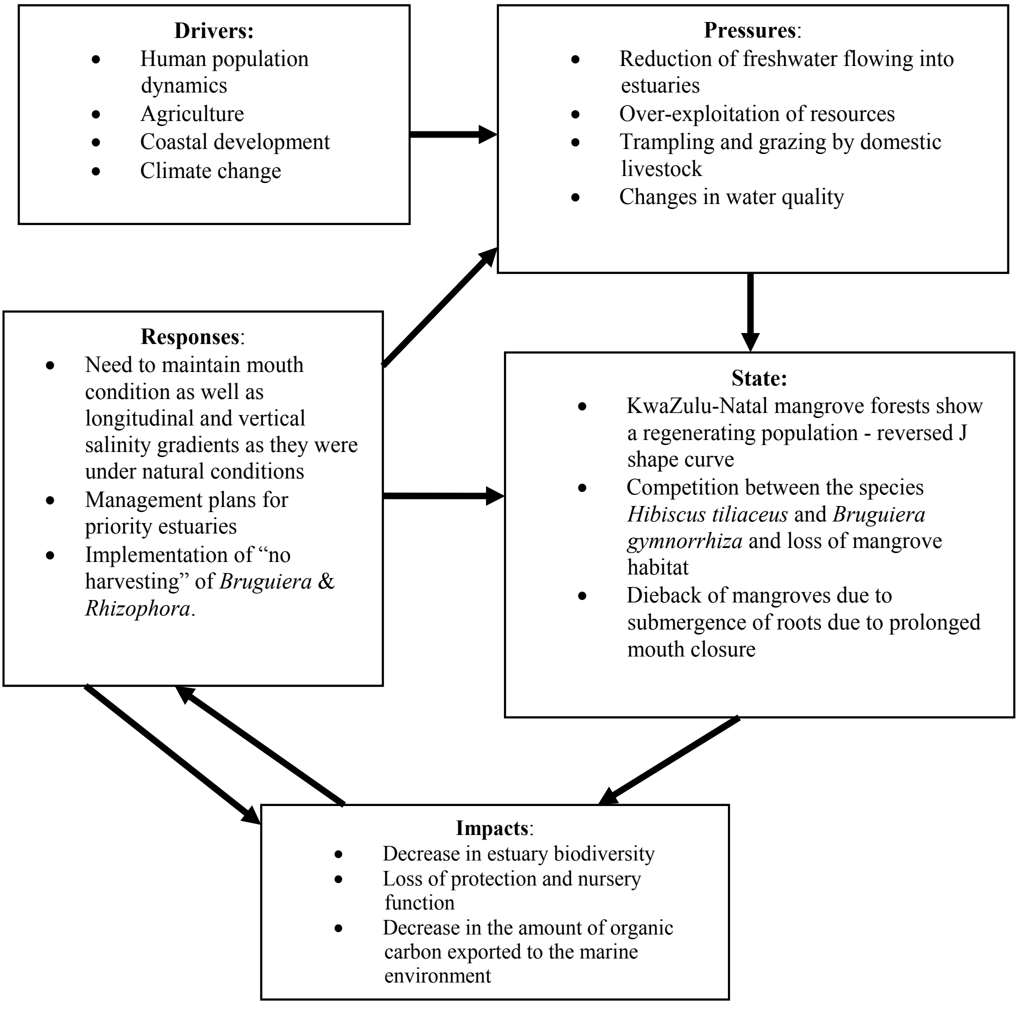

Fig. 9. Summary of DPSIR framework for the mangrove forests of South Africa.

5. Conclusion

Matrix modelling has allowed us to determine how much of a mangrove forest can be harvested while still maintaining a viable population. These data must be included in any management plan which includes the continual use of the forests as a wood resource for the local communities. The model presented here can be used by managers at other forests but growth data would need to be collected first as vital rates presented here will differ to other mangrove forests.

6. Acknowledgements

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20

Individual-Based Models and Scaling Methods for Ecological Forestry: Implications of Tree Phenotypic Plasticity

Nikolay Strigul
Stevens Institute of Technology
USA

1. Introduction

The concept of sustainable forest management (SFM) has been developed across traditional disciplinary boundaries, including natural resource management, environmental, social, political, economical, climatic sciences and ecology. The Montreal process (www.mpci.org) has established multidisciplinary criteria for the SFM of temperate and boreal forests. In parallel with the Montreal process, the pan-European forest policy process (www.foresteurope.org, Forest Europe, The Ministerial Conference on the Protection of Forests in Europe, MCPFE) has developed criteria for SFM in Europe. Practical implementation of SFM criteria requires the development of scaling methods to link individual-level processes, pollution effects, climatic changes and silvicultural operations to large-scale ecosystem patterns and processes. A general problem is that data obtained in numerous experimental studies that address effects at the individual level cannot be translated to the ecosystem level without a large amount of uncertainty. Forested ecosystems have a complicated spatially heterogeneous hierarchical structure emerging from numerous interdependent individual processes. The fundamental ecological questions are how macroscopic patterns emerge as a result of self-organization of individuals and how ecosystems respond to different types of environmental disturbances occurring at different scales (Levin, 1999).

The SFM employs the ecological forestry (EF) silvicultural approach, which is significantly distinct from the intensive (traditional) forestry and, therefore, requires different modeling tools than traditional forestry models. Traditional or intensive forestry is focused on wood production to maximize productivity of land use and usually involves tree plantations of commercially important trees (Nyland, 1996; Perry, 1998). Different silvicultural tools help increase wood fiber production. In particular, use is made of fast growing and disease resistant cultivars, vegetation control via thinning and regeneration harvesting techniques, soil management, and forest pests and noncrop vegetation control. Intermediate cutting operations include low, crown and mechanical thinning target future stand growth on higher valued trees to improve the stand yield at final harvest while providing some financial return on the shorter time scales. Traditional forestry also employs prescribed fire, cutting and application of herbicides for regulation of species composition and promoting growth of economically important tree species in the mixed stands.

This chapter is focused on modeling tools for the SFM and EF. The objective of this approach is the optimization of land use (such as wood production and carbon storage) while maintaining biocomplexity of forested ecosystems. The models discussed in this chapter are to be implemented within the SFM framework to optimize land use (such as wood production and carbon storage with the criteria 2 and 5 of Montreal process "Maintenance of productive capacity of forest ecosystems" and "Maintenance of forest contribution to global carbon cycles", respectively; and criteria 1 and 3 of the MCPFE process (Ministerial Conference on the Protection of Forests in Europe) "Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles" and "Maintenance and encouragement of productive functions of forests", respectively. The fundamental challenge for ecological forestry is to effectively manage a forest - complex ecological system, rather than a plantation of trees as in the traditional forestry approach. The biocomplexity challenges for ecological forestry are the understanding of why different plant species coexist, and which forces drive forest community structure and dynamics. One of the keystones of ecological forestry is the development of forest management systems in concert with natural processes in forested ecosystems, such as natural disturbances, forest dynamics and succession (Franklin et al., 2007). In particular, development of regeneration harvest approaches that have ecological effects similar to natural disturbances has been considered crucial for ecological forestry. Natural disturbances may occur at different spatial scales resulting in heterogeneity of forested ecosystems. The most common natural disturbances include wind-related disturbances on the individual (forest gaps) and large-scale (for example created by hurricanes and tornadoes), fire-related disturbances, and pest or disease related disturbances. These disturbances may significantly alter ecosystem structure and dynamics; however even the most dramatic events do not completely destroy ecosystems. Certain biological patterns or biological legacies, specific for each type of disturbance, remain unchanged and facilitate forest post-disturbance recovery.

Forest heterogeneity, which emerges as the result of various disturbances, is an essential element of ecological forestry, in contrast to the traditional approach, where stands are spatially homogeneous to reduce tree competition and improve timber quality (Oliver & Larson, 1996). Morphological plasticity allows trees to compete with neighbors and survive in a heterogeneous environment. In particular, open-growing trees, as well as trees growing in plantations without intense crown competition, tend to have symmetrical crowns, straight trunks, and, as a result, high quality timber. Trees growing in mixed spatially heterogeneous stand tend to exhibit plasticity patterns as every individual tree needs to adjust to its local unique neighborhood. These trees have much less value in term of timber than plantation trees. Such trees often have non-symmetrical crowns and curved trunks as they lean towards sunlight due to intense individual tree competition.

Forested ecosystems demonstrate multiple-scale self-organization patterns in response to disturbances. At present, we lack the predictive modeling tools that can combine the effects of forest disturbances occurring at different scales. An ideal model would present an analytically tractable model predicting landscape-level vegetation dynamics using individual ecophysiological traits as variables and available forest survey data as initial conditions. How can we develop such models?

Simon Levin, in a seminal paper (Levin, 2003), considered a modern theoretical approach to multiscale ecological modeling. In particular, he introduced ecological systems as complex adaptive systems which result from self-organization on multiple levels, where individual organisms are linked through interactions between each other and the abiotic environment. In this chapter the framework of complex adaptive systems is applied to forest ecology

and management. The forest is considered as a complex adaptive system (as a mosaic of individual plants, each of which grows adaptively in its biotic and abiotic environment in dynamic interaction with its neighbors). These interactions occur simultaneously at different temporal and spatial scales, both above and below ground, and lead to the development of self-organized patterns and structural complexity. The central question of this chapter is how forest patterns emerge as a result of the self-organization of individual trees. Individual tree plasticity is a critical process for forest modeling, though it has previously not been taken into account. The plasticity patterns of tree crowns in response to light competition enable directional growth toward available light, and lead to tree asymmetries caused by stem inclinations and inhomogeneous branch growth. Recently developed individual-based forest simulators, Crown Plastic SORTIE (Strigul et al., 2008) and LES, focus on forest self-organization at the stand level. These models, by incorporating individual crown plasticity, predict substantially different macroscopic patterns than do previous models (regularity in canopy spatial structure, for instance, which has only recently been noticed in field studies). Most importantly, the simulator's structure allows to derive an accurate approximation of the individual-based model, the Perfect Plasticity Approximation (PPA). This macroscopic system of equations predicts the large-scale behavior of the individual-based forest simulator, using the same parameter values and functional forms (Strigul et al., 2008). In particular, the PPA offers good predictions for 1) stand-level attributes, such as basal area, tree density, and size distributions; 2) biomass dynamics and self-thinning; and 3) ecological patterns, such as succession, invasion, and coexistence.
This chapter also introduces a theoretical framework for the scaling of forest spatial dynamics from individual to the landscape level based on the PPA model. The major objective of this approach is to scale up forest heterogeneity patterns across the forest hierarchy. The major idea is that the forest dynamics at the landscape level can be modeled by separating dynamics within forest stands caused by individual-level disturbances from the dynamics of the stand dynamics caused by large disturbances. The model, called Matreshka (after the Russian nesting doll) employs the PPA model as an intermediate step of scaling from the individual level to the forest stand level (or patch level). To describe the patch dynamics at the next hierarchal level, i.e., the forest stand mosaic, we employ the patch-mosaic modeling framework (Strigul et al., 2012). The Markov chain model for the mosaic of forest stands in the Lake states (MI, Wi, and MN) has been recently parameterized using the FIA data (Strigul et al., 2012). The Matreshka model unites already known models and uses the notion of ecological hierarchy that has been widely employed in landscape ecology (Bragg et al., 2004; Clark, 1991; Wu & Loucks, 1996).
The chapter consists of three sections: Section 2 introduces tree morphological plasticity as a fundamental pattern for the canopy self-organization. In section 3 the individual-based modeling approach is considered with a special focus on the development of the Crown Plastic SORTIE model, LES, and PPA models. Section 4 introduces a theoretical framework for the scaling of forest dynamics from individual to the landscape level based on the PPA model.

2. Individual tree plasticity and canopy self-organization

2.1 Crown plasticity and leaning of individual trees

In competing for light, trees invest carbon and other resources to achieve such above-ground form as will provide them with enough light for photosynthesis. To achieve this goal, individual trees demonstrate amazing phenotypic plasticity, using advantages of modular organization (Ford, 1992). Numerous factors constrain tree development, for instance, gravity

Fig. 1. Tree growing across a small forest river. A typical example of tree plasticity and "riverside behavior". The Institute for Advanced Study Woods (Princeton, NJ).

(McMahon & Kronauer, 1976) and other abiotic factors such as wind (Grace, 1977) and snow (King & Loucks, 1978), neighborhood effects (Ford, 1992), and, also genetic and physiological constraints such as the need to provide an efficient connection between their own above- and below-ground parts (Kleunen & Fischer, 2005). One permanent goal of a given individual tree is to develop an optimal crown under the current limitations in the dynamic environment. This includes different wood-allocation strategies in open-growing trees and trees in dense stands (Holbrook & Putz, 1989), as well as the development of sun branches and the degradation or physiological modification of shaded branches (Stoll & Schmid, 1998).

The physiological mechanisms underlying plant-phenotypic plasticity and phototropism have received significant attention in recent decades, yet many phenomena remain unclear (Firn, 1988; Kleunen & Fischer, 2005). Apical control can partially explain interspecific differences in tree leaning, crown shapes, and also differences in growth patterns between understory and overstory trees (Loehle, 1986; Oliver & Larson, 1996). Plants have also a variety of photosensory systems to detect their neighbors and select an optimal growing strategy. A better-investigated, phytochrome-signaling mechanism triggers some adaptive morphological changes such as adaptive branching (Stoll & Schmid, 1998) and stem elongation (Ballaré, 1999) in response to the alterations in far-red radiation caused by the reflection of sunlight by neighbor plants.

Tree-plasticity patterns relating to the competition for light and phototropism include the development of an asymmetrical crown, as a result of both the growth of individual branches and the phototropism of the whole tree (resulting in trunk elongation and inclinations). Tree-plasticity patterns caused by competition for light are more pronounced near forest margins, such as road cuts or riverbanks (Fig. 1). At these places trees develop asymmetric crowns and lean toward the gap, this pattern was called "riverside behavior" (Loehle, 1986).

Crown asymmetries and tree leaning can be also caused by factors not related to light competition, for example, soil creep (Harker, 1996), wind (Lawrence, 1939), and destruction of the apical meristem by insects. Trees growing on hillsides often have special trunk inclinations induced by soil creep, which geologists call a "d" curve (Harker, 1996). In this case, the base of the tree trunk starts at an angle to the vertical, with this angle continuously decreasing toward the top of the tree. However, such trees can have symmetrical crowns. This type of curved trunk is used as an indicator of soil creep. It was suggested that downward trunk inclinations of understory trees growing on the slope may have adaptive significance for light competition (Ishii & Higashi, 1997), however other authors disagree with this hypothesis (Loehle, 1997).

While crown asymmetry and trunk inclination represent two closely related patterns providing for tree morphological plasticity in light competition, the development of asymmetrical crown has received more attention than the tree leaning process. It was recognized since the earliest stages of the forest science development that an understanding of how tree crown is changed in competition for light is critical for forest growth predictions (Busgen & Munch, 1929; Reventlow, 1960). Tree crown area is naturally connected with total leaf surface, photosynthetic activity, carbon gain, and tree growth (Assmann, 1970; Smith et al., 1997). Crown competition is analyzed using crown class classification, individual tree zone of influence and by computing different competition indices. In forestry practice these methods are applied under the implicit assumption that trees grow vertically and the center of the zone of influence is also the center of tree growth. This is an important assumption in silviculture, since traditional foresters typically considered curved-trunk trees to be abnormal and unconditioned, and ignored them (Macdonald & Hubert, 2002; Westing & Schulz, 1965). Methods to measure such trees were also not developed (Grosenbaugh, 1981). The main objective of traditional silviculture to produce qualitative wood from well-formed trees (i.e., trees with symmetrical crowns and straight stems). Therefore for foresters tree leaning is in fact a problem which causes the development of bad-formed trees, rather than an important ecological property (Macdonald & Hubert, 2002). Typical planting and thinning regimes in silviculture significantly reduce the frequency of trunk inclinations (Assmann, 1970; Oliver & Larson, 1996; Smith et al., 1997). Only a few studies are concerned with adaptive trunk inclinations associated with the phototropism of the whole tree. In the beginning of the last century these patterns were described by German forester Arnold Engler (Engler, 1924). More recently, Loehle (Loehle, 1986) reported connections between trunk inclinations and the phototropism of the whole tree, based on data collected in Georgia and Washington State.

2.2 Gap dynamics and community-level patterns

Forest gaps are defined as small, localized disturbances, such as treefalls, which cause asynchronous local-forest regeneration processes (Oliver & Larson, 1996). In contrast, large-scale catastrophic disturbances, such as hurricanes or clearcutting, cause synchronized forest regenerations on the stand level. Gap dynamics is an important ecological process in which tree-plasticity patterns are exhibited (Fig. 2). Since forest gap dynamics constitutes a major process of regeneration, succession, and species coexistence (McCarthy, 2001; Ryel & Beyschlag, 2000), tree plasticity patterns can be associated with major trade-offs determining the strategies of trees.

Typically, large trees located at the gap border extend their crowns toward the gap (Hibbs, 1982) significantly reducing gap size and affecting canopy recruitment (Frelich & Martin, 1988). Gap closure, in turn, involves an interplay between two processes. The first process consists of lateral gap closure, brought about by crown encroachments of large trees at the

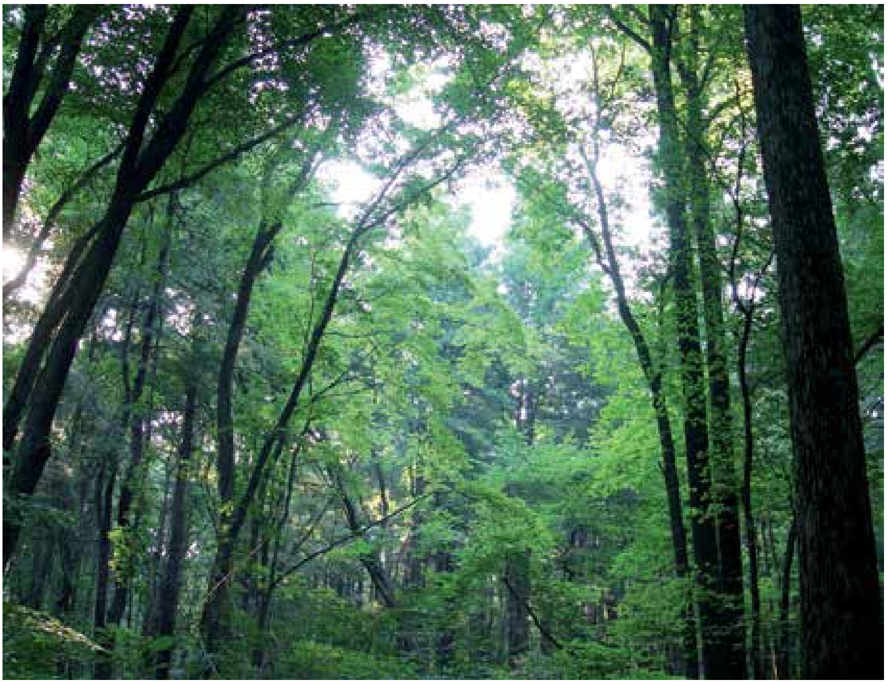


Fig. 2. A typical forest gap in the Institute for Advanced Study Woods (Princeton, NJ). Trees growing on the gap boundaries demonstrate plasticity patterns and phototropism, they modify their crowns and lean toward the gap.

gap borders and growth; the second process involves the crown development of small trees in the gap. The relative contributions of these two processes can be regulated by the gap size and species composition of both saplings and neighbor trees. To capture a small gap, saplings must be able to grow fast enough to compete with expanding crowns of dominant and co-dominant trees at the gap borders; in large gaps, by contrast, saplings have more opportunities to establish a canopy (Cole & Lorimer, 2005; Gysel, 1951; Webster & Lorimer, 2005; Woods & Shanks, 1959).

Individual tree plasticity leads to the development of a regular spatial canopy structure; in particular, crown centers are spaced more evenly than are the bases of plants. This pattern was reported for natural forest stands on Hokkaido (Ishizuka, 1984), and in the pure stand of *Atherosperma moschatum Labillardiere* (Monimiaceae) in Tasmania (Olesen, 2001), where crown-center distributions of all canopy were close to the uniform. Similar patterns were also discovered in the natural mature *Pinus sylvestris L.* forest in Eastern Finland (Rouvinen & Kuuluvainen, 1997); however, in that case the direction of crown asymmetry was strongly weighted in a southern and southwestern direction, which is the direction of most abundant solar radiation. It was suggested that in this forest, both factors, i.e., light competition with neighbors and phototropism toward the south, led to crown asymmetry and regular crown spacing patterns (Rouvinen & Kuuluvainen, 1997). Regular spacing of crowns in the canopy has been also established in computer simulations, where individual plants are able to exhibit adaptive crown plasticity (Strigul et al., 2008; Umeki, 1995a). However, forest simulators that does not include tree plasticity do not predict canopy regularity (Strigul et al., 2008).

2.3 Interspecific differences and cost of tree plasticity

Most tree species of different systematic and ecological groups demonstrate some tree plasticity patterns. It has been reported, for instance, for conifers (Loehle, 1986; Stoll & Schmid, 1998; Umeki, 1995b) and broad-leaf trees (Brisson, 2001; Woods & Shanks, 1959), in tropical (Young & Hubbell, 1991) and temperate forested ecosystems (Frelich & Martin, 1988; Gysel, 1951; Stoll & Schmid, 1998; Webster & Lorimer, 2005). At the same time, different tree species vary significantly in their ability to execute plasticity patterns; this raises questions concerning the different life histories and ecological strategies associated with tree plasticity and light competition. In particular, gap closure by crown encroachment of adjacent dominant and co-dominant trees was reported to be a typical process in the replacement of chestnut (*Castanea dentate* (Marsh.) Borkh.) by *Quercus prinus* L. and *Q. rubra* L. in the Great Smoky Mountains (Woods & Shanks, 1959). Northern red oak *Q. rubra* L. significantly surpassed yellow poplar (*Liriodendron tulipifera* L.) in its capacity for crown encroachment (lateral extension rates are 16.5 cm/year and 9.2 cm/year respectively) in Appalachian hardwood stands (Trimble & Tryon, 1966). The average lateral crown growth toward the small tree gaps of seven tree species in hemlock-hardwood forests in Massachusetts varied from 6 to 14 cm/year (Hibbs, 1982). *Quercus rubra* L. demonstrated the fastest lateral crown growth, with an average 14.03 ± 1.65 cm/year and a maximum 26.4 cm/year. The other six species were ranked according to their average lateral crown growth (in cm/year) toward the gap, as follows: *Betula papyrifera* Marsh. $10.87 \pm 1.39 >$ *B. lenta* L. and *B. alleghaniensis* Britt. $10.68 \pm 1.58 >$ *Tsuga Canadensis* (L.) Carr. $10.68 \pm 1.58 >$ *Acer rubrum* L. $8 \pm 0.72 >$ *Pinus strobus* L. 6.10 ± 0.94. Average annual crown lateral extensions toward the gaps of 13 tree species in the Southern Appalachians (Runkle & Yetter, 1987), varied from 8.6 cm/year (*Fraxinus americana* L.) and 13.1 cm/year (*Tsuga Canadensis* (L.) Carr.) to 31.4 cm/year (*Magnolia fraseri* Walt.) and 28.7 cm/year (*Acer rubrum* L.). Three species had shown a lateral extension rate of more than 20 cm/year (*B. alleghaniensis* Britt. 22.3 cm/year, *Liriodendron tulipifera* L. 21.8 cm/year, and *Acer saccharum* 20.8 cm/year Marsh.), and the other six broad-leaved tree species showed very similar rates of $17.1 - 18.8$ cm/year (Runkle & Yetter, 1987). This brief review demonstrates that lateral crown growth rate toward the gap can vary between the stand and tree species. While some species (for example, *Q. rubra* L.) typically demonstrate more plasticity than others, many species exhibit similar patterns, and some (for example, *T. Canadensis* (L.) Carr.) apparently have much less ability to extend their crowns toward the gap. These estimates are employed in the LES model (section 3.2)

German foresters' studies of the first half of the 20th century (see Engler (1924), Busgen & Munch (1929) p. 41, Assmann (1970) pp. 244, 284, 348 and subsequent references) found that conifer trees are less plastic than broad-leaved trees, which are capable of filling highly variable types of growing space. To account for these differences, it was suggested that broad-leaved trees, such as oaks and beeches, exhibit more phototropism than conifers, such as spruces and silver firs, which have "extremely energetic geotropism" (Assmann, 1970; Busgen & Munch, 1929). This conclusion is supported by the later studies (Loehle, 1986; Umeki, 1995b). It was suggested that the contrast plasticity patterns of conifers and broad-leave trees can be explained by the apical control differences (Loehle, 1986; Waller, 1986).

One important open problem is the lack of quantitative estimates of physiological traits associated with tree plasticity. Gravity is the universal force affecting tree form and growth. This force favors a vertical trunk and a symmetrical crown, which compose the typical form for an open growing tree. In this case the crown center of mass and the tree base are located on the same vertical line, which is the axis of tree symmetry. The execution of tree plasticity patterns, such as adaptive growth of branches and tree leaning, results in tree asymmetries and changes of the crown mass center that can make the tree less stable. Then, crown asymmetries and tree leaning should have some additional cost per tree compared to a symmetrical crown expansion (Busgen & Munch, 1929; Olesen, 2001). In a wet lowland tropical forest, tree asymmetry can increase the likelihood of the tree fall (Young & Hubbell, 1991). Tree anatomy studies and mechanical considerations show that the development of tree asymmetry causes stem tensions which should correlate with the development of additional structural tissues (Ford, 1992; McMahon & Kronauer, 1976). Umeki (Umeki, 1995a) assumed that the cost of tree asymmetry can be expressed by a reduction of tree height proportionally to the distance of the crown center movement. This assumption is also made in the Crown Plastic SORTIE and LES models (section 3.2). Loehle (Loehle, 1997) assumed that small trees with an elastic trunk can grow at an angle at practically no cost, and suggested that cost estimations are important only for large trees.

3. Scaling of vegetation dynamics: from individual trees to forest stands

The mainstream research approach in modern forestry is to use mathematical modeling in concert with experimental approaches. Certain limitations of experimental approaches make mathematical modeling especially useful. In particular, in experimental studies it is often necessary to concentrate on one focal level of organization while ignoring processes at other scales. Conclusive experimental results to support land-management decisions on different silvicultural techniques may not be obtained on a reasonable time scale and can be too expensive. Despite the availability of different forest models for use in either traditional forestry or in ecological studies, these models are often not suitable for ecological forestry. Forest yield tables is one of the oldest biological models with more than a 200-year history of development and practical applications to plantations with reduced tree competition (Mitchell, 1975; Shugart, 1984). However, forest yield tables is of an empirical nature and limited applications to more spatially heterogeneous silvicultural systems with intensive crown competition. Individual-based models (IBMs) simulating stand development emerged in the 1960s, when computer technology allowed for doing spatially-explicit simulations. Spatially explicit models can incorporate processes that occur at different scales and predict the dynamics of a forest by predicting each individual's birth, dispersal, reproduction and death and how these events are affected by spatial competition for resources with neighbors. Forest growth IBMs were developed in different directions. Foresters have developed stand simulators in order to estimate and optimize stand production; meanwhile, ecologists needed tools to study succession, species coexistence, and dynamics of indigenous forests. This difference in initial goals is reflected in the model structures, as forester and ecological models each concentrate on different aspects of forest development. Ecological models, such as the family of gap models originated from JABOVA include detailed descriptions of ecological processes which are considered to be most important, such as succession and gap dynamics (Botkin, 1993; Shugart, 1984). Forester IBMs, such as TASS (Mitchell, 1975), focus on overstory dynamics and on detailed descriptions of individual tree growth in the given neighborhood, which is important for plantations, ignoring seed production, gap and understory dynamics.

3.1 Individual-based forest simulators and tree plasticity

With respect to crown competition, individual-based forest simulators embody a wide range of assumptions. JABOVA-FORET models and many of their descendants, such as gap models, are based on the premise that a forest can be represented as a mosaic of homogeneous patches, i.e., gaps, each of which can be modeled independently. The size of every gap is usually assumed to be equal to the size of one large overstory tree. These patches have a horizontally homogeneous structure-i.e., the crowns of all trees in a gap extend horizontally over each patch (Botkin, 1993; Bugmann, 2001). The SORTIE model, descended from the JABOVA-FORET family, is a gap model in which trees in the gap have explicit spatial crowns (Pacala et al., 1996). The aboveground part of a single tree in SORTIE is represented as a rigid cylindrical crown, described by a species-specific radius and a crown depth around the vertical trunk, tree-plasticity patterns are not included (Pacala et al., 1996). This representation allows for the simulation of both light distribution in the canopy and tree growth in accordance with the availability of light, depending on local light heterogeneity.

Numerous individual-based stand simulators employ the zone of influence concept (Biging & Dobbertin, 1995; Bugmann, 2001; Mitchell, 1980), and crown competition is often accounted for by means of calculation of competition indices (Burton, 1993; Liu & Ashton, 1995). A zone of influence is usually defined as a circle around a tree center, where a focal tree can interact with its neighbors. This concept was used in studies of above-ground and below-ground competition (Aaltonen, 1926; Biging & Dobbertin, 1995; Casper et al., 2003). In the 19th century the term "crown ratio" was introduced to describe the ratio between d.b.h. and the average crown spread of a tree (Lane-Poole, 1936). This parameter was used as a stand characteristic reflecting the intensity of light competition in every crown class to optimize the thinning strategy, by reducing crown competition in silviculture practice (Krajicek et al., 1961; Lane-Poole, 1936). Later, the dominant-tree class was replaced by open-grown trees as the universal standard of trees which are not affected by their neighbors (Krajicek et al., 1961), and the crown area of open-grown trees was defined as a zone of influence for all trees with similar d.b.h. (Biging & Dobbertin, 1995). Comparison of zone of influences with realized dimensions yields different quantitative characteristics, the so-called "crown competition indices" (Biging & Dobbertin, 1995; Krajicek et al., 1961). Individual competition indices, calculated for a representative sample of trees from every crown class, can be averaged to produce a competition measure at the stand level. This scaling approach has several inherent limitations due to the static nature of competition indices, which restricts their usefulness in both practical silviculture and forest ecology (Burton, 1993).

The forest simulators employing the competition indices were united in a class of tree-stand models; in contrast to the crown-stand models (Mitchell, 1980), this old classification emphasizes the importance of simulating the crown and bole development. The crown-stand simulator TASS (Mitchell, 1969; 1975) employs the crown and bole as primary operating units. Crown competition in TASS is calculated as a result of the spatial intersection of the crown-profile functions of neighborhood trees. Similar crown competition algorithm was independently developed for modeling of *Eucalyptus obliqua* stands (Curtin, 1970). This modeling approach was employed in the Crown Plastic SORTIE (Strigul et al., 2008).

The next step in enhancing the realism of crown-plasticity representation is to explicitly simulate the growth of individual branches, instead of calculating a generalized crown-profile function. In 1980, K.J. Mitchell (Mitchell, 1980) included branch-stand models in the stand-model classification; however, such models were not yet developed, due to unrealistic computational resource demand. Technological progress made such models possible, and recent branch-level models were widely used in simulations of the development of form of individual plants and the simplest, evenly distributed, even-aged single-species stands (Godin, 2000; Takenaka, 1994). Several models have been developed to simulate the effects of crown plasticity caused by independent-branch development at the stand level. The WHORL model simulates a two-dimensional forest, where an open-tree crown is represented a system of horizontal disks, simulating a crown layer (Ford, 1992). Disks and their sectors can grow and die independently depending on local light availability in the stand. As a result, a tree crown in the stand develops as an asymmetrical system of whorls stacked along a central vertical axis, representing the tree trunk. A similar crown representation, using a pyramid of independently growing discs (which are also represented by independently growing segments), was employed in the BALANCE model (Grote & Pretzsch, 2002). The LES model (section 3.2) belongs to this group of models; as the next-generation model, it simulates indigenous forests with multiple species (typical simulations are 1000 years of 1 ha plots).

Stand models such as TASS, WHORL, and BALANCE as well as SORTIE and other gap models share a similar assumption concerning tree growth: In these models, trees are assumed to grow vertically, and the zone of influence is centered at the stem base. As a result, these models do not allow for tree leaning as a mechanism of adaptive tree-morphological plasticity. An alternative approach to simulate crown plasticity was developed by K. Umeki, using the crown-vector notion proposed by S. Takiguchi (see Umeki (1995a) for details and cross references). The crown vector is the vector between the stem base position and the centroid of the projected crown area of an individual tree. The centroid's coordinates were calculated using a competition index based on a circular zone of influence (Umeki, 1995a). This approach is also employed in the Crown Plastic SORTIE and LES models to simulate changes of crown center of mass (Fig. 4) .

This brief review demonstrates that a number of individual-based forest simulators vary in their attention to the tree-morphological plasticity patterns. Ecological models, such as SORTIE, describe tree growth in great detail as it relates to fine-scale resource heterogeneity and competition, seed production, and dispersion; however, they ignore both crown competition and tree plasticity. Crown-stand simulators such as TASS provide a detailed description of crown competition and of the underlying-branch plasticity patterns; they do not include ecological patterns, however, and they ignore tree leaning. Finally, the crown-vector approach (Umeki, 1995a) represents a simple and convenient method for simulating tree-leaning patterns.

The Crown Plastic SORTIE model (Strigul et al., 2008) combines the advantages of ecological and forest management IBMs considered above, and incorporates tree plasticity patterns. In particular it includes all the ecological complexity from the SORTIE model, a crown competition algorithm similar to the TASS model, and a crown plasticity algorithm based on the crown-vector approach. This IBM is suitable for predicting prescriptions of ecological forestry concerning management of multi-species and multi-age stands. This IBM gives more realistic predictions than the previous models; in particular, it allows for the observation of canopy regularity patterns emerging as a result of canopy self-organization (Strigul et al., 2008). At the same time, in more simplified simulations without crown plasticity algorithm, crown plastic SORTIE gives the same predictions as SORTIE or TASS depending on the model parameterization employed. This model also allowed derivation of tractable macroscopic equations for forest growth called the Perfect Plasticity Approximation (Strigul et al., 2008). The next generation individual-based model, LES, is introduced below.

3.2 The LES model

An individual-based forest simulator called LES (after the Russian word for forest) simulates spatially explicit tree competition above ground for light and below ground for water and nutrients. The LES model is based on the crown plastic SORTIE model, but operates at the individual branch and root levels (Fig. 3). Trees in the LES model execute phenotypic plasticity patterns considered in section 2. In this model, trees adaptively develop their crowns and root systems to their own unique local neighborhoods.

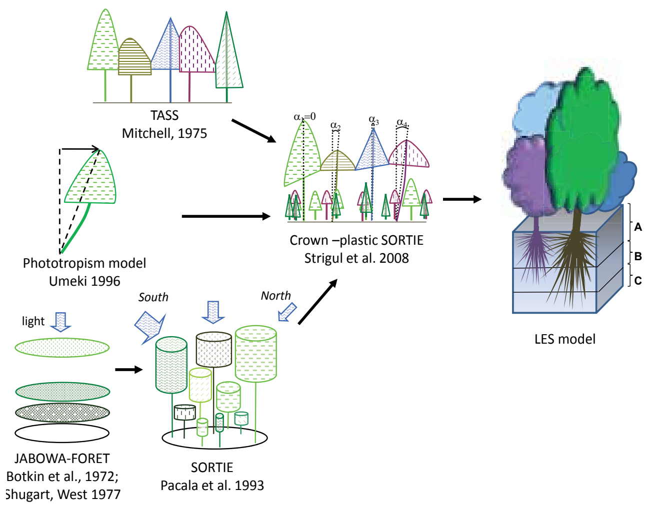
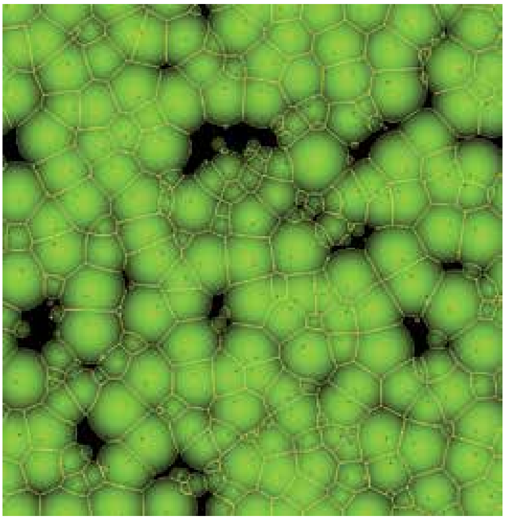


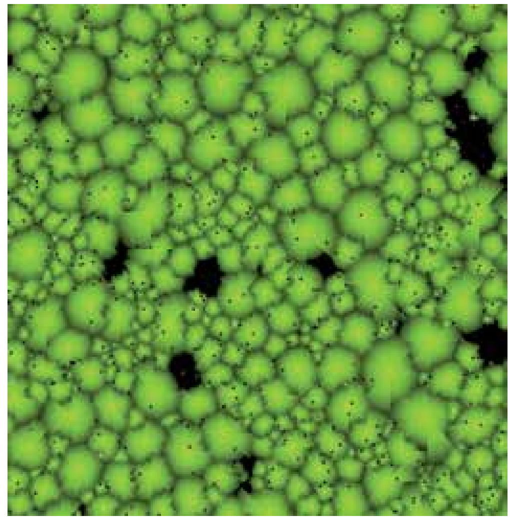
Fig. 3. Genealogy of the LES model

The most important new elements of the LES model compared to its predecessors (Fig. 3) are the following: 1) An individual tree develops a unique crown and root system within a local neighborhood to optimize spatial resource acquisition and allocation. 2) Vertical forest stratification emerges from the branch level competition. The model simulates the development of canopy, midstory and understory levels, allowing for tree classification as dominant, codominant, intermediate and suppressed trees. 3) Tree root systems are described by individual roots, and a vertical soil stratification emerges from individual root competition in three distinct soil horizons.

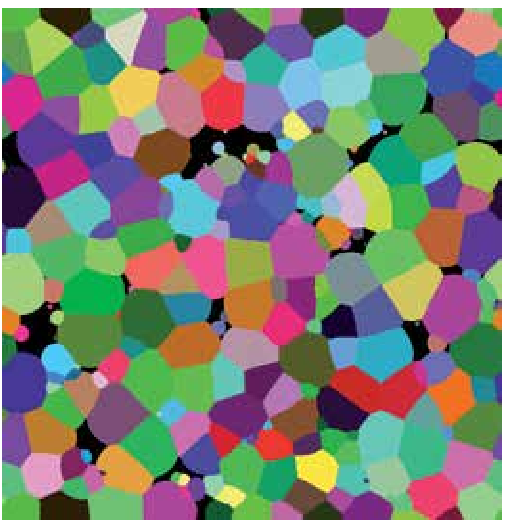
With respect to crown competition the Crown Plastic SORTIE model (Strigul et al., 2008) includes two essential elements: 1) Crown parametrization and competition algorithm similar to the TASS model, and 2) phototropism algorithm similar to one developed by Umeki (Umeki, 1995a). This model assumes that every tree has a species-specific potential crown shape, which is rotation-symmetrical about the vertical axis through the center of the crown. The realized tree crown is part of the potential crown determined by the spatial tessellation algorithm (Strigul et al., 2008). The advantage of this crown representation is that it leads to the computationally simple and fast algorithm as a horizontal cross section of the potential crown at any height is a circle. However, the major disadvantage is that the Crown Plastic SORTIE assumes the existence of a symmetrical potential crown shape for any tree growing within the forest stand. In the LES model this assumption is relaxed, and the individual crown shape develops as the result of adaptive tree growth within the unique local neighborhood. The new crown algorithm introduced in the LES model (Fig. 4) results in the development of a more realistic canopy than in the Crown Plastic SORTIE model.



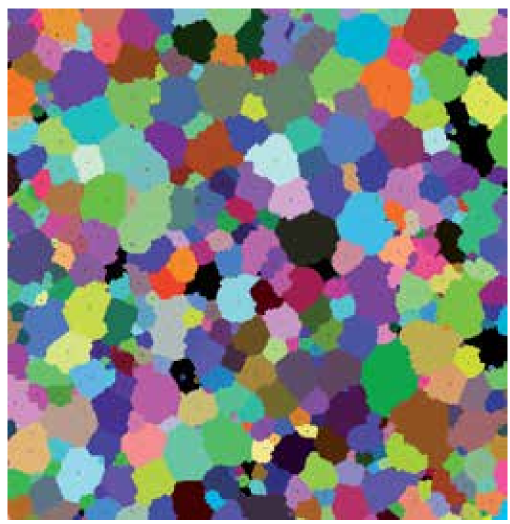
(a) Canopy simulated with the Crown Plastic SORTIE model.



(b) Canopy simulated with the LES model.



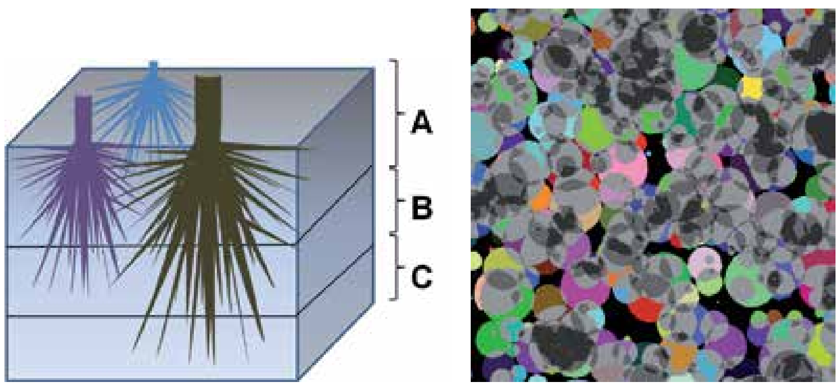
(c) the crown shape projection on the ground in the Crown Plastic SORTIE simulations.



(d) the crown shape projection on the ground in LES model simulations.

Fig. 4. Forest canopy simulations in the Crown Plastic SORTIE model (Strigul et al., 2008) and the LES model. In the Crown Plastic SORTIE model (a and c) a realized tree crown is determined as a part of the symmetrical potential crown by the tessellation algorithm (Strigul et al., 2008). In the LES model an individual tree crown develops as the result of adaptive spatial crown development within the unique local neighborhood. Tree crown develops on three hierarchical levels: leave and small branches, large branches (represented as independent spatial sectors) and crown level, represented by the crown center of mass. The figure demonstrates two different canopy visualizations: height-density plots (a and b) and crown ground projection plots (c and d) of canopy trees in a simulated White Pine forest stand (0.25 ha) 200 years after a major disturbance. The brightness level in figures a and b indicates the crown height at every point.

The LES model simulates a tree crown as a hierarchical three-dimensional spatial structure that develops and changes in response to environmental conditions on multiple levels. The LES model incorporates adaptive and random changes on three structural levels: 1) small branch and leaf level, 2) large branch level, 3) crown level. The first level of the crown organization in the LES model corresponds to leaves and small branch level, where every point represents an area of approximately 10 cm^2. Every such crown unit is represented as a point on a two-dimensional grid with the height of this crown component as a parameter. The canopy competition occurs independently at every point, where the highest crown wins the spatial competition. The second level of crown organization in the LES model is the level of large branches represented as independent crown sectors. Every sector is characterized by its width, the height of the lowest leaves and the leave distribution profile within the sector. These characteristics of every individual crown sector are determined by the results of spatial competition in the given neighborhood. The model can simulate crowns with 2^n sectors; most of the simulations are conducted with 8 crown sectors. The largest level of crown organization is the crown level determined by the center of the crown (center of mass); its position is determined by the algorithm of phototropism and crown leaning developed in the Crown Plastic SORTIE model (Strigul et al., 2008).



(a) Three dimensional root competition in the LES model.

(b) Root systems in the A soil horizon. 40 years after a major disturbance.

Fig. 5. Simulation of underground root competition in the LES model. A tree root system develops in three soil horizons: top (A), intermediate (B) and low (C). The soil within each horizon is represented as a collection of disjoint spatial units (cuboids), where each soil unit has its own available water content and can be occupied by roots of one or several trees competing for water and nutrients. An individual root system develops independently in different spatial directions corresponding to large roots (simulated as spatial sectors). Trees optimize water uptake by investing available resources in growth of the most efficient root sectors in different soil horizons.

The Crown Plastic SORTIE and all its ancestors focused entirely on the tree competition for light, and ignored below-ground competition for water and nutrients. In the LES model trees have spatial three-dimensional root systems and compete for water and nutrients (Fig. 5(a), 5(b)). Therefore tree growth and resource allocation can be simulated depending on multiple resource limitations (Fig. 6), and, in particular, carbon and water balance are considered at the tree level. The major patterns of belowground tree competition in the LES model are: 1)

The three independent soil horizons (A, B and C on Fig. 5(a)), 2) The spatially heterogeneous water/nutrient distribution within horizons, 3) Several trees can occupy every unit of soil, 4) Directional root growth within each horizon, 5) Individual trees optimize root system growth in three dimensions according to competition constraints and resource availability.

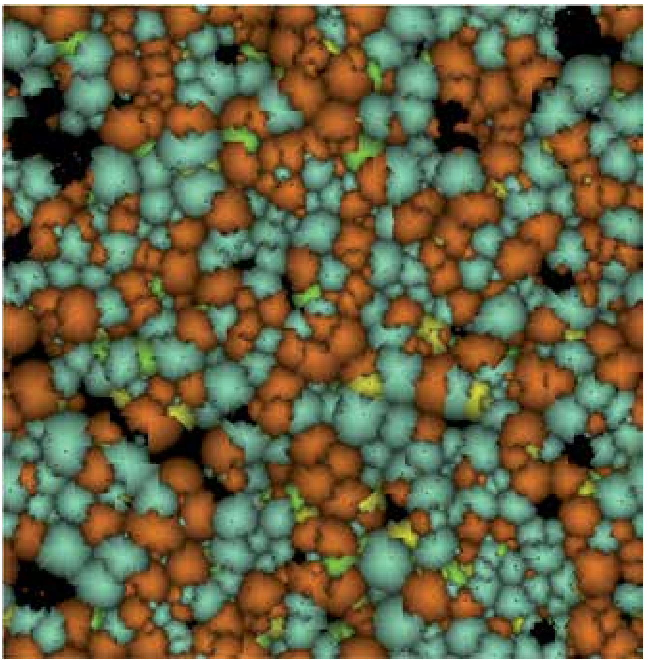


Fig. 6. Canopy level in the LES model simulation of stand development over 200 years, where trees compete for light and water simultaneously. Two different tree species that are colored grey and brown when trees are water-limited, and green and yellow when trees are light-limited, respectively. Most of the canopy trees are water-limited; only several trees with insufficient crowns are light-limited.

3.3 The Perfect Plasticity Approximation (PPA) model

Forest simulation models are effective tools in scaling individual-level spatio-temporal processes to the stand level because they are able to simultaneously incorporate tree ecophysiological traits such as carbon allocation, and capture tree level disturbances and gap dynamics. Individual-based models can also be applied to simulate vegetation dynamics at the landscape level using GIS-based inputs. The major disadvantage of forest individual-based models is that these spatial stochastic processes are not analytically tractable, so their general properties and sensitivities to the choice of parameters and functional forms are uncertain. However, analytically tractable approximations of individual based forest simulators can be developed. In particular, the Perfect Plasticity Approximation (PPA, Strigul et al. (2008)) is a recently developed model predicting the stand-level forest dynamics by scaling up individual-level processes. The PPA offers good predictions for 1) stand-level attributes, such as basal area, tree density, and size distributions; 2) biomass dynamics and self-thinning; and 3) ecological patterns, such as succession, invasion, and coexistence. The model includes a system of von Foerster partial differential equations and the PPA equation.

Unlike the individual-based simulator, the PPA model is both analytically tractable and computationally simple. Initially the model was developed as an approximation of the crown plastic SORTIE model (Strigul et al., 2008), but it was also demonstrated that the PPA model captures the dynamics of the temporary forests. Purves at al. (Purves et al., 2008) estimated the parameters of the PPA model by using the data collected by the Forest Inventory and Analysis (FIA) Program of the U.S. Forest Service (FIA data) for the US Lake states (Michigan, Wisconsin, and Minnesota). It was demonstrated that the PPA model, applied even in its simplest form, carefully predicts forest dynamics and succession on different soil types.
The PPA model is a cohort model assuming time is discrete and is the following boundary value problem if the time is measured continuously (Strigul et al., 2008). The continuous version of the PPA model for m tree species consists of m von Foerster equations (1) with initial $N_i(s,0)$ and boundary conditions (2) for every species $i = 1, \dots, m$ connected by the integral PPA equation (3) for the threshold canopy size $s^*(t)$:

$$\frac{\partial N_i(s,t)}{\partial t} = -\underbrace{\frac{\partial\left(G_i(s,s^*(t),t)N_i(s,t)\right)}{\partial s}}_{growth} - \underbrace{\mu_i(s,s^*(t),t)N_i(s,t)}_{mortality}, \tag{1}$$

$$N_i(s_{i,0},t) = \int_{s_{i,0}}^{\infty} N_i(s,t)F_i(s,s^*(t),t)ds / G_i(s_{i,0},s^*(t),t), \tag{2}$$

$$1 = \sum_{i=1}^{m} \int_{s^*(t)}^{\infty} N_i(s,t)A_i(s^*(t),s)ds, \tag{3}$$

where i indicates one of m tree species, s is the size of the tree that can be either tree height or dbh connected with height by a species specific allometric equation, $N_i(s,t)$ is the mean density of individuals of species i of size s at time t, $G_i(s,t)$ is the growth rate of these individuals i.e., $ds/dt = G(s,s^*(t),t)$), $\mu_i(s,s^*(t),t)$ is their death rate, $F_i(s,s^*(t),t)$ is their fecundity, $A_i(s^*(t),s)$ is the crown area function that gives the area of the crown at s, and $s_{i,0}$ is the size of a newborn of the ith species. Growth, death and fecundity functions depend on time t and tree size s as well as the canopy threshold level $s^*(t)$.
Strigul et al. (Strigul et al., 2008) considered transient and stationary regimes of tree monocultures as well as simple invasion and coexistence problems. The model was parameterized for different soil types so the patch (stand) dynamics at different soil and forest types types can be considered separately.

4. The Matreshka model: hierarchical scaling of forest dynamics to the landscape level.

This section introduces a modeling framework, called Matreshka (after the Russian nesting doll), for the scaling of vegetation dynamics from the individual level to the landscape level through the ecosystem hierarchical structure (Figure 7, see also Strigul et al. (2012)). The Matreshka model is a particular realization of the hierarchical patch dynamics concept (Levin & Paine, 1974; Wu & Loucks, 1996) in application to forested ecosystems. The model (Fig. 7) represents forest dynamics at the landscape level as an interference of separated processes occurring at different spatial and temporal scales: 1) within forest stands dynamics caused by individual-level disturbances, and 2) dynamics of the mosaic of forest stands caused by large disturbances. The Matreshka model can be presented as a continuous or a discrete model, where partial differential and integral equations and Markov chains are employed,

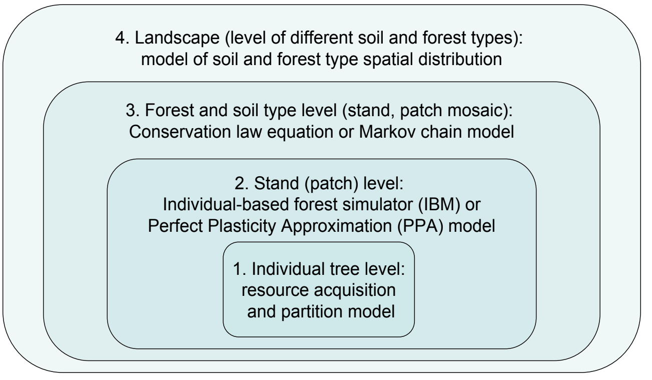


Fig. 7. The Matreshka framework for hierarchical scaling of vegetation dynamics to the landscape level

respectively. The highest hierarchical level is the landscape level comprising a mosaic of different soil and forest types. The vegetation dynamics at this level are the composition of vegetation dynamics of different forest types. The forest and soil type level consists of the mosaic of forest patches that are in different successional stages. In this model, forest patches are considered as spatial units of a considerably large size (0.5 - 1 hectare). A broad discussion of the model assumptions can be found in a recent paper (Strigul et al., 2012) focusing on the dynamics of forest stands (level 3 on Fig. 7). The Matreshka employs previously developed models for the processes at smaller scales. In particular, tree dynamics within the forest stands can be modeled by an individual-based forest growth model (for example, SORTIE, Crown Plastic SORTIE or LES models) or by forest growth macroscopic equations, specifically, the Perfect Plasticity Approximation model (PPA). The individual tree level model captures growth, mortality, and reproduction of individual trees depending on tree size, light and nutrient availability, soil type, and other factors. Several empirically determined parameters approximate these individual-level processes in the SORTIE and PPA frameworks (Pacala et al., 1996; Strigul et al., 2008).

At the next step of scaling, age-structured dynamics of forest stands (patches) on the given soil type can be described by the conservation law following Levin and Paine (Levin & Paine, 1974) in the continuous case:

$$\frac{\partial n(t,a,\xi)}{\partial t} = -\frac{\partial n(t,a,\xi)}{\partial a} - \frac{\partial g(t,a,\xi)\,n(t,a,\xi)}{\partial \xi} - \mu(t,a,\xi)\,n(t,a,\xi), \tag{4}$$

where, $n(t,a,\xi)$, $g(t,a,\xi)$ and $\mu(t,a,\xi)$ are density, mean growth rate and extinction rate of a stand of state a and size ξ at time t, correspondingly. The initial stand distribution $n(0,a,\xi)$ and the "birth rate" of new stands should be specified to simulate given forested ecosystem.

The original model operates with two variables (patch age, a, and size, ξ), but it has been indicated (Levin & Paine (1974) p. 2745) that age is just one of the possible "physiological" variables. In this chapter, we consider a special case of equation (4), where the forest patches are fixed in size, so the rate of patch growth is zero $g(t,a,\xi) = 0$. Variable a is considered as a successional stage of forest stand, and is discussed in another paper (Strigul et al., 2012). This formulation of the Matreshka model (equations 1-4) is analytically tractable in special cases, though the general analysis is a significant challenge. In particular, in the following example we consider the stationary distribution of tree monoculture stands.

In the discrete case Strigul et al. (Strigul et al., 2012) proposed a discrete time Markov chain model for stand (patch) dynamics that can be easily generalized to a continuous time framework by taking random times between transitions. However, the discrete modeling approach has certain advantages such as that the transition of stands between stages can be explicitly defined, the probability matrix is easy to interpret and estimate using forest inventory data. In the general Markov chain model for the stand transition (Strigul et al., 2012), the states in the Markov chain are represented by stand successional stages $\{1, 2, \ldots, m\}$ characterizing the forest stand development up to a certain maturity stage m. In certain applications, such as forest fire models, the successional stage is characterized by the absolute stand age, i.e., the time since the latest major fire disturbance. However, in general, the choice of the parameter characterizing stand successional stage can be a challenging problem. The model for development of one stand (patch) may be represented using a graph as in Figure 8 and is described using a general transition probability matrix (5):

$$P = \begin{pmatrix} r_1 & p_1 & 0 & 0 & \dots & 0 & 0 \\ q_{2,1} & r_2 & p_2 & 0 & \dots & 0 & 0 \\ q_{3,1} & q_{3,2} & r_3 & p_3 & \dots & 0 & 0 \\ \vdots & & & \ddots & \ddots & & \\ \vdots & & & & \ddots & \ddots & \\ q_{m-1,1} & q_{m-1,2} & q_{m-1,3} & q_{m-1,4} & \cdots & r_{m-1} & p_{m-1} \\ q_{m,1} & q_{m,2} & q_{m,3} & q_{m,4} & \cdots & q_{m,m-1} & r_m \end{pmatrix} \tag{5}$$

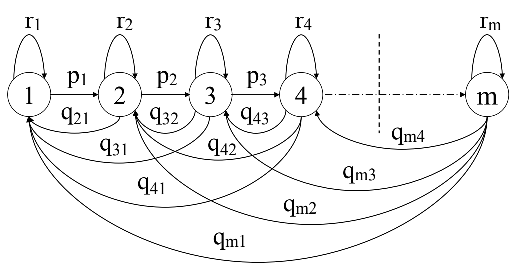


Fig. 8. A graph of the complete stage dynamics model of forest stands (after Strigul et al. (2012)).

The model assumes that the patch (forest stand) is observed frequently enough relative to the succession process so that the forest does not grow through two consecutive successional states. Each time the forest stand moves to the next stage with probability p_i or stays at the same stage with probability r_i (due to some minor forest disturbances or a small interval between forest inventories). The $\{q_{ij}\}_{i\in\{2,\ldots,m\},j\in\{1,\ldots,m-1\}}$ probabilities describe disturbances affecting stand succession. The disturbances include disaster events which completely destroy forest stands ($q_{x,1}$, $x = 2, \ldots, m$) or smaller-scale events which change the stand successional stage to one of the previous stages with certain probabilities ($q_{h,k}$, $h > k > 1$). These disturbances determine the development of forest as a mosaic of patches (stands). The model makes no distinction or explanation between the causes of the disturbances leading to the successional stage, in particular, both silvicultural operations such as forest harvesting or natural disturbances would lead to larger $q_{i,j}$ probabilities (Strigul et al., 2012).
The Matreshka model is considered as a first step in development of an analytically tractable model capable of capturing the forest dynamics on multiple scales. However, the PPA (1-3)

model and the forest stand model (7) as well as its discrete counterparts such as Markov chain models (8), are only partially analytically tractable. In particular, the stationary states of these models and their stability can be relatively easily investigated, while the transient dynamics is a challenging problem. Therefore, we are still far away from complete mathematical theory of multiscale forest dynamics.

The key element for the Matreshka model is to simulate forest dynamics as a patch-mosaic phenomenon at two distinct hierarchical scales: at the individual level and the stand level (7). In forest ecology the two focal scales (i.e. individual and stand levels) have been broadly discussed with respect to forest dynamics and disturbance regimes (Bragg et al., 2004; Strigul et al., 2012). Patch-mosaic dynamics of larger forest units (stands) have also been considered in different studies, such as in forest fire models, forest disease models, and anthropogenic disturbance modeling (Bragg et al., 2004; Forman, 1995; Wu & Loucks, 1996). In the Matreshka model, we use the PPA model to scale up gap dynamics to the stand level and consider forest patches as much large spatial units (about 0.5-1 ha, see (Strigul et al., 2012) for more details).

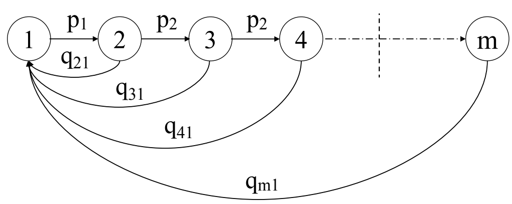


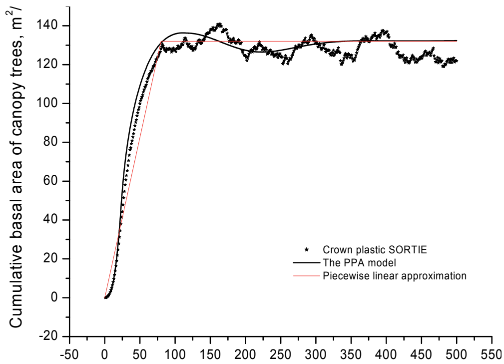
Fig. 9. A graph for a simplified forest stand model. The Birth and Disaster Markov chain.

4.1 Fire disturbance model: a case study

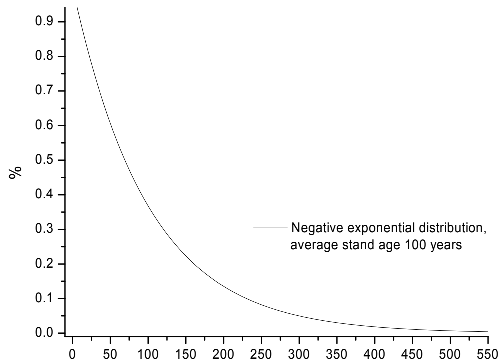
In this example, an analytically tractable case of the Matreshka model is considered. The simple case of the PPA model-the flat-top model-is employed to describe a tree monoculture stand (Strigul et al., 2008). The flat-top model was parameterized and validated for the Lake states (Purves et al., 2008). This model is a special case of the PPA model (equations 1-3) where tree growth and mortality are characterized by several species-specific constant parameters such as understory and overstory rates of growth as well as mortality and fecundity parameters. Using these simplest possible functional forms makes the model analytically tractable (Strigul et al., 2008). In particular, there exists a unique stable stationary state of a flat-top monoculture stand, and stationary age and size distributions of trees within the stand can be calculated. The transient dynamics are less tractable; however, the self-thinning exponents were analyzed analytically (Strigul et al., 2008), and a good approximation of the total length of transient period (t^*) for the case of the invasion into an empty habitat was derived (unpublished results). The length of the transient period curve corresponding to the stand development, starting from the invasion into an empty habitat until the stationary state, may be approximated by a piecewise linear model:

$$x(t) = \begin{cases} \alpha\, t, & t \le t^*, \\ x^*, & t > t^* \end{cases} \tag{6}$$

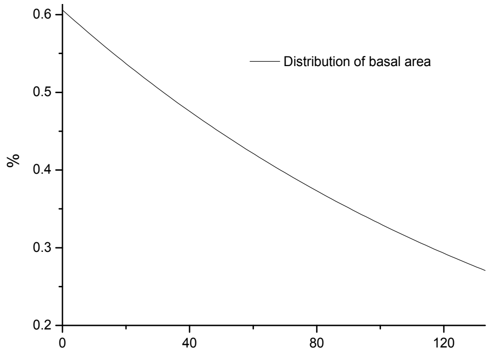
where $x(t)$ is a stand characteristic (such as biomass or cumulative basal area), x^* - stationary value of the quantity $x(t)$, t^* is the length of the transient period, and a parameter α is x^*/t^*, so it can be determined if the values x^* and t^* are known. This piecewise-linear approximation is commonly used in microbiology to approximate sigmoidal growth in microbial cultures. Sigmoidal growth models, for example, Gompertz and logistic curves, are often used to describe growth of stands and individual trees as well as microbial cultures (Dette et al., 2005; Yoshimoto, 2001).



(a) Dynamics of the cumulative basal area of the hypothetical stand of white pine simulated by the crown plastic SORTIE model (black points) and the PPA model (black line) (see Strigul et al. (2008) for the details and parameter values), the red line is a piecewise linear approximation.



(b) The stationary stand age distribution of the mosaic of forest patches represented by the negative exponential distribution in the forest fire model (after Van Wagner (1978)).



(c) The stationary distribution of stand basal area of the mosaic of forest patches.

Fig. 10. An example of the fire disturbance model for a tree monoculture.

We consider a special case of equation (4) to describe the stand level dynamics of tree monoculture. In particular, we assume that the stands are fixed in size, i.e. $g(t, a, \xi) = 0$ and have constant extinction (disaster) rate μ, to obtain the following model:

$$\frac{\partial n(t,a)}{\partial t} = -\frac{\partial n(t,a)}{\partial a} - \mu\, n(t,a). \tag{7}$$

This model describes the patch-mosaic pattern of stands, given some initial stand distribution $n(0, a)$ and assuming that new stands emerge in place of extinct stands. The discrete version of equation (7) is a birth-disaster Markov chain with constant parameters p and q (Fig. 9). This model, with the successional stage a considered as stand age, is mathematically equivalent to the classical forest fire model developed by Van Wagner (Van Wagner, 1978). It is a simple mathematical exercise to show that the model (7) has a stable stationary distribution described by a negative exponential law which after standardization can be presented as the negative exponential distribution with the following probability density function:

$$f(a, \mu) = \begin{cases} \mu e^{-\mu a} & a \geq 0, \\ 0, & a < 0. \end{cases} \tag{8}$$

The negative exponential distribution as well as its discrete version - the geometric distribution are employed in forest fire models to describe stationary age distributions of forest stands (Johnson & Gutsell, 1994; Van Wagner, 1978).

Using the Matreshka framework, we can now scale up the predictions of the PPA model to the level of mosaic of forest stands. We can invert equation (6) as a function of $t(x)$ on an interval $[0, t^*]$ and there are infinitely many values of t corresponding to the value x^*. Substituting this result in equation (8) we obtain the stationary probability distribution of the quantity x:

$$f(x, \mu, \alpha) = \begin{cases} \mu\, e^{-\frac{\mu x}{\alpha}} & 0 \leq x < x^*, \\ \left(1 - \int_0^{x^*} \frac{\mu}{\alpha}\, e^{-\frac{\mu x}{\alpha}} dx\right) \delta(x - x^*), & x = x^*, \\ 0, & x < 0 \text{ and } x > x^*, \end{cases} \tag{9}$$

where $\delta(x)$ is the Dirac delta function that accounts for all the stands which are in the stationary state. In the discrete case, the geometric distribution may be considered instead of distribution (8). In that case, the transformed distribution corresponding to (9) will have only a finite number of values. The coefficient for the resulting Dirac delta function in (9) will be the last value corresponding to x^* in this distribution.

As an illustrative example, we consider a stand of white pine (*Pinus strobus*) simulated by the crown plastic SORTIE and the corresponding PPA model. Figure 10(a) presents the simulation results (reproduced with permission from (Strigul et al., 2008)). The model functional forms and parameter values are available in the latter reference. In this example, the parameter $x(t)$ is a stand cumulative basal area; however, biomass, average canopy height etc. may be employed instead. Figure 10(b) illustrates the negative exponential distribution of stand ages corresponding to the stationary state of equation (7) with $\mu = 0.01$. This parameter value corresponds to an example considered by Van Wagner in his classical work on forest fire modeling (Van Wagner, 1978). Figure 10(c) presents the distribution (9), where 44.93% of all stands have the stationary state basal area $x^* = 132\, m^2/ha$. Note that the shape of the distribution (9) is determined by the values of μ, x^*, and t^*. Therefore, the stationary distribution (9) predicted by this simple modification of the Matreshka model may be observed and verified subject to data availability.

This example is based on the tree monoculture model (Strigul et al., 2008) and therefore is of limited practical value for the SFM of indigenous multispecies forests. However, even this simplified model can be implemented directly for certain forest types that are naturally dominated by one tree species. One particular example is the longleaf pine (*Pinus palustris* Mill.) forest that has historically dominated the Southeastern United States. This natural monoculture ecosystem was supported by forest fires, as the longleaf pine is fire resistant. Development of its competitors, such as loblolly (*Pinus taeda*) and slash (*Pinus elliottii*) pines, has been limited by frequent forest fires. Over the last 150 years the landscape has changed radically due to overexploitation and fire suppression. Intensive longleaf pine forest restoration projects at the Southeastern U.S. are currently on-going within the SFM framework (www.longleafalliance.org).

This example demonstrates the potential advantages of using the multiple-scale modeling for SFM applications. The Van Wagner fire-disturbance model operates with the stand age after a major disturbance (Van Wagner, 1978). Therefore, the forest management plans within this model should be based on the fire-disturbance history. In practice, the exact fire history is often hard to determine. Forest surveys, such as USDA FIA data and Canadian forest service data, determine the stand age empirically as an average age of canopy trees. This parameter is unfortunately not very reliable for modeling purposes (Strigul et al., 2012). The Matreshka model allows one to develop the forest management plans using forest stand stratification with respect to the stand successional stage, basal area, or stand biomass. The stand biomass or basal area can be easily calculated using available survey data (Strigul et al., 2012) and the forest management plan can be designed based on these stand characteristics. This makes the model suitable for the needs of criterion 1.1 (Ecosystem diversity) of the Montreal process.

4.2 Application of the Matreshka model to criterion 5 of the Montreal process

The Matreshka model is developed for ecological forestry and SFM applications. Specifically, it allows one to incorporate natural and anthropogenic disturbances occurring at different scales, ranging from individual trees to stands to predict forest growth at the landscape level. To address criteria 2 and 5 of the Montreal process, the model can naturally incorporate effects of climate change on individual tree growth through modification of either the forest individual-based model or the PPA model. Changes of the natural disturbance regime due to climatic factors can be incorporated by modification of tree mortality functions or by changing elements and structure of the transition matrix 5 (for stand-level disturbances). Similarly, changes in forest policy, silvicultural practices, and anthropogenic disturbances can also be incorporated in the model through modification of tree mortality functions and the transition matrix 5. While the Matreshka model is formulated as a non-spatial model at the stand level, the model can also be presented in a spatially explicit form by using GIS-based simulations of forest stands at the landscape level. This can be essential if the forest stewardship in the focal area varies due to different landowner policies.

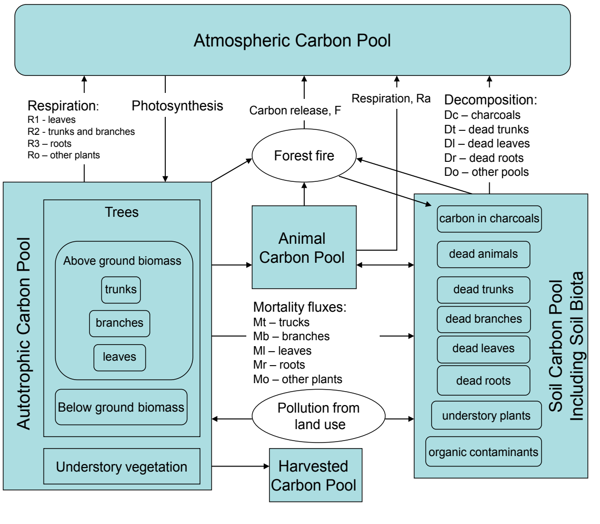


Fig. 11. The framework for modeling of forest carbon footprint for the SFM applications. The Matreshka model is used for the modeling of Autotrophic and Soil Carbon Pools.

Current on-going research is focused on the application of the Matreshka model to the carbon cycle modeling of temperate forests in the North-Eastern Part of the USA and Quebec in agrement with criterion 5 of the Montreal process "Maintenance of forest contribution to global carbon cycles". The carbon footprint of forest ecosystems is determined by the dynamics of carbon sequestration and release, and can be affected by harvesting and other anthropogenic activities. In this project, the Matreshka model is used to predict the forest carbon cycle according to a conceptual model presented in Figure 11. Most of the carbon influx into the ecosystem is derived from photosynthetic assimilation of atmospheric CO_2 by the autotrophs (overstory trees, understory trees, shrubs, and groundcover vegetation) that determine the gross primary productivity (GPP). The major effluxes of carbon in the atmosphere occur as the result of autotrophic respiration (which is defined as the sum of maintenance respiration and growth respiration), heterotrophic respiration, and the processes of physical decomposition of organic matter, such as fire. Carbon is also removed from a forest ecosystem by wood harvesting. The typical parameters of interest in calculating carbon footprints are the net primary production (NPP, defined as GPP minus autotrophic respiration), and the net ecosystem production (NEP). The NEP is determined as the net exchange of CO_2 between the atmosphere and ecosystem, which is measured on an annual basis, and equal to the NPP minus heterotrophic respiration. During recent decades, carbon fluxes presented in Figure 11 were evaluated, however, the current models operate with the carbon balance at the macroscopic level using average estimates of the carbon pools and fluxes. In this project, the key element for predicting forest carbon cycle is the Matreshka model. This model provides a scaling of carbon balance from an individual tree level to the stand level, and simulates the autotrophic carbon pool (Fig. 11). Therefore, the carbon balance model incorporating the Matreshka model scales up the effects of the silvicultural practices and other anthropogenic activities from the individual tree-based level to the ecosystem level, and can predict changes in the forest structure and carbon dynamics at different time horizons.

5. Conclusion

In this chapter the framework of complex adaptive systems is employed to address the basic challenge of the ecological forestry and SFM, i.e., to understand and predict how natural and anthropogenic disturbances occurring at different scales propagate through the forested ecosystems and affect forest structure and dynamics. This framework naturally combines experimental and theoretical approaches. This framework consists of three major components: 1) the development of individual-based models (IBMs) to simulate multiple scales processes in complex systems, and their parameterization with experimental data (in particular, by using USDA forest inventory data, FIA); 2) the development of different scaling methods that approximate individual-based processes; and 3) validation with real data and practical applications. The first component involves mostly computer simulations of what are, in general, analytically-intractable stochastic processes. Forest growth IBM can serve as an intermediate research step in the derivation of macroscopic equations (i.e., tractable analytic models approximating this stochastic process), and, as an independent research tool, to simulate forest carbon balance, stand dynamics, natural disturbances (such as disease outbreaks), and the outcomes of silvicultural prescriptions. Scaling methods may allow models to be reduced to analytically tractable objects, macroscopic equations-such as stochastic and deterministic dynamical systems-which are both more robust in their predictions and, also, computationally simpler. Recently developed models including the Crown Plastic SORTIE, LES, and PPA have been developed within this research framework to address the scaling of vegetation dynamics from the individual to the stand level. All these models employ individual tree plasticity as a crucial factor for canopy development and forest self-organization within the stand level. The Matreshka model generalizes these models operating on the individual level for scaling of vegetation dynamics to the landscape level using the hierarchical patch dynamics concept. It is anticipated that these new modeling tools will be employed for the SFM of indigenous forests. Practical applications of the developed modeling approach address criteria 2 and 5 of the Montreal process. The ongoing research focuses on the modeling of the temperate forest carbon cycle in the North-Eastern USA and Quebec. The Matreshka modeling framework can help natural resource managers to understand how changes in forest management practices can affect the forest carbon footprint, and to manage the key ecosystem processes that control carbon and nutrient dynamics in a forest ecosystem.

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21

Decision Support Systems for Forestry in Galicia (Spain): SaDDriade

Manuel Francisco Marey-Pérez, Luis Franco-Vázquez,
and Carlos José Álvarez-López
Universidad de Santiago de Compostela
Spain

1. Introduction

Ever since they were created in the 70s, Decision Support Systems (DSS) have been a great source of help with different management problems such as the optimization of travel times in airlines or train companies, medical diagnosis, business management, natural resource management, agriculture and forestry. Forest planning uses forest simulators that usually include growth and performance models in order to generate the different alternative management programs which will give rise to different production processes and programs. The selection of the best choice according to predefined criteria, which are generally related to the number of alternative management options, production programs and assessment criteria, require the use of optimization methods. These optimization methods range from whole linear programming, goal programming to heuristic methods, among which tabu search, genetic algorithm and simulated annealing are worth noting.

1.1 Development and current situation of forest DSS

In recent years, the aims for the development of forest DSS have changed. They used to have only one objective, which was to provide information about: Site index for reforestation (Hackett and Vanclay, 1998); soil fertility (Louw and Scholes, 2002); habitat requirements (Store and Jokimäki, 2003); tree growth (Hackett and Vanclay, 1998); forest management (Kolström and Lumatjärvi, 1999); wildfires (Kaloudis 2005 y Bonazoutas et al. 2008); trees brought down (Mickouski et al. 2005; Olofsson and Blennow, 2005; Zeng et al. ,2007); seed bank long-term planning (Nute et al. 2005 y Twery et al. 2005); river flow and its relation with trees (MacVicar et al., 2007) and profits (Huang et al. 2010).
More recent DSS methods have several aims. the management planning problem is focused on two or more objectives, some of which may be in conflict (Stirn, 2006). Næsset (1997) states the need for new tools to help planning aims related to biodiversity and wood production. The needs of sustainable forest management must be considered and included in the design of DSS for forestry (Wolfslehner and Vacik, 2008). Moreover, forest management actions cannot be considered in isolation or with just one objective in mind, despite the difficulties of integrating them spatially and temporally (Kangas and Kangas y MacMillan and Marshall, 2004). The aim is to offer a general view of alternative focus to face the uncertainty from the perspective of forestry and natural resource and ecosystem management (Rauscher (2000).

Some remarkable systems developed with multiple aims in forestry and natural resources are those developed for: hydrographic basins in Australia by Bryan and Crossman (2008); carbon in Canada by Kurz et al. (2009); plague control in Poland by Strange et al. (1999); landscape management in South Carolina by Li et al. (2000); visualization tool for landscape valuation by Falcão et al. (2006).

The prediction of forest activity future behaviour in its multiple dimensions is inherent to the main aim of forest DSS, therefore, the temporal scale is specially important in the development of this type of applications. Temporal scales are classified into three types: strategic, tactic and operational.

Long-term management planning or strategic is that which has a planning horizon of more than 15 years. Potter et al. (2000) state that forest ecosystem management implies the need to forecast the future state of complex systems, which often experience structural changes. It is by means of strategic planning that ecological integrity and sustainability (Gustafson y Rasmussen, 2002), risk management (Borchers, 2005 y Heinimann, 2010) and future landscape (Aitkenhead and Aalders, 2009) are guaranteed. Næsset (1997) stresses the importance of the integration of GIS with quantitative models for long term forest management. Some interesting examples of strategic forest DSS are those presented by Boyland et al. (2006) for a planning horizon of 250 years; Wolfslehner and Vacik (2008) for 120 years; Díaz-Balteiro and Romero (2004) for 100 years divided into periods of ten; Baskent et al. (2001) for 85 years; Huth et al. (2005) for 60 years and Lasch et al. (2005) for 50-year simulations.

In the case of strategic planning, it is not only the temporal scale that is higher, but also the spatial scale (Kangas and Kangas, 2005).

When the planning period is between 1 and 15 years long, it is called tactical planning or mid-term. Kangas and Kangas (2005) point that in tactical planning the number of alternative forest plans can be considered infinite. Different examples of this type of planning are those presented by Anderson et al. (2005) and Snow and Lovatt (2009). Anderson et al. (2005) present FTM (Forest Time Machine), which simulates the development of a forest area and calculates the stand development in five-year intervals. Snow and Lovatt (2008) examine the use of the general planner for agro-ecosystem models (GPAM) in pasture rotation length, building a decision tree.

Short-term or operational planning is that whose planning period lasts for a maximum of a year. Acuña et al. (1997) remark the usefulness of transparent, operative, easily validated processes provided by experts. Ducey and Larson (1999) state that sustainability assessment requires a careful balance between short-term and long-term goals. Mowrer (2000) considers that the temporal scale on which the operative tool or DSS works will have an effect on the level of uncertainty of the analysis. Uncertainly is lower in short-term planning. The more empirical the models, the more accurate they will be (Porté and Bartelink, 2002). Some remarkable examples of operative forest DSS are: Vacik and Lexer's (2001) which assesses nine species mixes and seven multiobjective regeneration methods at stand level. Newton (2003) tabulates the annual management of spruce plantation. Thomson and Willoughby (2004) present a web system of forest management consultancy. Kurz et al. (2009) comment a version on operative scale of the model of carbon-dynamics. Newton (2009) shows the usefulness of the modular-based structural stand density management model (SSDMM) for decision-making in operational management.

Another relevant aspect in forest DSS analysis is the spatial scale of the unit of analysis. The highest level is on a regional or, in some cases, national scale. In this type of works planning is strategic and establishes the guidelines (Anderson et al. 2005; Ascough, 2008; Carlsson et al. 1998; Crookston and Dixon, 2005; Heinimann, 2010; Kurz, 2009; Mathews,1999; Mowrer, 2000, Potter, 2000; Maitner et al. 2005; Nute, 2005; Reynolds, 2005; Thompson et al. 2007). These works have been developed mainly in the USA and in different regions in Scandinavia.

A second scale is the lanscape or forest that has been the unit of analysis for the studies carried out in the USA (Rauscher in 1999, Twery and Thomson et al. in 2000; Twery and Hornbeck in 2001, Bettinguer et al. 2005; Borchers, 2005; Gärtner et al. 2008 and Graymore et al. 2009). In Scandinavia some remarkable studies are those by Anderson et al. 2005; Kurttila, 2001; Leskinen et al. 2003; Store and Jokimäki, 2003, among others. In other countries, the works by Seely et al. 2004; Stirn, 2006 and Wang et al., 2010 are worth noting. It is at this scale when the need to integrate GIS within DSS arises. It should be easy for regional managers to carry out forest zoning effectively in the areas where initiatives are necessary for sustainable progress (Martins and Borges, 2007). It is also necessary to integrate 3D visualization tools (Falcão et al., 2006).

The main planning scale so far is the stand, where units are homogeneous regarding ecology, physiography and future developments. Some remarkable works are those by Aerstenet et al. 2010; Anderson et al. 2005; Crookston and Dixon, 2005; Ducheyne et al. 2004; Huth et al. 2005; Kolström and Lumatjärvi, 1999; Mathews et al. 1999; Mette et al. 2009; Torres-Rojo and Sanchez-Orois, 2005; Seely et al. 2004; Snow and Lovatt, 2008; Twery et al., 2000 and Varma et al. 2000. They have been applied in different areas such as Scandinavia, Australia, Austria, Canada, Malaysia, Scotland, Germany and Turkey for mixed stands of different species such as firs, spruces and tropical species among others. Works on this scale with differenciating characteristics are those by Baskent et al. (2001) about simulated stands; by Vacik and Lexer (2001) and Kurttila (2001) applied to stands from natural regeneration; and those by Chertov et al. (2002) and Goldstein et al. (2003) which analyze the consequences on natural ecosystems. The studies by Nute et al., (2005), Twery et al. (2005) and Salminen et al. (2005) enable the user to update the investment assessment on a stand level and on a whole exploitation level, developing thus scenarios of one or more treatments for management units. Martins and Borges (2007) point out that the search for sustainability of woods belonging to a high number of non-industrial private forest owners (NIPF owners) requires devising tools of the appropriate size for the properties and decision scale.

Flexibility in decision-making has become an essential element in the development of forest DSS in recent years. There has been a development from methods that allowed only unilateral decisions, only one person has the decision-making power (Thomson et al. 2000; Leskinen et al. 2003; Kaloudis et al. 2005). Even if unilateral systems have been maintained , new ones have been developed where decision is collegial, that is, multiple participants express their preferences to support an only actor in the decision-making. In Kangas and Lekinen (2005), some experts choose the explicative variables that will be used in the model after a careful study of the forest area. The software for damage reduction by fire proposed by Kaloudis et al. (2010) has been initially tested and evaluated by three different groups of users.

The most recent, interesting and complex issue in forest DSS are those with participative decision-making by several stakeholders who must reach an agreement for a final decision. In Nute et al. (2000) decision-making is developed with a social participative and environmentally sensitive methodology in Central America. Mendoza and Prabhu (2003) use MCA methodology to carry out an assessment of the Criteria and Indicators (CandI) structure in an environment of participative decision-making. In a context of public participation, Sheppard and Meitner (2005) describe the managers needs in sustainable forest planning, outlining the criteria for the design of support processes for these decisions. According to Mendoza and Martins (2006) the qualitative method allows a more participative decision-making process. In public participation processes, Kangas et al. (2006) state the importance of questions such as equity, representativity and transparency. Martins and Borges (2007) interprets the design of a forest management plan as a case of participative planning. Ramakrishnan (2007) uses participative management methods in sustainable forestry. Other illustrative examples are those presented by Vainikainen et al. and Wolfslehner and Vacik in 2008 and Anderson et al. in 2009.

It is important to highlight the development of mathematical tools and their implementation by means of information technology for efficient problem solving. Díaz-Balteiro and Romero's work entitled "Making forestry decisions with multiple criteria: A review and an assessment" (2008), makes an excellent assessment of the different issues in forest management and the different problem-solving tools. Finally, there are some references that haven't been used in the afore mentioned work and that have incorporated some different types of relevant techniques for forest DSS: Aitkenhead and Aalders (2009) use Bayesian networks; Martín-Fernández and García-Abril (2005) and Zeng et al. (2007) use genetic algorithms and tabu searches; Stirn (2006) uses dynamic programming and fuzzy techniques; Chertov et al. (2002) use data mining; Wolfslehner et al. (2005) use AHP and ANP; and MacMillan and Marshall (2004) use lineal programming.

2. Forestry sector in Galicia¶

The forestry sector is crucial in Galicia from a strategic, social and economic point of view, Figure 1 (Marey-Pérez and Rodríguez-Vicente, 2008). In recent years, Galicia has produced half the wood in Spain, becoming in some periods the ninth country in the wood harvest rank in the EU, even above the United Kingdom (FEARMAGA, 2009). In the past ten years, forest producers in Galicia have perceived 1,000 million euros due to wood selling. Due to this production capacity, Galicia has a wood transformation sector with around 3,500 companies, mostly family business, which employs 26,000 people directly and 50,000 indirectly. In fact, it is overall the third most important industrial activity and in twenty out of the fifty-six forest regions it is either the first or the second. These forest regions are located mainly in rural environments, so it is one of the main assets for the sustainability of rural population (Marey-Pérez and Díaz-Varela, 2010).

One of the most serious problems for this sector is its atomization (Marey-Pérez et al. 2006). This stems from the small size of the property of each owner and of each parcel. This results into 700,000 forest owners with less than 3 ha of average property divided into more than 8 parcels (Rodríguez-Vicente and Marey-Pérez, 2010).

3. A supporting decission forest system: SaDDriade

3.1 Origins

New information and communication technologies are currently, and will be to a higher extent in the future, the basic pillar on which the economic development of our society will lie. In rural areas in Europe the currently-existing digital divide will be overcome by different means: public funding, training and the initiatives of organizations and companies

Fig. 1. Location of Galicia in Europe and Spain

regarding these types of activities. These initiatives are going to depend on the DSS problem-solving capacity.

In the Research group "Proyectos y Planificación de la Universidad de Santiago de Compostela (GI-1716)" we have been working on solutions to the problems experienced by agricultural and agroindustrial sectors in Galicia and in other rural areas in Europe and Latin America. SaDDriade is the result of a process that started with the analysis of the weaknesses and strengths of the forestry sector. Strategic plans regarding forest industry were revised, wood, furniture, energy, environmental preservation and land planning, among others. Special emphasis was placed on the revision of information technologies and sustainable development with the idea of gathering as much information as possible. The experience of the research group was incorporated to the corpus of knowledge, providing it with a scientific and practical dimension.

Symposia, conferences, forums and meetings with owners associations, industry, administrations and scholars have been organized in recent years. This provided very valuable reflections for the definition of a DSS adapted to the reality of our region. Finally, financial, technical and scientific support to start the project has been obtained as a result of the combination of the Xunta de Galicia research Project "*Sistema de apoio a decisión para montes veciñais en man común (SadMvmc)*" *(07MRU035291PR*) and the different collaborations with public administrations, forest associations and private companies within the framework of the project *COST Action FP0804 - Forest Management Decision Support Systems (FORSYS).*

3.2 Motivations

The study of the data gathered provided the keys about the demands that SADDriade should answer and also of the way in which this should be done to provide the right answers to potential users. Below, there is a selection of weak points of different aspects within the forestry sector from different reports chosen due to their different degrees of usefulness for the development of the system.

- Make information available to facilitate strategic decision-making by the companies in the forest chain and guide the definition of public policies.
- Promote the activities of support services (researcher training or services to companies) that encourage industries' main activities.
- Decrease the lack of appropriateness of the formative level of the general population to the requirements of the companies. Despite the recent efforts and advances, there is still an imbalance between the needs of the companies and the development of technologies.
- Collaborate in a rational exploitation and use of natural spaces and in the reduction of industrial impact on the ecosystem.
- Encourage the collaboration of universities, technological centers and companies.
- Promote forest as a source of income for companies and private owners. Forest smallholder property poses difficulties for sustainable developments since forest areas are considered a secondary source of income and specific forestry is not developed.
- Provide businessmen with competitive advantages in tangible resources such as financial, technological or natural resources to counteract the competence in Price of markets with cheaper labour.
- Promote the implementation of technology in companies where the presence of technology is not enough, which limits their competitivity and the development of forest activities.

In our proposals, our aim was to make a forest DSS of immediate use for companies. It would have no cost for them in implementation and in licenses, and it would not require specific training for its users or an equipment update. It would also reach the highest possible number of users in the shortest possible time, which made it necessary to have a fluent transmission of knowledge. Our experience as university instructors enables us to identify the most efficient means in which users acquire knowledge. After considering different possibilities, we reached the conclusion that the only option that fulfilled all the requirements was the world wide web, using a web application.

Within the creation process of a software tool, the choice of a certain development platform is a key issue that conditions the rest of the actions. It is necessary to make a detailed analysis of the weaknesses and strengths of each programming environment and compare them with potential user profiles and the requirements for a satisfactory user experience. Currently, the technological developments and the increase in telecommunications favours the development of web applications and their merge with mobile ones, instead of with desktop applications, which are becoming less important. Some of the reasons to choose a web platform are:

- Multi-platform compatibility: Several technologies such as PHP, Java, Flash, ASP and Ajax allow an effective development of programs supporting the main operating systems.
- Update: Being always updated without pro-active user actions and without calling the user's attention or interfere in his/her working habits.

- Immediacy of access: Web-based applications don't need to be downloaded, installed or configured. They can be accessed via an internet address and be ready to work, regardless of their configuration or hardware.
- Easy trial: There are no obstacles for easy and effective tool and application trials.
- Lower memory requirement: They have more reasonable RAM memory requirements for the end-user than programs installed locally. Since they are stored and run in the provider's servers these web applications use these servers' memory in many cases.
- Fewer errors: They are less likely to 'freeze' and cause technical problems due to hardware conflicts with other existing applications, protocols or internal personal software. In web applications, they all use the same version and all the errors can be corrected as soon as they are found.
- Price: They don't require the distribution infrastructure, technical support and marketing required by traditional desktop software.
- Multiple concurrent users: These applications can be used by multiple users at the same time. There is no need to share screens when multiple users can jointly see and even edit the same document.
- Usability: Web browsers are by far the computer application with the highest presence in computers around the world due to the expansion of internet as a channel of communication for businesses and particulars.

3.3 Who made SaDDriade?

The SaDDriade team is made up of ten people with the collaboration of different professionals and technicians from the forestry sector, the administration and more than twenty well-known international experts in the field.

Gradually, technical-economical models have been built for the different forest species and their different locations in Galicia. These models include the most advanced techniques in forestry and individual tree or forest growth, together with the parametrized financial component of the different forest management phases or tasks: land preparation for planting, tasks linked to forestry and the use of wood or biomass.

3.4 Users of SaDDriade

During the development of all its components, we considered the potential users, their demands and their previous knowledge. In this way, we have developed a user-friendly, accessible, usable application, with a clear presentation of results. It also enables the user to determine in which phase of the work he/she is without being trapped within the program.

The characteristics of the potential clients and the demands that SaDDriade answers are outlined below:

- Forestry business: Companies related to reforestation, forestry, wood and forest biomass harvest, first transformation industry (sawmills, wood plank industry, paper and pulp industry and biomass power generation) and consultancies and engineering technical offices that advise owners.
- Forest owner associations: their needs and demands are very similar to those of the previous group. However, most of them lack the qualified technical staff to rigorously go through the planning process. SaDDriade has been conceived with the idea of helping them improve the quality and quantity of their forest product.
- Research groups in universities and research institutes: SaDDriade can perform the role of a lab and database with useful information to understand how forest reality works.

All this knowledge and experience will result into new studies that will contribute to quality forestry based on multifunctionality, energetic use and rational resource planning. Its ultimate aim is the excellence and sustainability of forest farms that will secure the financial future of small forest farms in short, middle and long term.

- Forest training centers. Forest training centers and specially those departments at universities devoted to forest activity can use this application for teaching. Students can acquire experience in forest planning and management and use forest simulators in their area seeing the possibilities of evolution and acquiring practical knowledge that would be impossible otherwise.

3.5 Objectives of SaDDriade

The main objective is to provide information about the productive cycles of the different forest species in Galicia. The data provided are going to enable the knowledge and assessment of the different steps to be taken in the productive processes, the costs associated to each of them and the expected final yield.

Users receive answers to queries regarding different aspects of forest management such as:

- Forecasts: They are obtained from data of potential stock for the different models in the different parcels where the simulation is carried out. They provide information about: the works to be done, the costs associated to them, the forecast profits for different years and for the end of turn or cycle of the technical and financial model developed.
- Situation reports: They are "snapshots" of the state and value of wood or biomass at a certain moment. They help determine the investment to be made over a certain period of time, the value of existing products in the parcels and the years and operations necessary to get profits from the proposed models.
- Investment and profitability analyses: They provide knowledge beforehand about the effort necessary to be made in forest activity in a certain parcel. Indicators such as IRR and NPV will be useful to calculate the expected profitability in each scenario or technical-economical model developed and the aspects that have more influence in such profitability.
- Improvement of training procedures (training/extension): The use of this application requires user training. Using this program improves the knowledge of the implications of forestry and forest management.
- Stock control: Users are able to consult the volume of wood that their parcels have at any given moment. This enables them to forecast their production or report losses in case of fire, wind or snow damage.
- Cost reduction: The availability of information entails a possible cost reduction. Two factors are key in this fact: first, the knowledge of average costs of the operations enables the management of the provider and contractor offers according to contrasted terms of reference; second, the knowledge of the optimum production models for each technical-economical models avoids unnecessary operations.
- Market transparency: The availability of accessible information has a direct influence in flux transparency and commercial relations that contribute to the clarification of a sector, which is traditionally considered to lack transparency. Knowing the real average cost of obtaining a product and its market value simplifies the buying and selling process and the accessibility to raw materials.
- Technical documentation and management system: Being an application based on a central server with all the security protocols, it is going to provide the users with a

space where they can make and store their operations with the guarantee of recovering and using them in the future.

- Help for administrative processes: A high number of activities related with forest activity productive processes are either publicly-funded or regulated by the administration. This administrative process comes with the need for technical documentation, which can be obtained with the format suitable for administrative use. In the future, electronic administration will be wide-spread so this application will be essential.
- Database with legislative information. The application has an associated database with legislative information (Lexplan module), with all the legislation regarding forest activity in its different phases. The user has access to all regulations and public funding that he or she can be eligible for according to the activity carried out in each particular moment

3.6 How does SaDDriade work?

The different sections below explain how SaDDriade works. We will start by the explanation of the programming language(s) used, the operative environment, the architecture goals and how it actually works.

3.6.1 Programming language

SaDDriade has been designed using more than one programming language. This was due to the complexity of the calculation processes, the web environment, the diversity of data sources, the need to have a GIS WEB tool and the different formats of exporting results. The languages used are: PHP (PHP Hypertext Preprocessor), Javascript, Mapscript and SQL (structured Query Language) All the components used in this application are open source components. They have been selected not only because of our agreement with the social philosophy and support of knowledge of the open source movement, but also for the financial advantages for both users and developers.

3.6.2 Operating environment

The main component of the Geographic Information System (In Spanish Sistema de Información Geográfica, SIG) integrated in the application is the open source platform Mapserver. This application was created in order to publish spatial information and to create interactive applications for maps. SaDDriade uses a combination of servers of relational databases of different kinds: on the one hand MySQL, and on the other hand PostgreSQL, with support for spatial data by means of the extension POSTGIS. Such services, together with a variety of files in shape format are the ones that feed the map server. The graphic interface was made in XHTML (eXtensible Hypertext Markup Language), a markup language whose specifications are developed by the World Wide Web Consortium (W3C).

3.6.3 Architecture

SaDDriade is a web application, so it can be used by accessing a web server by means of a client, typically a web browser. Through this client-server scheme, there is no need for a specific installation client side. It also makes it easier to install and maintain the application without having to distribute specific software to the clients.

The model-view-controller (MVC) design pattern was used to develop an internationalizable modular application which is easily extensible and has the capacity to access different database management systems. Three different frameworks were used, each of them with specific functions to optimize the general work of all the decision support system. **Codeigniter** framework is the base of the structure. Some of its advantages are: its speed, its excellent documentation, PHP compatibility backwards, small software and hardware requirements, native support for user-friendly URLs, drivers for a wide range of server database (MySQL, PostgreSQL, SQL server, SQLite, among others) extensible by helpers, plugins and libraries.

Codeigniter optimizes its performance eliminating non-essential distribution elements. Thus, its functionalities are more limited than those of other frameworks. In order to solve this problem, SaDDriade includes **Zend Framework**. Its main features are: low coupling among its components, wide unit test code coverage, multipurpose (capacity to generate PDFs, Access to LDAP, SOAP, Lucene, DOM, JSON, ACL system, email, authentication, automatic pagination, among others), use of design patterns, access to database by PDO and possibility to have an ORM (Object Relational Mapping), interoperatibility with the most important web services (Akismet, Amazon, AudioScrobbler, Delicious, Flickr, Nirvanix, Recaptcha, OpenID, Technorati, Twitter, Yahoo, Google, Youtube and Picassa) and high frequency of updates and new versions.

The third framework used is **Jquery,** a javascript library oriented to objects in the form of a resource collection that facilitates the reuse of the code and ensures the compatibility between different versions and types of browser. Special care was taken so as not to make an intrusive use, following the recommendations of the World Wide Web Consortium regarding accessibility. Within the general scheme of the application Jquery is responsible for the improvement of the user interface such as sortable tables and asynchronous communication with the server to increase the speed by the development of the XmlHttpRequest API (Application Programming Interface), a wide-spread technique known as AJAX (Asynchronous Javascript And XML).

SaDDriade GIS WEB structure revolves around Mapserver, which can process geographical information from different sources (WMS servers, WFS, shapes, databases, raster,…) and create very complex representations (depending on their configuration) which are shown in a client map. This configuration is normally done via GET parameters through a URL to a CGI script. It is an easy way to send requests to be analyzed by the server. A downside is that the possibilities of a dynamic answer using this method are limited and don't allow a complex application creation. To avoid this problem Mapserver has its own language, mapscript, which can access its API directly. In this way, programmers have all its library of functions to create interactive applications or RIA (Rich Internet Applications).

Mapserver itself can show the maps it generates or use a client to do so. SaDDriade uses a client. Maps are visualized using pmapper, a set of libraries made in PHP, jquery and mapscript, that offer more possibilities than those included in mapserver as a light client map. Pmapper has General Public License, so it can be used for free. It was chosen for its capacities, for being user-friendly and its integration with other program components. For instance, they share the jquery framework and the TCPDF library to create PDFs.

From the data obtained in the parcel shape and the zoning ecological criteria, pmapper displays a GIS client with capacity to add orthophotos of the SIGPAC project, parcel and municipality search, measurements, identification, zoom, annotation and map downloads as image or PDF. The user can choose a parcel, by clicking the "Identificar" button on the right side menu, and start the simulation process using the link offered by pmapper.

3.6.4 Work description

In SaDDriade, there are forest management models implemented for twelve different species (see table 1). In this way, 146 models have been parametrized in the forty areas in which Galicia has been divided. This has encompassed 13,108 tasks and subtasks, and the use of 160 different types of materials, machinery and so on.

Species	Number of models
Pinus pinaster	17
Pinus sylvestris	9
Pinus radiata	15
Pseudotsuga menziesii	8
Quercus robur	8
Quercus rubra	12
Quercus pyrenaica	6
Juglans regia	6
Populus sp. (hybrid)	8
Eucalyptus nitens	7
Castanea sativa	23
Castanea hybrid	27
TOTAL	**146**

Table 1. Species and number of models developed for each species.

Designed according to technical criteria:

- Possible ways of mechanization.
- Selection of the best available technique.
- Limit of appropriate densities.
- Programming activities on land and trees.
- Programming of intermediate harvests.

Financial criteria:

- Establishing technical shifts.
- Maximum rent shifts.
- Expense minimization.
- Benefit estimate.

Model choice

The first thing that the user must do is to choose which available SaDDriade module he/she would like to use: SAD Castanea, SAD Eucalyptus, SAD Pinus, SAD Populus and SAD Quercus.

Starting from the GIS-WEB, as stated above, the actual process starts once the user selects his/her parcel. Once the link shown is clicked, a window appears with basic data regarding the choice. Questions guide the user throughout the decision support process. By the location of a chosen parcel, a first filter of qualities and species has been set, so only those technical-economical models considered ecologically and financially viable are accessed. Models are classified by species and production destination to simplify the choice. The user must select a model among all the options to continue the calculations.

Fig. 2. SaDDriade initial screen.

Management variables

After the basic characteristics of the model are presented, some parameters that influence the task development and performance of the chosen model are shown: Characteristics of the place: slope, percentage of rocks, stone content, soil type; Initial state of vegetation: type, height, density and presence of stumps; Management alternatives: choice between base-root

Fig. 3. Characteristics of a selected model.

and pot planting, work line and plant and plantation protection; other management alternatives: characterization of protection and grubbing.
Options arise parallel to the simulation process and the user always has the option that the SaDDriade "remembers" his/her answers throughout the process just by clicking on the checkbox beside each answer. Twelve thousand possible combinations of profits, 13,000 yearly subtasks (that generate more than 4,300 partial products), 38 types of different machinery, 72 types of inputs and 47 different types of final outputs. They are all associated to prize, restriction, modelization and classification databases.

Results

Once the simulation is finished, a screen appears with the results in different formats.

- **Interactive support graphs.** They are interactive graphs that show the financial evolution of planting in time, broken down into expenses and income and cost distribution of labor and machinery.
- **Economic indicators. Total expenses:** result of the sum of all labor, machinery and input expenses. Total income: sum of the income obtained by product sale. Profits: Difference between income and expenses. NPV: Net Present Value. IRR: Internal Rate of Return calculated by the Newton- Raphson method.

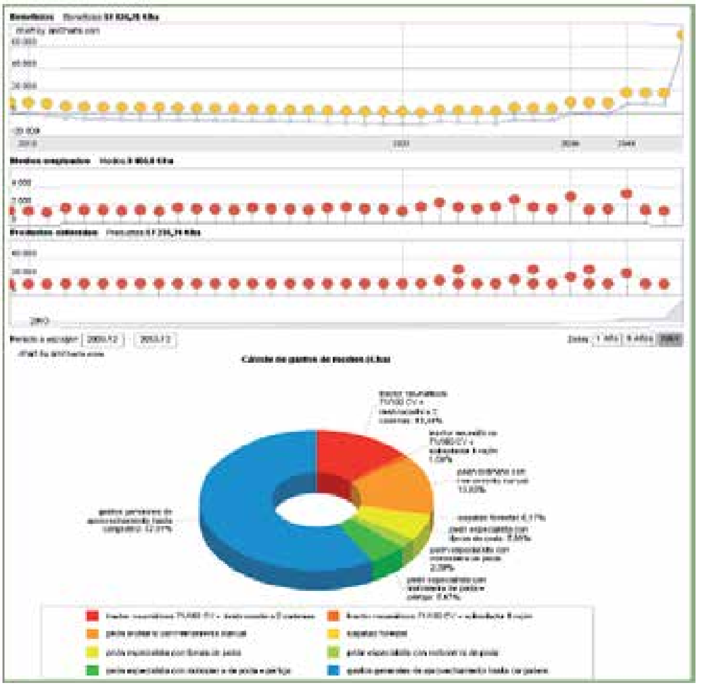

Fig. 4. Different economic indicators for a simulation.

- **Planting visualization with Google Earth**: The evolution of the planting of the parcel is shown by means of a Google Earth plugin (at this moment only available for Windows and Mac OS X 10.4 +). This enables a landscape analysis of the different decisions and operations carried out. It is a 3D representation of a planting on the square of the whole parcel or just a section, where the user can fly virtually into the trees, "walk" between them or make visualizations within a wider context.

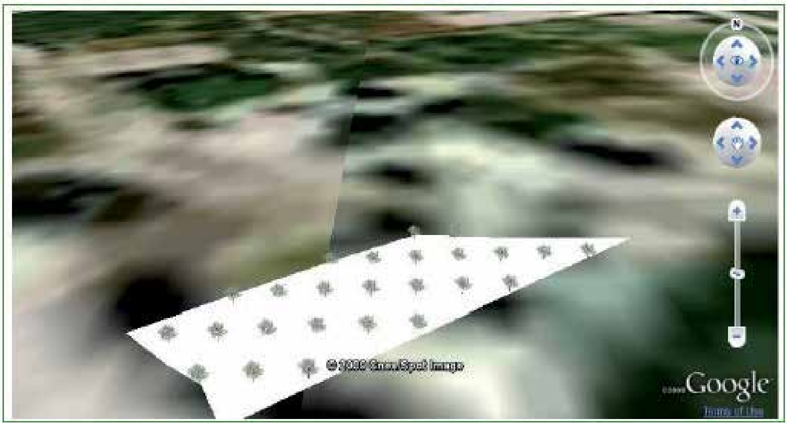

Fig. 5. Location of trees in a parcel and selected model.

Exportation

Exportation can be made in three formats, depending on the interest of the user: **kmz (**Compressed file containing geographical data and tridimensional models used to present the evolution of time in the planting. It can be opened with Google Earth), **pdf y xls.**

4. Conclusions¶

Decision Support Systems have proven to be useful in the different economic fields in which they have been developed because of their capacity of simulation and optimization. Forest DSSs have evolved over time thanks to IT, on the one hand, and due to the need to introduce higher social and environmental restrictions to forest management, on the other hand.

The development presented in this chapter includes the most advanced techniques in IT (web application and virtual reality simulation). It is also easy to use, which would allow a higher number of users to have access to it without much IT or forestry knowledge. Thus, it could, and it should, become a tool for forestry extension, which would make this type of management more sustainable and will give the possibility of increasing its level of technification.

New lines of development that will result into a better tool to the service of owners and forest managers are: the inclusion of optimization options and stochastic processes, the resolution by heuristic methods in which the risk of forest activities in different geographic environments will be taken into consideration and an improvement in the visualization options of virtual reality.

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22

Application of Multi-Criteria Methods in Natural Resource Management – A Focus on Forestry

Mario Šporčić
University of Zagreb, Faculty of Forestry
Croatia

1. Introduction

Natural resource management refers to the management of natural resources such as land, water, soil, plants and animals which in accordance with the concept of sustainable development, a distinct emphasis puts on the way the management affects both present and future generations. In management and utilization of forests and forest land, as one of the most significant natural resources, the principle of the sustainable development is incorporated in a way that adheres to biological diversity, productivity, regeneration capacity, vitality and potential of the forests to fulfil, now and in the future, its important economical, ecological and social functions.

Forest resources and benefits that derive from them represent an important part in fulfilling the needs of humanity for energy, raw materials and quality of life. These benefits cover a broad range of goods and services. Among other, they include: wood, recreation, water, soil preservation, clean air, game, scenic beauty, etc. Many of such benefits and services can be simultaneously gained from a single forest stand. And even though many countries have legislative regulations that prescribe the course of forest management and/or protection of certain forest functions, there are still many debates on the issue how to manage forests and to which purposes. In general, we could say that today the basic postulate of forest management is multifunctional or multiple use of forests. It represents the manner in which the most of many different functions of forests are being utilized. In that sense, forest management should enable the most prudent usage of forests and forest land to provide some or all of respective products and services, while ensuring productivity and stability of forest ecosystems at the same time. In realizing these goals careful planning and decision making play a major role, and are considered to be especially significant for effective natural resource management and achieving the principles of sustainable development.

Planning and decision making in forest management represent a very complex task mainly because of the multitude and a broad spectre of criteria enrolled in the decision making process. That means that any decision making is under many different influences, and that at the same time every decision made affects many criteria of different nature. These influences and criteria encompass (Diaz-Balteiro & Romero 2008):

a. economical issues – wood production, non-wood forest products (forest trees fruits and flowers, seeds, mushrooms, honey, resin, humus) livestock, game management, hunting;
b. ecological and environmental issues – soil erosion, watershed regulation, biodiversity conservation, carbon sink, scenic beauty, influence on climate;

c. social issues – recreational activities, tourism, level of employment, rural development, population settlement etc.

Moreover, the complexity of a large proportion of forestry issues is increasing due to the way in which different interest and social groups and organizations perceive the relative importance of specific criteria and appraise the management of forests, and thus judge the "quality" of forest resources management. The importance of specific criteria and evaluation of forest management in that sense depend on the personal standpoints and opinions of each individual i.e. group. Examples of such subjective assessments are often related to scenic beauty or recreational value of a certain forest area, or for example to game management and hunting. So, while someone preffers a specific game species and specific type of hunting, someone may want different kind of game and hunting, and someone else may be absolutely against hunting at all. Similar evaluations of forest management are related to the logging and creating certain revenue on the one hand and the protection and conservation of forests on the other hand.

All of the above mentioned daily increases the complexity of forest management, hinders the performance of forest operations and hardens the management conditions making the planning and decision making in forestry very demanding. And while in the past decision making and management in forestry have frequently been performed on the basis of common sense and/or past experiences, today's forestry with multiple criteria and functions calls for more flexible decision support. The complexity of today's business environment in forestry, the imperative of continuous increase of business and ecological efficiency, and multiple stakeholders with different interests impose the necessity to use new models and more precise methods. In that kind of a situation the joint use of multi-criteria decision making methods and different techniques of group decision making are becoming an important and potentially desirable way for solving forestry issues. It is considered that multi-criteria decision models and methods can provide to modern forestry, which has multiple aims and tasks, and multitude of interest groups with often conflicting interests, a strong and flexible support to decision making. Development and application of such methods that haven't traditionally been used in forestry could provide to management a new tool which can be a valuable aid both on strategic and operational level of decision making. The emphasis in doing so, is on the fact that decision proposals and decisions made must be based on the rational arguments.

This paper provides an overview of certain multi-criteria methods which can be used as a support for planning and decision making in forestry. Several methods of multiple-criteria decision making have been described and compared. Brief description and comparison presented in the paper includes following multi-criteria methods: Data Envelopment Analysis (DEA), Analytic Hierarchy Process (AHP), Multi-Attribute Utility Theory (MAUT), outranking methods, voting methods and Stochastic Multicriteria Acceptability Analysis (SMAA). The paper also gives a brief overview and analysis of problems and forest areas where multicriteria methods have been applied so far. The intention was to explain for which types of tasks and problems these methods can be applied in the field of forestry. That provides an insight into characteristics of the respective methods and a guideline to eventual choice of which method to apply. Many of the articles cited in the paper provide information on the existing experiences, reflect the actual role and significance of multi-criteria decision making in forestry and represent a valuable reference source that can be beneficial to students, researchers, experts and practitioners in forestry. The main aim of the paper is to raise the forestry profession's awareness about the importance and potential role that multi-criteria decision making can play in forestry. Concrete examples of the carried out investigations provide an insight into the possibilities, suitability and justification of the application of multi-criteria methods.

2. About decision making

Decision making process involves the choice of a specific solution among the set of different alternatives that solve a given problem. In a decision problem, there are goals to be achieved by the decision, the criteria used to measure the achievement of these objectives, the weights of those criteria that reflect their importance, and alternative solutions to a problem. Under the objective we consider the state of the system we want to reach by a decision, the criteria are the attributes that describe the alternatives and their purpose is to directly or indirectly provide information about the extent to which each alternative achieves the desired goal. In a given decision situation, the criteria are usually not equally important, and their relative importance is derived from the preferences of decision maker what is related to his system of values and other psychological characteristics. Data and information about these elements are with the appropriate actions summarized in one number for each alternative, and on the basis of these values the ranking of alternatives is determined. Figure 1 shows the basic procedures and steps in the process of decision making and problem solving.

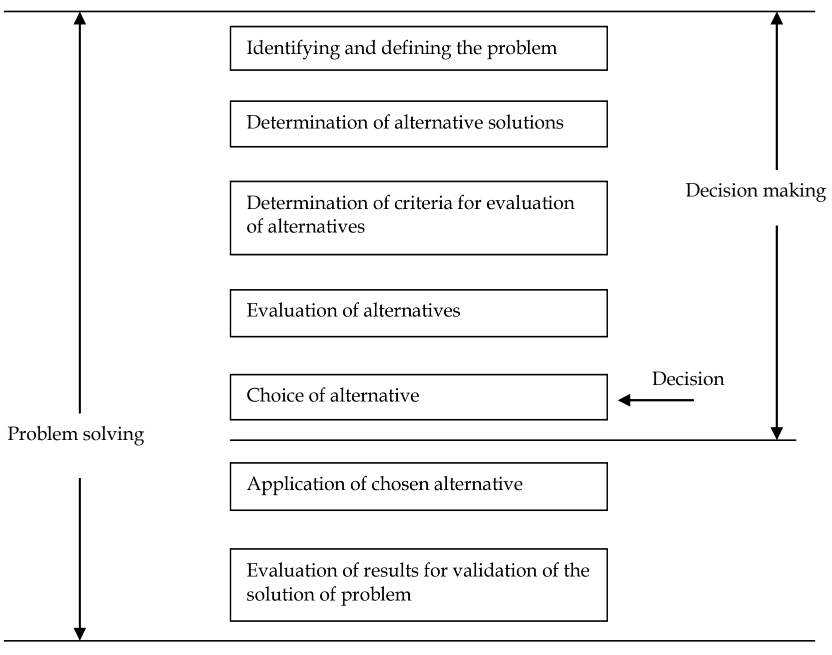

Fig. 1. Relationship between problem solving and decision making

Decision making is one of the major human activities, and one of the unavoidable tasks of managers. The decision situation is solved by adoption of a decision, which represents a selection of one action out of solutions available. The significance of decision making reflects in the fact that even if none of the possible solutions and actions have been chosen, the decision has been made - it has been decided not to choose or to do nothing.

3. Multi-criteria decision making approach

Multi-criteria decision making[1] falls within the wide range of operations research methods. As the name suggests, MCDM has been developed to enable analysis of multiple criteria situations and problems. It is usually applied in such cases where it is necessary to holistically consider and evaluate various decision alternatives, in which comprehensive analysis is particularly difficult due to a multiplicity of hardly comparable criteria and conflicting interests that influence the decision making process.

A number of MCDM methods have been developed, each of them with specific characteristics and different techniques that are applicable in appropriate circumstances and situations. For example, some methods are specially designed to manage risk and uncertainty, or for non-linear estimation, while others are focused on applications in conflict management tasks and objectives or on the use of incomplete or poor quality information. Many methods also come with a variety of settings and in different versions (eg, 'fuzzy' or stochastic versions, etc). Some are also slightly modified to better respond to tasks and problems in certain areas, including forestry. A detailed overview of operational research and multi-criteria decision making methods can be found in numerous sources (Vincke, 1992; Triantaphyllou, 2000; Koksalan & Zionts, 2001; Kahraman, 2008 etc).

The procedure of multi-criteria decision making involves the development of several alternatives that can no longer be improved by some criteria, while at the same time not ruined by the other criteria (Pareto optimality or efficiency). A comparison of selected alternatives is implemented considering all the previously set criteria and characteristics that influence the selection of a particular solution. As a result of a comprehensive comparison, the priority and rank of the observed alternatives is determined. In a group decision making individuals may, depending on their personal preferences, differently rank some alternatives. Comprehensive comparisons can also be made with assigning different weights to certain criteria, but also to opinions of individual participants. This includes the influence of different criteria and individual points of view which are taken into consideration together. In this way, MCDM methods can be used to analyze the situation of decision making and help in making the best possible or at least satisfactory decision.

Bearing in mind the above, it is considered that with the application of MCDM methods, many challenges in today's demanding and complex forest management planning can be facilitated and minimized. Many authors have written on that topic (Tarp & Helles, 1995; Krč, 1999, Kangas & Kangas, 2005; Herath & Prato, 2006 etc).

4. Main MCDM methods

This section gives a brief description of MCDM methods that can be applied in multifunctional forest management. Selected approaches represent different theories and schools as part of operational research. All presented methods have been tested and applied in forestry, and although many methods are not included in this paper, most of them are based on similar assumptions and theory as methods presented. For a more detailed study of specific methods and their application in forestry, relevant sources are given.

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[1] Multiple Criteria Decision Making (MCDM) ili MCD Support (MCDS) ili MCD Aid (MCDA)

4.1 Data Envelopment Analysis (DEA)

In recent years DEA has become one of the central techniques in the analysis of productivity and efficiency. It was used in comparing organizations (Sheldon, 2003), companies (Galanopoulos et al., 2006), regions and countries (Vennesland, 2005). In determining business efficiency it was applied in banking (Davosir, 2006), education (Glass et al., 1999), agriculture (Bahovec & Neralić, 2001), wood industry (Balteiro-Diaz et al., 2006), forestry (Lebel, 1996; Kao, 1998; Bogetoft et al. 2003; Šporčić at al., 2008, 2009). DEA bibliography records more than 3200 papers published to 2001 (Tavares, 2002).

DEA is a methodology for determining the relative efficiency of production or non-production units (Decision Making Units, DMU) that have the same inputs and outputs, and vary according to the level of resources available and the activity levels within the transformation process. Based on the information about the actual inputs and outputs of all observed DMUs DEA constructs an empirical efficiency frontier and calculates the relative efficiency of each unit. The most successful units are those that determine the efficiency frontier, and the degree of inefficiency of other units is measured based on the distance of their input-output ratio in relation to the efficiency frontier.

While typical statistical methods are characterized as the central tendency approaches, which make their estimations based on the average production unit, DEA is based on extreme values and compares every DMU only with the best units. The basic assumption is that if some unit can produce Y outputs with X inputs, the other units should be able to do the same if they work efficiently. The center of the analysis lies in finding the 'best' virtual unit for every real unit. If the virtual unit is better than the original one, regardless if it achieves more outputs with the same inputs, or achieves the same outputs with less inputs, then the original unit is inefficient.

DEA relative efficiency scores are interesting to forestry experts, managers and researchers because of three DEA properties:

- direct comparison of units with multiple inputs and outputs with no need to know the explicit form of relation between inputs and outputs which can also be expressed in different units of measure,
- characterization of each organizational unit with one relative efficiency score,
- improvements which model suggests to inefficient units are based on actual results of organizational units that operate efficiently.

4.2 Analytic Hierarchy Process (AHP)

Analytic Hierarchy Process (AHP) is widely used and very popular method in many areas, including management of natural resources. Mendoza & Sprouse (1989), Murray & Gadow (1991), Kangas (1992) are among authors who have applied AHP in forestry, and the number of applications is steadily raising (Pykalainen et al., 1999; Ananda & Herath, 2003; Wolfslehner et al., 2005; Šegotić et al., 2003, 2007).

AHP has several advantages from the standpoint of multi-criteria and group planning. With the use of AHP, objective information, expert knowledge and subjective preferences can be considered jointly and simultaneously. It can also take into consideration qualitative criteria, while other methods usually require quantitative values for the selection of the alternatives. Solving a complex decision problems using this method is based on their decomposition into components: goal, criteria (sub-criteria) and alternatives. These elements are then taken into a multi-level model (hierarchical structure) where the goal is on the top, and the main

criteria represent the first lower level. The criteria can be broken down into sub-criteria, and on the lowest level of hierarhical structure, there are alternatives. Another important component of the method is a mathematical model which calculates the priorities (weights) of the elements on the same level of hierarchical structure. The method is based on comparisons of pairs of alternatives, each one with the other, while expressing the intensity and weight preferences of one alternative over another. The criteria are compared in the same way, whereby preferences are expressed by using Saaty's scale (Saaty 1977, 1980).
Negative aspect of the method is that it does not allow any reluctance and hesitation in the comparisons. In the management of natural resources, much of the information and data underpinning the planning and decision making is characterized by a certain level of insecurity and uncertainty. Furthermore, the number of comparisons significantly increases with the number of alternatives and criteria, which can be expensive and demanding. To overcome these limitations different AHP models have been developed. A'WOT combines AHP and well-known SWOT analysis (Kurttila et al., 2000), Analytic Network Process (ANP) is an extention of AHP (Satya, 2001) etc. Such hybrid models also have the same basic idea of pair-wise comparisons as practical, pedagogical and intuitive approach. Popularity of the method is primarily based on the fact that it is very close to the way in which individual intuitively solves complex problems by dismantling them into simpler ones.

4.3 Multi-Attribute Utility Theory (MAUT)

MAUT is a structured decision-making procedure for making a selection among different alternatives in relation to fulfilling a selected criteria. It is based on the utility theory that systematically seeks to validate and quantify the user's choice, usually on a scale 0-1 (Keeney & Raiff, 1976). Based on MAUT methodology there have been developed methods such as HERO and SMART, which rank given alternatives directly by assigning them numerical values proportional to their importance (Venter et al. 1998; Kajanus et al., 2004).
Simple Multi-Attribute Rating Technique (SMART) was developed in the early 1970s within the multiattribute utility theory. SMART methodology has many similarities with the basic idea of AHP method, but the main difference is that SMART does not use the comparison in pairs. Instead, the ranking of alternatives is carried out directly. Direct ranking means that the criteria are directly assigned numerical values proportional to their importance. Accordingly, alternatives are assessed with respect to each decision criterion by simply giving them relative numerical values that reflect their priorities. Most often, after the selection of criteria, the main criterion is determined and given a value 100. All other criteria are assigned values between 0 and 100, depending on their importance to the main criterion. According to the same principle each alternative is assigned a certain value in relation to individual criteria. The best alternative is given the value 100, while all other alternatives have values between 0 and 100 depicting their rank. When the importance of certain criteria and priorities among alternatives have been identified, SMART uses the same computational procedures as AHP. Examples of using SMART in natural resource management include Venter et al. (1998), Kajanus et al. (2004), etc.

4.4 Outranking methods

Outranking methods represent European or French School of MCDM. Many different methods have been developed, and among them PROMETHEE and ELECTRE have been applied in forestry (Kangas et al., 2001). These methods compare the alternatives in pairs, on the basis of so-called pseudo-criteria. Pseudo-criteria are two threshold values, the indifference threshold and preference threshold, which describe the difference in the severity of preferences between two alternatives. If the difference is less than the indifference threshold, the alternatives are considered to be indifferent in regard to that criterion. If the difference exceeds the preference threshold, better alternative is considered to be better without a doubt. If the difference is larger than the indifference threshold, but less than the preference threshold, priority between alternatives is uncertain.

Calculations are carried out in different ways in PROMETHEE and ELECTRE, and both methods have several versions to suit different situations. The main advantage of these methods is that they do not require as complete preference data as AHP. Disadvantage is that these are fairly obscure methods that are quite difficult to understand and interpret.

4.5 Voting methods

Voting is a familiar way of expressing opinions and influencing important matters. Voting techniques can be applied in MCDM when determining the criteria. The criterion that gets the most votes is considered the most important. Another example might be a vote on the suitability of alternatives with respect to certain criteria. Voting can be conducted under the principle "one man, one vote" or by giving a voter a certain number of votes. In Approval Voting voter gives a vote to each option deemed acceptable. In so-called Borda Count each voter gives n votes for the best option, $n - 1$ votes for the next, and so on until one vote remains for the worst option. These methods are some examples of many voting techniques.

Voting techniques have been developed to handle situations with the low quality of data on preferences. Simplicity and comprehensiveness of the voting techniques are their main advantage, especially in group planning and decision making. By including more information, they increasingly resemble to SMART method. The general attitude is that voting methods should not be modified and further complicated for applications for which there already exist specific multi-criteria methods. The Multi-criteria Approval method is based on approval voting and has been applied in forestry (Laukkanen et al., 2002, Kangas & Kangas 2004). Shields et al. (1999) and Hiltunen et al. (2008) also applied voting methods.

4.6 Stochastic Multicriteria Acceptability Analysis (SMAA)

Similar to SMART, SMAA actually represents a set of methods. They were originally developed for discrete multi-criteria problems with uncertain or inaccurate criteria data, and where, for some reason, it was not possible to obtain data on weights and preferences from the decision makers. SMAA methods are based on determining the weight values that would make each alternative the preferred one, or that would give a certain rank to an alternative. Key indicators of SMAA include so-called acceptability indices, which describe the probability of placing an alternative in a certain rank. If the weight values of the criteria are not predetermined, the acceptability indices show the dominance of alternatives among all possible weighting combinations. The overall acceptability index can be calculated as a weighted average of the probable alternative ranks, with the most weight for the first place, then second and so on. This method is close to forest management where, due to a strong uncertainty in the planning, usually none of the alternatives under consideration can be safely declared as the best one.

The first applications of SMAA methods in forestry have been implemented in the context of ecosystem management planning (Kangas et al., 2003, Kangas & Kangas, 2004). Since SMAA includes many useful characteristics it is increasingly gaining interest in today's forestry and natural resources management (Kangas et al. 2006; Diaz-Balteiro & Romero, 2008).

4.7 Comparison of MCDM methods

Presented methods significantly differ one from another. Neither of reviewed methods is universal or the best, not even applicable in all cases. In fact, to a different situations and problems best suite different methods. Selection of appropriate method requires knowledge of various methods, their preferences, strengths and limitations as well as the characteristics and requirements of specific situation and problem in planning and decision making. Table 1 shows the comparison of presented and some additional MCDM methods.

MCDM method	Cost of implementation	Data reqiurement	Ease of sensitivity	Economic rigor	Management under-standing	Mathematical complexity	Parameter mixing-flexibility
DEA	M	M	L	M	L	H	M
AHP	M	M	L	L	M	L	H
Expert systems	H	H	L	H	M	H	H
Goal program	M	M	M	H	L	H	L
MAUT	H	H	M	M	M	M	H
Outranking	M	M	L	M	L	M	M
Simulation	H	H	H	H	H	H	M
Scoring models	L	L	L	L	H	L	H

H- high; M - medium; L - low

Table 1. MCDM methods' characteristics (Sarkis & Weinrach, 2001)

Table 1 shows that none of the methods does not dominate over the other methods. For example, when compared to other methods DEA is moderately demanding regarding costs of implementation and data collection. Sensitivity to changes in data is small, and the managerial understanding of the method is relatively low, mainly due to its mathematical complexity. The results are easy to interpret because it ranks compared units by their efficiency while flexibility allows including more parameters and factors in the analysis.

It is generally difficult to directly compare different methods. Each method has its advantages and disadvantages. The application often depends on the decision environment, where the availability of data, time and costs influence the selection of specific method. In any case, when applying in analysis researchers, experts and managers should be aware of their characteristics, both advantages and limitations.

5. Applications of MCDM in forestry

Although MCDM has been present in forestry for more than 30 years (Field, 1973), some newer approaches and techniques of multi-criteria and group decision making have become more significant in the early 1990s (e.g. Kangas, 1992). In that time period, a significant number of papers dealing with various problems of forestry have been published. This section will present some examples of conducted investigations and MCDM applications in certain forestry areas. Conditionally determined areas of forestry in which MCDM methods have been applied so far can be defined as follows (Diaz-Balteiro & Romero, 2008):

- Harvesting
- Extended harvesting
- Forest biodiversity
- Forest sustainability
- Forestation
- Regional planning
- Forestry industry
- Risk and uncertainty

Forest harvesting and it's planning is the first forestry area in which MCDM paradigm has been widely applied (Kao & Brodie, 1979; Hallefjord et al., 1986). Howard & Nelson (1993) used MAUT methods for solving a specific problem of forest harvesting. Diaz-Balteiro & Romero (1998) used AHP in planning of forest harvesting, while Heinonen & Pukkala (2004) in harvest scheduling issues used a version of HERO method.

Extended forest harvesting besides timber and logging, includes the problems of non-wood forest products. Thus, Arp & Lavigne (1982) in proposed multi-criteria model included timber, recreation, hunting and wildlife. Hyberg (1987) set a MAUT model with two attributes: production of wood and aesthetic values. Rauscher et al. (2000) with regard to more non-wood criteria, evaluated four management alternatives using AHP. Laukkanen et al. (2002, 2005) applied different voting techniques to several problems of forest exploitation in Finland. Kangas et al. (2005) used SMAA method with recreational and environmental criteria, and Pauwels et al. (2007) compared several silvicultural alternatives of Larix stands with the use of ELECTRE.

The field of forest biodiversity has been, from the position of MCDM, associated with the management of national parks, reserves, etc., where the selection of management activities leads to application of different MCDM methods. For example, Kangas (1994) applied AHP in the management of protected natural areas in Finland. Rothley (1999) used MCDM methods for designing optimal biodiversity network in Canada. Kurttila et al. (2006) used MAUT to find the optimal compensation for forest owners who orient their forest management towards biodiversity conservation.

Efforts to connect issues of forest sustainability with MCDM approach are relatively new. Its applications in this area are mainly related to the assessment of management quality based on the analysis and aggregating of different sustainability indicators in a single index as an overall measure of forest systems sustainability (Mendoza & Prabhu, 2003; Manessi & Farrell 2004). Kant & Lee (2004) used voting techniques and Borda method for the evaluation and ranking of forest management plans with regard to sustainability. In a similar problem Mendoza & Dalton (2005) used AHP, and Huth et al. (2005) PROMETHEE.

In the area of re/af/forestation the first MCDM paper was published by Walker (1985) who developed a methodology for reforestation planning, taking into account several species, silvicultural treatments, etc. More authors combined MAUT and AHP in their approach to this issue (Kangas, 1993; Nousiainen et al., 1998). Liu et al. (1998) used AHP to assess regional forestation projects in China. Giliams et al. (2005) compared AHP, ELECTRE and PROMETHEE in choosing the best afforestation alternative in Belgium.

In the field of regional planning MCDM methods are represented in papers which deal with planning and efficiency of forest management practice in certain national or regional area (Buongiorno & Svanquist, 1982; Faith et al., 1996; Liu et al., 1998). Kangas et al. (2001) analyzed a forest management case in eastern Finland by three multi-criteria techniques:

MAUT, ELECTRE and PROMETHEE. By applying DEA method Kao (1998) measured the efficiency of forest districts in Taiwan, Vennesland (2005) measured the efficiency of subsidies in supporting regional development in Norway and Hiltunen et al. (2008) used five voting methods in strategic forest planning in state forests of Finland.

Considering forestry industry, most papers are related to efficiency evaluation with the use of DEA methods. For example, Yin (1998) analyzed efficiency of 44 paper companies in United States, Nyrud & Bergseng (2002) measured the efficiency of 200 sawmills in Norway, Sowlati & Vahid (2006) evaluated efficiency of Canadian wood product industry, Diaz-Balteiro et al. (2006) analyzed efficiency and innovation activities in Spanish wood industry.

Risk and uncertainty are strongly present in forest management where incomplete data and insufficient information in planning and decision-making often do not allow more accurate assessments and plans. MAUT techniques are therefore most widely used MCDM approach to problem of risk and uncertainty (Pukkala, 1998; Lexer et al., 2000; Ananda & Herath, 2005). Leskinen et al. (2006) used AHP to evaluate the uncertainty associated with the preferences of forest owners in Finland, Kangas (2006) used SMAA for analyzing risks in an actual decision making process.

Cited papers are just some examples of the conducted investigations. The number of MCDM papers in forestry has evolved significantly in the last years. Some authors give a survey of multi-criteria applications in forestry and list more than 250 papers published in major English language journals in the last 30 years (figure 2).

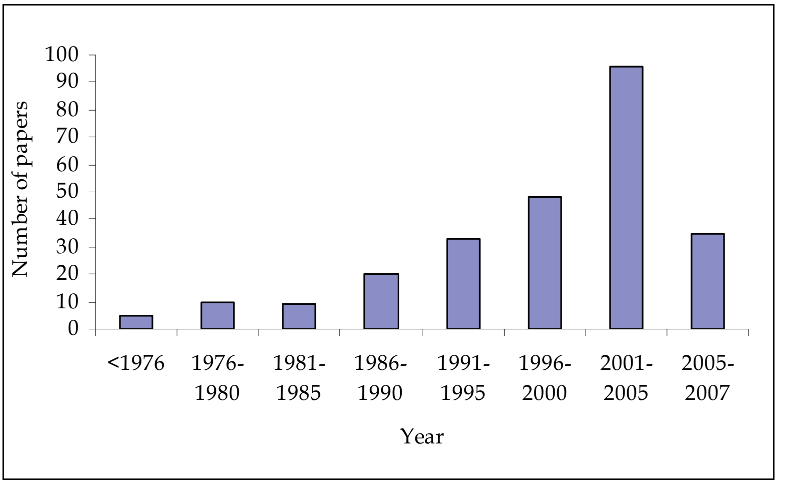

Fig. 2. Number of published multi-criteria papers in forestry (Diaz-Balteiro & Romero, 2008)

The literature also shows that MCDM methods have been applied to a wide range of forestry issues. The main forestry topics in which MCDM methods have been applied could be roughly categorized as already stated to: harvesting; extended harvesting; biodiversity; sustainability, etc. The classification itself cannot be precise because some papers can be divided into several areas or they use more than one method. Still, overview of published papers provides information on the investigated problems and applied MCDM methods in forestry. Figure 3 gives the number of multi-criteria papers in different forestry topics.

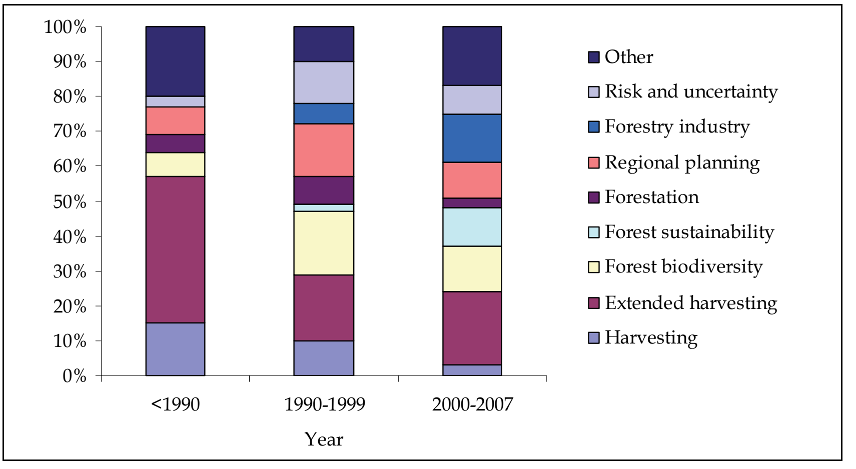

Fig. 3. Shares of MCDM papers in different forestry topics (Diaz-Balteiro & Romero, 2008)

6. Some examples of conducted investigations

This section gives more detailed overview of two investigations where MCDM approach was applied in forestry. Investigations were carried out within research projects and the needs of the state forestry company in the Republic of Croatia. One is related to biological parameters in the evaluation of natural resources (Posavec, 2005), and the other to efficiency of organizational units in forestry (Šporčić, 2007). The presented examples will point out the justification, and the applicability of multi-criteria methods in forestry.

6.1 Selection of biological parameters in the evaluation of natural resources

This research identified the values and value principles applied in evaluation of natural resources. The processed data are related to the Forest Management Unit "Gaj" of the Forest Administration Našice, Croatia. Using the potential method and the eigenvector method, the biological parameters that participate in the calculation of the total value of the natural forest resources were analysed. The adopted premise was that current methods have not been sufficiently exact, so that the new dynamic model should be used for the determination of the forest value. Conducted analysis and the development of new dynamic model included the application of AHP method.

The basic objective was to set up a scientific approach to evaluating a forest resources and establish a model applicable in practice. Parameters needed for forest value assessment, were evaluated by the experts (decision makers) from the field of forestry (Faculty of Forestry, University of Zagreb). Not all of the parameters in the evaluation had the equal weights. To the decision makers, a "verbal scale" for priority expression of one alternative related to another was available. In re-calculation of these verbal priorities into numerical ones, one of the twenty-seven most often used scales was used, as described in Saaty T.L. (1980). The following verbally expressed priorities were considered: Indifference; Moderate Priority; High Priority; Significant Priority; Absolute Priority, and their intermediate degrees, if a decision maker needed them in expressing priorities. Group decision as a potential method consisted of each group member defining their hierarchy, and a consensus at an alternative level (Čaklović et al., 2001). Thereby a group preference graph was made as

a "sum" of individual preference graphs, followed by a group potential. This makes sense particularly if the decision makers do not agree with the criterion choice. Another reason for using the model of group decision was the possibility of measuring the distances among decisions of group members. If group members have coinciding opinions on alternative ranking, there is no need to insist on adjusting the standpoints related to the criterion choice. The comparison by pairs was based on Analytical Hierarchy Process (AHP). The method is supplied by the programme Expert Choice that helps in decision-making on complex issues with several criteria and possible actions. It is designed to model our way of thinking and simplify the process of decision-making (Šegotić, 2001).
In order to be used in a dynamic model for determining the value of selected management unit, the calculated values have been classified in four basic management aims as presented in Figure 4: economic target, management, direct use and indirect use.

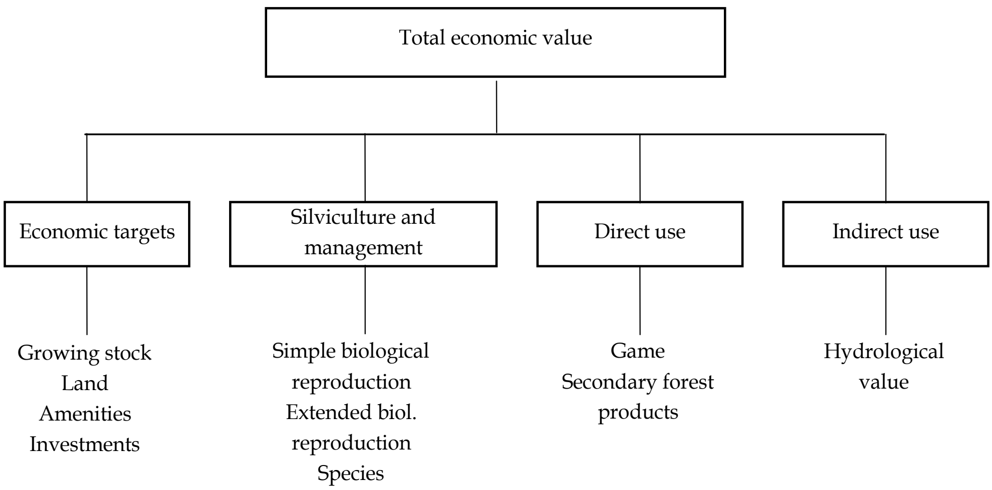

Fig. 4. Hypothetical criteria and parameters to be used in decision-making

Accordingly, forest value is the function of the economical, silvicultural/management, direct and indirect values expressed by the formula:

$$Vf = f\,(Ve + Vu + Vd + Vi)$$

Ve = economic value $(Vgs + Vl + Va + Vi)$
Vu = the value of silviculture and management $(Vsbr + Vebr + Vspecies)$
Vd = direct value $(Vg + Vsfp)$
Vi = indirect value $(Vh + Vnwfp)$
The presented aims and parameters of management represent parts of the common formula for determining the total value (Posavec, 2001). The total value of the management unit was presented through the total sum of the parameters and their weights (w):

$$Vt = (w1\,Vgs)+(w2\,Vl)+(w3Vsbr)+(w4\,Vebr)+(w5Vsfp)+(w6Vg)+(w7\,Vh)+(w8Va)+(w9Vi)+(w10\,Vnwff)+(w11\,Vspecies)$$

Vt = total forest value
Vgs = growing stock value

Vl = land value ($\sum_{i=1}^{i=n} V_z$)
Va = amenities value (reduced by amortisation)
Vi = investments value
Vsbr = value of simple biological reproduction
Vebr = value of extended biological reproduction
Vg = value of hunting management
Vsfp = value of secondary forest products ($\sum_{i=1}^{i=n} V_{sp}$)
Vh = hydrological value
Vnwff = value of non-wood forest functions
Vspecies = value of managed dominant forest species

In table 2, the results are the ranks of all parameters that contribute to forest value and were obtained by potential method. Variable X is the potential value of each parameter. The angle of 18.60 degrees is the measure of inconsistency within the allowed limits. One significant detail is that angle as a measure of group inconsistency does not have any impacts, although the programme displays it. It is significant to measure mutual distances between group members in terms of differences in individual preferences. The obtained distances make up the distance matrix as the basis for the clustering of the group. The sums of total weights form value 1, while individual parameters are presented by their size, which means that the highest priority in this case is the one of non-commercial forest functions.

- Group ranking -
Forest value

Members

Person 1	Person 2	Person 3	Person 4	Person 5	Person 6	Person 7
0.143	0.143	0.143	0.143	0.143	0.143	0.143

showWeights: groupAim base = 2 Options = weight

Level 2: alternatives
Comp_1

Weight = 1.000 InvInc = 0.337 (Angle = 18.60 deg)

Nodes:

Node	Weight	X
nwff	0.121	(X= 0.453)
species	0.119	(X= 0.434)
vsbr	0.106	(X= 0.265)
vgs	0.103	(X= 0.230)
game	0.096	(X= 0.117)
vebr	0.090	(X= 0.022)
vsfp	0.088	(X= -0.008)
vl	0.085	(X= -0.059)
hv	0.082	(X= -0.101)
va	0.059	(X= -0.582)
vi	0.052	(X= -0.770)

Total weight = 1.000

Table 2. Group ranking of parameters by potential method

The AHP model in this case had a very simple structure (according to Figure 4). All parameters were alternatives, and were used for calculating total forest value. Supported by the eigenvector method, an attempt was made to obtain their weights. The basis for calculating the weights were estimates of the experts who carried out the comparisons per pairs of all given parameters. Supported by the programme Expert Choice, five rank lists with parameter weights for calculating forest value were made. An example of one expert's results is shown in Figure 5.

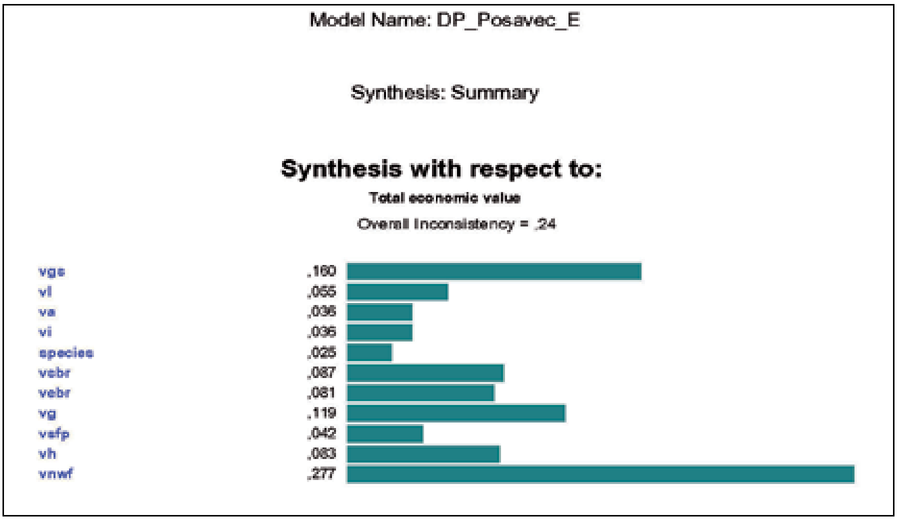

Fig. 5. Parameter rank list of the expert Posavec

If there are additional requirements for individual ranks (i.e. the feeling for forest value), a single rank can be adjusted to given reasons. A special programme can also calculate total forest value independently. In using the potential method, a constant exponential base is set (base value = 2). By changing this base, only relations between ranks can be changed, not their order. The total value of the management unit as calculated by the potential method amounted to 512,301,542.17 kunas (1 eur = 7.4 kunas). The total value calculated by the eigenvector method was 779,716,802.70 kunas. The difference between these two methods, depending on the estimate and parameter ranking, gave a value of 267,415,260.53 kunas. This result shows that a small difference in the size of the ranked parameter results in a great difference in final data. This relates particularly to calculations of the highly estimated non-wood forest values, which have the strongest impact on the final result.

The selected methods are based on pair comparison. Such comparison results in the development of a preference graph, while the number of comparisons per pairs grows in dependence of the given model. The advantage of the analysed dynamic model is obvious due to a decrease in input data. Another advantage of the analysed models is the possibility of clustering of particular groups, i.e. the measurements of the distances between the individual members and their preferences. The disadvantage of these methods is seen in subjective decisions made by individual experts (Posavec et al., 2006).

The developed dynamic models consider the characteristics of forest potentials, and follow the dynamics of the developing conditions within a forest stand, supporting the models of sustainable forest management. The method supports modern evaluation trends in forestry, using available computer program and multi-criteria methods in developing dynamic models for evaluation of forest resources' value.

6.2 Measuring efficiency of organizational units in forestry by nonparametric model

This research assesses the efficiency of basic organizational units in the Croatian forestry, forest offices, by applying Data Envelopment Analysis (DEA). Determination of efficiency is becoming increasingly important in many areas of human activity. Approach to this problem is particularly interesting when there are no clear success parameters, and when the efficiency of using several different resources/inputs is measured for achieving several different outputs. In forestry, the determination of efficiency of forestry companies is extremely complex because of multiple goals of forest management, i.e. its multiple inputs and outputs. In such conditions, the right evaluation method must be used in order to determine whether the resources are used efficiently.

The research included 48 forest offices. The selected forest offices were the representatives of four main regions in Croatian forestry: lowland flood-prone forests (I), hilly forests of the central part (II), mountainous forests (III) and karst/Mediterranean forests (IV). Each region was represented by two forest administrations i.e. by six forest offices from each forest administration (figure 6).

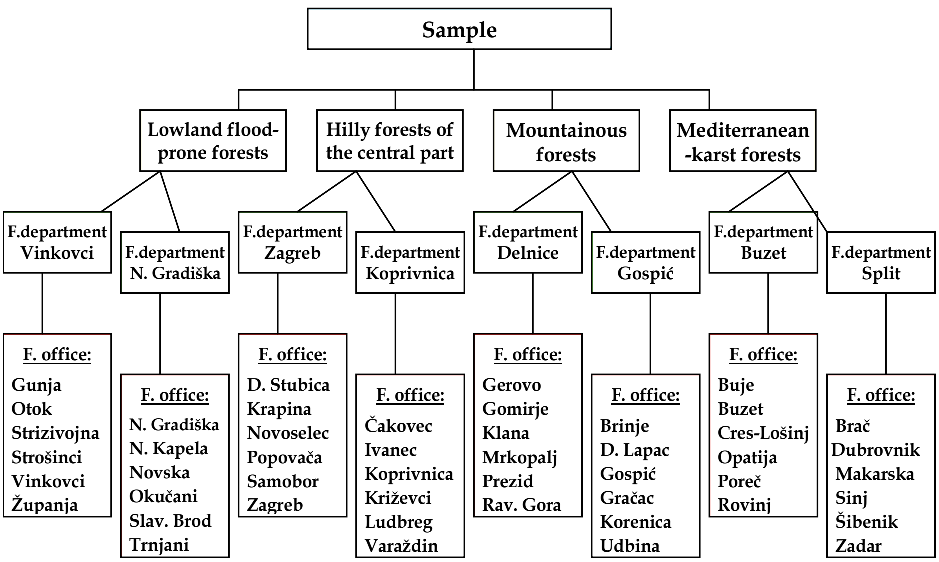

Fig. 6. Sample of the forestry organizational units involved in the research

The relative efficiency of compared forest offices was calculated with the most frequently used DEA models - CCR (Charnes-Cooper-Rhodes) and BCC (Banker-Charnes-Cooper) model. Since DEA was introduced by Charner, Cooper and Rhodes (Charnes et al., 1978) several analytical models have been developed depending on the assumptions underlying the approach. For instance, the orientation of the analysis toward inputs or outputs, the existence of constant or variable (increasing or decreasing) returns to scale and the possibility of controlling inputs. According to Farrell (1957), technical efficiency represents the ability of a decision making unit (DMU) to produce maximum output given a set of inputs and technology (output oriented) or, alternatively, to achieve maximum feasible reductions in input quantities while maintaining its current levels of outputs (input oriented). In this study, output oriented DEA was used, given it is more reasonable to argue that forest area, growing stock and other inputs should not be decreased. Instead, the goal should be increased outputs of forest management, and improved general state of forests. For computing the applied models, DEA Excel Solver software was used.

Given the selected orientation and the diversity of units characterizing the example, CCR model with constant returns to scale was applied first. Following Cooper et al. (2003), analysis began by the commonly used measure of efficiency (output/input ratio) and an attempt to find out the correponding weights by using linear programming. To determine the efficiency of *n* units (forest offices) *n* linear programming problems must be solved to obtain the value of weights (v_i) associated with inputs (x_i), as well as the value of weights (u_r) associated with the outputs (y_r). Assuming *m* inputs and *s* outputs and transforming the fractional programming model into a linear programming model, the CCR model can be formulated as (Cooper et al., 2003):

$$\text{Max} \qquad \theta = u_1 y_{10} + \ldots + u_s y_{s0}$$

$$\text{Subject to:} \qquad v_1 x_{10} + \ldots + v_m x_{m0} = 1$$

$$u_1 y_{1j} + \ldots + u_s y_{sj} - v_1 x_{1j} - \ldots - v_m x_{mj} \leq 0 \quad (j = 1, 2, \ldots, n)$$

$$v_1, v_2, \ldots, v_m \geq 0$$

$$u_1, u_2, \ldots, \mu_s \geq 0 \tag{1}$$

Due to lack of information concerning the form of the efficiency frontier, an extension of CCR model, BCC model was also used. This model incorporates the property of variable returns to scale. The basic formulation of the model is as follows:

$$\text{Max} \qquad \theta = u_1 y_{10} + \ldots + u_s y_{s0} - u_0$$

$$\text{Subject to:} \qquad v_1 x_{10} + \ldots + v_m x_{m0} = 1$$

$$u_1 y_{1j} + \ldots + u_s y_{sj} - v_1 x_{1j} - \ldots - v_m x_{mj} - u_0 \leq 0 \quad (j = 1, 2, \ldots, n)$$

$$v_1, v_2, \ldots, v_m \geq 0$$

$$u_1, \mu_2, \ldots, u_s \geq 0 \tag{2}$$

Where u_0 is the variable allowing identification of the nature of the returns to scale. This model does not predetermine if the value of this variable is positive (increasing returns) or is negative (decreasing returns). The formulation of the output oriented models can be derived directly from models described in (1) and (2) (Cooper et al., 2003).

The identification of inputs and outputs is, besides the choice of the basic model, considered to be the only element of subjectivity in DEA. They were selected so as to reflect business activities of the investigated DMUs – forest offices as the basic organisational units of the Croatian forestry, which perform the basic professional and technical operations in forest management and where the most income and direct costs incur.

As inputs the model included:

- Land, I1 – forest area in thousand hectares
- Growing stock, I2 – volume of forest stock in cubic meters per hectare
- Expenditures, I3 – money spent in hundred-thousand croatian kunas (7,4 kn ≈ 1 EUR)
- Labour, I4 – number of employees in persons

As outputs the model included:

- Revenues, O1 - yearly income in hundred-thousand croatian kunas (7,4 kn ≈ 1 EUR)
- Timber production, O2 – timber harvested in cubic meters per hectare
- Investments in infrastructure, O3 – forest roads built in kilometres
- Biological renewal of forests, O4 – area of conducted silvicultural and protection works in hectares

Table 3 presents the descriptive statistics of the variables used in the analysis. A wide variation in both inputs and outputs is noticeable. Such variation is not unexpected, since the sample involves all representative areas managed by Croatian forests Ltd. However, it may also be a sign of bad management of resources in individual forest offices.

Variable	Mean	St. deviation	Min	Max	Total
Inputs					
Area, 10^3 ha	11.42	10.36	2.60	49.87	547.96
G. stock, m^3/ha	214.98	91.94	51.85	418.00	-
Costs, 10^5 kn	152.35	93.61	23.24	470.31	7312.99
Employees, N	42	21	8	100	2.007
Outputs					
Income, 10^5 kn	157.20	106.40	21.12	538.41	7545.68
Harvest, m^3/ha	3.06	2.19	0.00	8.78	-
Investments, km	2.24	4.29	0.00	22.59	107.48
B. renewal, ha	422.26	606.34	30.21	3846.34	20268.47

Table 3. Descriptive statistics of the variables used in the DEA model

Technical efficiency was determined individually for each forest office. The average CCR efficiency of the investigated forest offices was 0.829, which means that an average (assumed) forest office should only use 82.9% of the currently used quantity of inputs and produce the same quantity of the currently produced outputs, if it wishes to do business at the efficiency frontier. In other words, this average organisational unit, if it wishes to do business efficiently, should produce 20.6%[2] more output with the same input level. According to the BCC model, the average efficiency is 0.904. This means that an average forest office should only use 90.4% of the current input and produce the same quantity of output, if it wishes to be efficient. In other words, to be BCC efficient it should produce 10.6%[3] more outputs with the same inputs. Scale efficiency (ratio between CCR and BCC scores) shows how close or far the size of the observed unit is from its optimal size. The scale efficiency of 0.919 means that the analysed forest offices would increase their relative efficiency on average by 8% if they adapted their size or volume of activities to the optimal value. The main results obtained by the output-oriented DEA are given in table 4.

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[2] It can be easily obtained that 20.6 % = (1 – 0.829)/0.829

[3] It can be easily obtained that 10.6 % = (1 – 0.904)/0.904

	CCR model	BCC model	Scale efficiency
Number of forest offices (DMUs)	48	48	48
Relatively efficient DMUs	15	24	16
Relatively efficient DMUs (in %)	31 %	50 %	33 %
Average relative efficiency, E	0.829	0.904	0.919
Maximum	1,000	1,000	1,000
Minimum	0.407	0.524	0.501
Standard deviation	1.170	0.129	0.138
DMUs with efficiency lower than E	23	18	12

Table 4. Results obtained with the base case DEA models

Based on the efficiency results of forest offices grouped according to their structural characteristics (surface area, growing stock, number of employees), it has been determined that the highest levels of efficiency were recorded for forest offices that manage 10 to 15.000 hectares, and for the forest offices with growing stock ranging between 200 and 300 m^3/ha i.e. over 300 m^3/ha. It has been also determined that the highest level of efficiency is achieved by forest offices with the highest number of employees, and that forest offices in the region of flood-prone forests have the highest efficiency scores.
DEA method gives to management the possibility to rank organizational units based on analysis and comparisons of their relative efficiency. For inefficient units the projections on the efficiency frontier and the sources of inefficiency are determined. In this way, potential changes in inputs/outputs required to achieve technical efficiency are determined, and the objectives which inefficient units should fulfil in order to become efficient are recognized.

7. Conclusion

In the last twenty years or so, the general framework of forest management has changed dramatically. Multiple goals are today typical for forestry. Forest management has to produce a certain revenues while at the same time it should promote protection and preservation of forests, recreational services, etc. In addition to harvesting and wood production, some other criteria are receiving increased attention in choosing the ways of forest management. In other words, forests are simultaneously used for multiple purposes. Multiple benefits and many advantages provided by forests as well as the non-market nature of a part of these products, make the planning and decision making in forestry especially demanding. This has led to a need for models that can be applied in multifunctional sustainable forest management. In particular, such support, through various methods and models is needed in planning and predicting, as well as in the analysis of forest management results.
Forest management involves the decision making related to the organisation, use and conservation of forests. Management decisions are made both for long-term planning and daily activities. Good forest management requires solution of the issues related to problems of energy, raw materials and life quality. Mathematical models are not new in forestry. The multiplicity of the available data on forests requires computer-aided mathematical methods. The problems of forest management involve a variety of different variables. They may be biological, such as growth and increment, type of soil; economic, such as the price of timber and labour costs; and social, such as ecological laws. All these

variables and their interrelations make up a system. The complexity of forestry systems makes predicting of consequences of the decisions made a difficult task. This is where models come in use. Most models calculate the consequences of particular decisions. The models may be classified according to their properties. Thus, they may be deterministic, or stochastic; they may optimise one, or several goals and they use a particular algorithm. First models used linear programming. Many authors used dynamic programming as a method for making a series of optimal decisions (Amidon & Atkin, 1968; Brodie & Kao, 1979; Zadnik, 1990). Realising the necessity of stochastic methods in forest management, the Markov process was introduced into forestry (Hool, 1966). Multiple uses in forest management were first expressed through goal programming (Field, 1973; Mendoza, 1986). Goal programming uses the weights that are the reflections of the significances of each criterion. To join these weights is the greatest problem in goal programming, so that different authors suggested different methods (Bare & Mendoza, 1988; Gong, 1992; Howard & Nelson, 1993). Group decision is the most complex form of decision-making. It basically does not differ from the multi-criteria decision. The only difference is organisation of hierarchy and the sequence of taking the individual steps in the decision.

Planning and decision making in forestry is especially complex because of multiple objectives of forest management, and numerous wide ranging, often hardly comparable and conflicting criteria and interests that influence the decision making process. Multi-criteria methods can thereby facilitate the decision making process and reduce the risks and challenges in today's demanding and complex forest management planning. It is sure that MCDM and operations research can not resolve all issues and problems in forestry, but they can serve as a platform on which the results of different scientific fields can be used in a comprehensive decision-making process.

It should also be pointed out that managing any organization requires the ability to effectively assess and analyze information generated in the business process. For organizations, such as forestry companies, which manage natural resources and by business decisions affect the environment, that is from the viewpoint of ecological acceptability and environmental management even more critical. Development and application of methods that have not been traditionally used in natural resources management can provide a valuable assistance at the strategic, tactical and operational level of planning and decision making. Methods that have in this respect experienced a wide range of applications in recent years are for example AHP and DEA.

This paper, besides AHP and DEA also presents the other major MCDM methods: MAUT, ouranking methods, voting methods and SMAA. Paper gives the basic features of methods and a brief overview of forestry problems and areas where they have been applied so far. The aim was to provide information on existing experience, and thus contribute to making forestry profession aware of the significance and potential role that MCDM methods can play in forestry. Many cited articles can also be a valuable reference source for students, researchers, forestry experts and practitioners. The results show that in the last 30 years a significant number of forestry MCDM papers was published dealing with various forestry issues and problems such as harvesting, biodiversity, sustainability, regional planning, etc. Frequency of published papers shows that the number of such papers is increasing at a very high rate what indicates a trend of increased use of MCDM in forestry in recent years.

In this very dynamic period of natural resources management, when forestry experts face the challenges of professional and responsible management of forests and forest land, having to observe at the same time the protection requirements of their ecological, social and economic functions, as well as challenges of profitable management of forestry companies, managers need different models for converting natural, accounting, financial and many other variables and data into useful information. This paper points to the justification and possibilities of application of MCDM in multifunctional forest management, with the emphases on conservation of biodiversity, regeneration capacity and sustainable management. Paper also shows how multi-criteria methods can be used for analyzing the choice of the best or at least satisfactory decision, and thus contribute to more reliable planning and more objective decision making in forestry. It is generally considered that MCDM methods in forestry, as well as in other business systems, can be a very strong support to management and decision making.

8. References

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23

A Decision-Support Model for Regulating Black Spruce Site Occupancy Through Density Management

P. F. Newton
Canadian Wood Fibre Centre, Canadian Forest Service, Natural Resources Canada
Canada

1. Introduction

Regulating site occupancy through stand density management has been a cornerstone of silvicultural practice since it was first introduced in forestry by Reventlow in 1879 (Pretzsch, 2009). Density management continues to be a dominant intensive forest management practice throughout boreal and temperate forest regions (e.g., Canada (CCFM, 2009) and Finland (Peltola, 2009) treat over 500,000 ha annually). Operationally, density management consists of manipulating initial planting densities at the time of establishment (initial espacement; IE) and (or) reducing stand densities during subsequent stages of stand development (e.g., precommercial thinning (PCT) at the sapling stage, and (or) commercial thinning (CT) at the semi-mature stage). As documented by numerous case studies, density management can result in a wide array of benefits at the tree, stand and forest levels. These include increased growth and resultant yields leading to enhanced end-products (e.g., Kang et al., 2004), attainment of early stand operability status (e.g., Erdle, 2000), reduced density-dependent mortality losses (e.g., Pelletier & Pitt, 2008), increased spatial and structural uniformity resulting in lower extraction, processing and manufacturing costs (e.g., Tong et al., 2005), and increased carbon sequestration rates (e.g., Nilsen & Strand, 2008). Density management also has consequential effects on other important non-timber values. These include regulating the production of coarse woody debris to met wildlife habitat requirements (e.g., pine marten (*Martes americana*) (Sturtevant et al., 1996)), provision of thermal protection and hiding requirements for ungulates by regulating stand structure (e.g., elk (*Cervus elaphus nelsonii*) and mule deer (*Odocoileus hemionus*) (Smith and Long, 1987)), controlling successional pathways in order to prevent the establishment and development of ericaceous shrub species (e.g., Lindh and Muir, 2004), and increasing biodiversity (e.g., Verschuyl et al., 2011). Although thinning effects are largely positive in nature, inappropriate treatments can have serious detrimental implications. These include (1) PCT treatments which result in an extended period of openness in which individual trees are allowed to build up extensive crowns resulting in an increase in juvenile wood production and larger knot sizes (e.g., Tong et al., 2009), and (2) CT treatments which are implemented within structurally unstable stands resulting in increased mortality during high wind or heavy ice and snow events.

Determination of the optimal density management regime for a given objective is a complex process given the multitude of variables that a forest manager needs to consider. For example, deciding on initial establishment densities, the timing of thinning entries and associated removables, discount and interest rates, and fixed and variable cost values. Furthermore, the selected regime must be considered within the broader regulatory framework which can impose additional constraints on the decision-making process (e.g., specific minimum pre-treatment tree size and basal area requirements before CT treatments can be implemented (McKinnon et al., 2006)). Fortunately, however, the complexity of decision-making has been greatly reduced for traditional volumetric-based objectives with the advent of stand density management diagrams (SDMDs; Ando, 1962; Drew & Flewelling, 1979; Jack & Long, 1996; Newton, 1997).

Briefly, SDMDs are graphical decision-support tools that are used to determine the density management regime required for the realization of a specified mean tree size or volumetric yield objective. Recently, in order to address the evolving paradigm shift in management focus from a singular volumetric yield maximization objective to a focus on a multitude of diverse objectives, including the end-product quality (Barbour & Kellogg, 1990), product value maximization (Emmett, 2006), bioenergy and carbon sequestration potential, and ecosystem services, the SDMD modeling framework was expanded. Specifically, Newton (2009) introduced the modular-based structural stand density management model (SSDMM) for jack pine (*Pinus banksiana* Lamb.) stand-types. The model has a hierarchical design in which 6 integrated estimation modules collectively enable the estimation of volumetric productivity, log distributions, product volumes and values, and fibre attributes, for a given density management regime, site quality, and cost profile.

The objectives of this study were to describe the upland black spruce (*Picea mariana* (Mill.) BSP) variant of the modular-based SSDMM and demonstrate its utility in designing density management regimes within an operational context. More specifically, the stand-level examples are placed within the broader context of sustainable management at the landscape level in which a portion of the productive forest land base is allocated and managed for timber related objectives (i.e., early operability within natural-origin stands, and production of enhanced end-products within plantations) and the remainder, for non-timber related objectives (i.e., production of coarse woody debris (CWD) for maintenance of wildlife habitat).

2. Methods

2.1 Modular-based SSDMM for upland black spruce stands

The SSDMM for upland black spruce stands was developed by expanding the dynamic SDMD modelling framework through the incorporation of diameter, height, log-type, biomass, carbon, product and value distribution, and wood quality recovery modules (Figure 1). Analytically, the principal steps involved the development of a dynamic SDMD and the subsequent incorporating of (1) a parameter prediction equation (PPE) system for diameter distribution recovery, (2) a composite height-diameter prediction equation for height estimation, (3) a composite taper equation for recovering log product distributions and calculating stem volumes, (4) composite biomass equations for estimating above ground components and their carbon mass equivalents, (5) sawmill-specific product recovery and associated product value functions, and (6) composite wood density and maximum mean branch diameter equations. Computationally, Module A (Dynamic SDMD) provides a set of annual stand-level variables which are required as input to Modules B-F. Module B utilizes the PPE system and the composite height-diameter function to recover the grouped-diameter frequency distribution and estimate corresponding tree heights for each diameter class (Diameter and Height Recovery Module), and similar to Module A, provides prerequisite input to the remaining modules. The taper equation is used to derive estimates of the upper stem diameters for each tree within each diameter class from which the number of sawlogs and pulplogs, residual tip volumes, and merchantable and total stem volumes, are calculated (Taper Analysis and Log Estimation Module). The composite biomass equations are used to predict masses and carbon equivalents for each above-ground component (Biomass and Carbon Estimation Module). The product recovery and value functions are used to predict sawmill-specific (stud mill (SM) and randomized length mill (RLM)) chip and lumber volumes and associated market-based monetary values (Product and Value Estimation Module). The composite fibre attribute functions are used to estimate mean wood density for merchantable-sized (≥ 10 cm diameter classes) trees, and the mean maximum branch diameter within the first 5 m sawlog for trees ≥ 15.1 cm in diameter (Fibre Attribute Estimation Module). Refer to Newton (2012a) for a complete description of the approach used in the development and calibration of the modular-based SSDMM for upland black spruce stands.

Given the model's complexity and the computation burden associated with its use, an algorithmic analogue was developed in the Visual Basic (VB.NET (Ver. 1.1); Microsoft Corporation) programming language. Denoted, Croplanner, the program predicts and tabulates site-dependent annual and rotational diameter-class and stand-level estimates of volumetric yields, log distributions, biomass and carbon outcomes, recoverable products and associated values by sawmill-type, economic efficiency profiles and fibre attributes, for 3 density management regimes per simulation. The user is required to specify the following information for each simulation: (1) provincial region (e.g., Ontario); (2) stand-type (natural origin or plantation); (3) simulation year; (4) site quality (site index); (5) rotational age; (6) establishment densities; (7) expected ingress during the establishment period (n., applicable to plantations only); (8) merchantable specifications (i.e., length and upper threshold diameters for pulp and saw logs, and merchantable top diameter); (9) interest and discount rates; (10) operability targets (i.e., number of merchantable trees per cubic metric of merchantable wood, and total merchantable volume per unit area); (11) establishment costs (e.g., fixed site assessment or preparation expenses and planting costs); (12) genetic worth effects and selection ages (n., applicable to plantations only); (13) operational adjustment factors; (14) product degrade estimates; (15) variable cost estimates accounting for stumpage and renewal charges, harvesting, transportation and manufacturing expenses at the time of harvest; and (16) regime-specific thinning treatments and associated costs (i.e., time of entry (stand age), type of thinning (PCT or CT), removal densities (stems/ha) or basal area (%) reductions, and fixed and variable thinning cost values).

For each year, the program recovers the grouped-diameter frequency distribution and for each recovered diameter class, calculates height, number of pulp and saw logs, merchantable and total volumes, biomass and carbon equivalents for each above-ground component (bark, stem, branch and foliage), sawmill-specific recoverable chip and lumber volumes and associated monetary values, and mean tree fibre attributes. Cumulative stand-level values and performance indices are subsequently derived. The output is presented in both tabular and graphical formats and consists principally of a traditional SDMD graphic, regime- specific annual estimates at the individual diameter-class level and stand-levels, regime-specific treatment and rotational summaries, and across-regime rotational comparisons. The comparisons employ a comprehensive set of performance indices which include measures of (1) overall productivity as measured by the mean annual merchantable

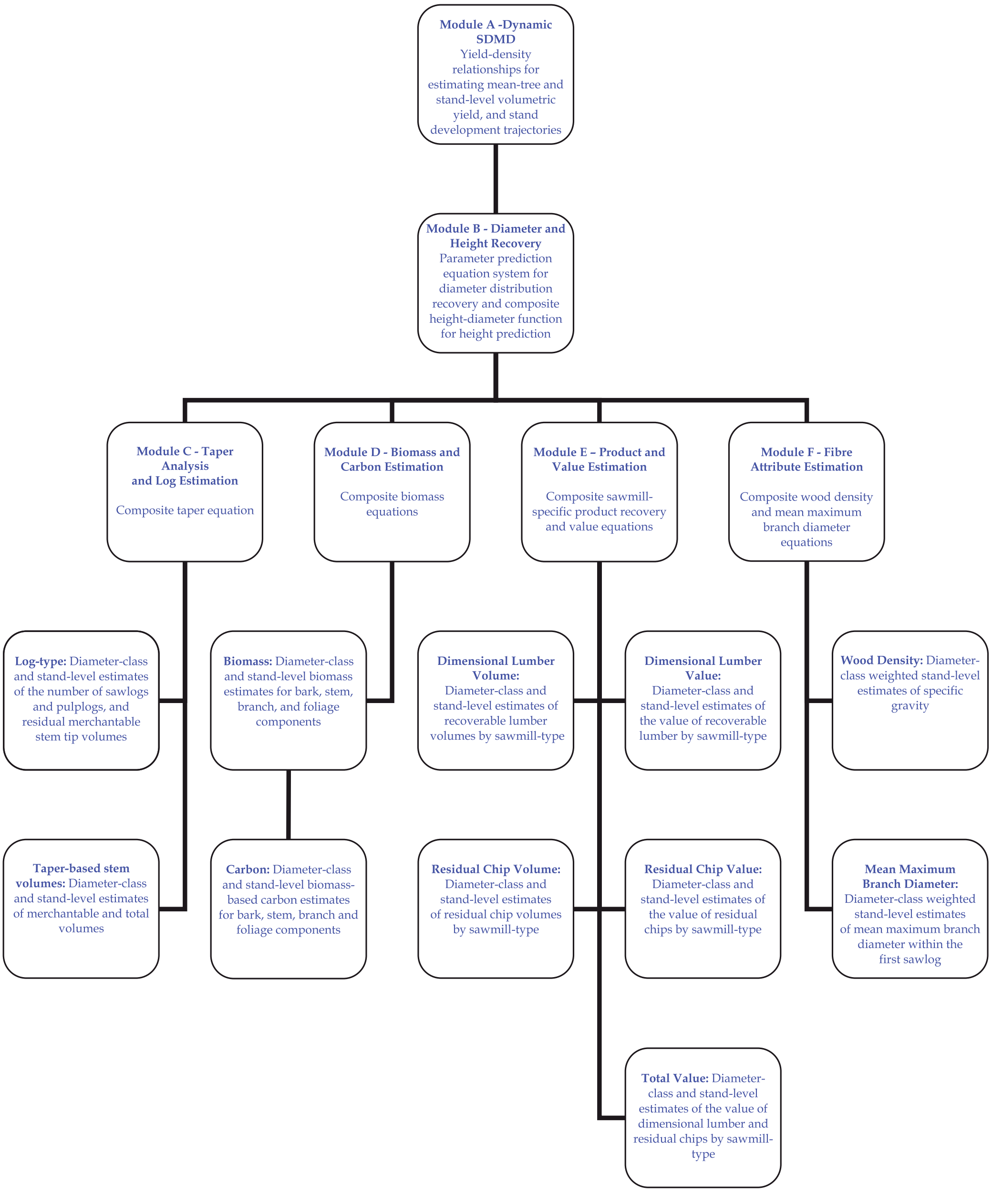

Fig. 1. Schematic illustration of the modular-based SSDMM.

volume increment (m^3/ha/yr), mean annual biomass increment (t/ha/yr) and mean annual carbon increment (t/ha/yr), (2) log production in terms of the percentage by sawlogs produced, (3) end-products recovered as quantified by the percentage of lumber volume produced by each sawmill type, (4) economic efficiency based on land expectation values (i.e., the maximum an investor could pay for bare land to achieve a specified rate of return (discount rate)) of a given manipulated regime relative to the control regime for each sawmill type), (5) optimal site occupancy (number of years that a size-density trajectory was within an optimal production zone as delineated by relative density indices of 0.32 and 0.45 (Newton, 2006)), (6) stand stability as reflected by the mean height/diameter ratio for trees within the dominant crown class, (7) fibre quality attributes as summarized by mean wood density and mean maximum branch diameter, (8) accelerated operability based on the reduction in the number of years that a stand took to reach harvestable status as defined by target piece size and merchantable yield thresholds, and (9) time to full occupancy as quantified by the number of years required to reach initial crown closure status.

2.2 Simulations

The treatment regimes as stated within an operational forest management plan are used to exemplify the utility of model. Specifically, the silvicultural matrix presented in the 2009-2019 forest management plan developed for the Romeo Malette Forest in the Timmins District of the Northeastern Region of Ontario, Canada, by Tembec Inc. (Anonymous, 2009), was used. These ecosite-specific treatment regimes reflect best management practices for a given stand and forest management objective as defined within the NEBIE silvicultural intensity framework (Bell et al., 2008).

For the natural regenerated stand-type (forest unit SP1 (Ecosite 2)), an extensive silvicultural intensity employing an early operability objective, was evaluated. For the plantations (forest unit SP1 (Ecosite 5f)), an elite silvicultural intensity with an enhanced end-product value objective, was evaluated. These objectives reflect ongoing discussions regarding the management of boreal conifers in the central portion of the Canadian Boreal Forest Region: (1) implementing PCT treatments within density-stressed natural-origin stands in order to shorten the time to operability status; and (2) employing CT treatments within genetically-improved plantations so that merchantable volume losses normally attributed to density-dependent mortality at the later stages of stand development are minimized, and reducing the technical rotation age in regards to the production of high quality wood products.

The protocol for implementing the CT treatments followed the provincial recommendations as espoused by McKinnon et al. (2006). Specifically, preferable CT density management regimes are those which (1) increased mean tree size without incurring declines in stand volume growth, (2) do not unacceptably increase the risk of volume losses to wind, snow, insects, and disease, and (3) minimize the rate of density-dependent mortality within the merchantable-sized classes during the later stages of stand development thus enabling the recovery of some of the expected merchantable volume losses through thinning. Operationally, the CT treatment should occur within previously density regulated stands which are approximately 15-20 yrs from rotation age. The CT treatment should reduce basal areas by a maximum of 30-35% from an initial minimum basal area of 25 m^2/ha and be implemented only when density-dependent mortality is occurring or imminent within the merchantable-sized classes. Lastly, CT treatments should only occur within stands where the mean live crown ratio exceed 35%. Table 1 provides a summary of the input parameters required to run these scenarios with the Croplanner algorithm.

3. Results and discussion

3.1 Extensive silviculture: Natural-origin black spruce stand-types subjected to PCT

The resultant mean volume-density trajectories for the natural-origin black spruce stands within the context of the traditional SDMD graphical format are illustrated in Figure 2. It is

Input Parameter (unit)	Stand-type and Treatment					
	Natural-origin Stands subjected to PCT			Plantations subjected to IE+PCT+CT with genetic worth effects		
	Regime 1 – Control	Regime 2 - PCT	Regime 3 – PCT	Regime 1 – Control	Regime 2 - PCT	Regime 3 – PCT+CT
Silvicultural intensity	Extensive			Elite		
Objective	Early operability			End-product value		
Simulation year	2011	2011	2011	2011	2011	2011
Site index (Carmean et al., 2006)	16	16	16	18	18	18
Rotation age (yr)	80	80	80	50	50	50
Initial density (stems/ha)	5000	5000	5000	2750	2750	2750
Ingress density (stems/ha)	-	-	-	0	0	0
Merchantable specifications						
Pulplog length (m)	2.59	2.59	2.59	2.59	2.59	2.59
Pulplog minimum diameter (cm)	10	10	10	10	10	10
Sawlog length (m)	5.03	5.03	5.03	5.03	5.03	5.03
Sawlog minimum diameter (cm)	14	14	14	14	14	14
Merchantable top diameter (cm)	4	4	4	4	4	4
Rates						
Interest rate (%)	2	2	2	2	2	2
Discount rate (%)	4	4	4	4	4	4
Operability Targets						
Piece-size (stems/m^3)	10	10	10	10	10	10
Merchantable yield (m^3/ha)	130	130	130	200	200	200
Site preparation ($/ha)	100	100	100	300	300	300
Planting ($/seedling)	-	-	-	0.6	0.6	0.6
Genetic worth (%)	-	-	-	15	15	15
Selection age (yr)	-	-	-	15	15	15
Operational adjustment factor (%)	1	1	1	1	1	1
Product degrade (%)	15	5	5	15	10	5
Variable costs for harvesting, stumpage, renewal, transportation and manufacturing ($/m^3)	100	80	80	75	65	55
PCT Treatments						
Time of treatment (yr)	-	14	14	-	13	13
Number of trees removed (stems/ha)	-	1943	2943	-	907	907
Fixed cost of PCT ($/ha)	-	300	300	-	300	300
CT Treatment						
Time of treatment (stems/ha)	-	-	-	-	-	30
Number of trees removal (stems/ha)	-	-	-	-	-	604
Fixed cost of CT ($/ha)	-	-	-	-	-	100
Variable cost for harvesting, stumpage, transportation and manufacturing for volume removed ($/m^3)	-	-	-	-	-	65

Table 1. Stand-type specific input parameters used in the Croplanner simulations.

instructive to familiarize oneself with the overall structure of the diagram, particularly, in relation to the static and dynamic components. Essentially, the yield-density isolines are used for positioning a given stand in the size-density space and deriving corresponding yield estimates. The size-density trajectories in combination with the isolines provide a graphical pictorial of overall stand dynamics (density changes due to thinning treatments and density-dependent and independent mortality) in addition to enabling users to derived structural characteristics at various key phases of stand development, through interpolation. For example, the intersection of the size-density trajectories with the diagonal line denoting crown closure status indicated that the stand thinned to a residual density of 3000 trees (stems/ha; Regime 2) re-attained crown closure status by an age of 18 yr whereas the stand thinned to a residual density of 2000 trees (stems/ha; Regime 3) re-attained crown closure status by an age of 22 yr. Knowing the period of time a stand is open-grown is an important metric when attempting to control early branch development within the lower portion of the stem through density regulation.

The graphic also shows that at an approximate mean dominant height value of 10 m, the stands enter a period of accelerated self-thinning, as evident from the degree of curvature of the size-density trajectories. The degree of self-thinning was most pronounced in the control stand and less so for the PCT treated stands. Numerically, from the time of treatment to rotation, the unthinned control stand lost 3373 trees (stems/ha; Regime 1) compared with only 1746 trees (stems/ha) for Regime 2, and 1124 trees (stems/ha) for Regime 3. By the time the stands reached rotation age (80 yr) they were positioned just below the 20 m mean dominant height isoline. The control stand was just below the 18 cm quadratic mean diameter isoline, just above the 0.9 relative density index isoline, and just below the 35% mean live crown ratio isoline. For the thinned stand PCT to a residual density of 3000 stems/ha (Regime 2), the trajectory terminated at a position that was slightly above the 18 cm quadratic mean diameter isoline, just above the 0.8 relative density index isoline, and slightly below the 35% mean live crown ratio isoline. Similarly, for the thinned stand PCT to a residual density of 2000 stems/ha (Regime 3), the trajectory terminated at a position that intersected the 20 cm quadratic diameter isoline, just above the 0.7 relative density index isoline, and intersected the 35% mean live crown ratio isoline. Although the graphic is very useful in terms of understanding and visualizing stand development, the algorithmic revision readily facilitates the estimation of a much broader array of yield, end-product, economic, and wood fibre attribute metrics (Table 2), and associated performance measures (Table 3), at various temporal scales (annual, periodic and rotational).

The thinning treatments resulted in an increase in the duration of the pre-crown-closure period by 4 and 8 yr for Regimes 2 and 3, respectively. Given that the dominant height of the stands would be in the 5.5 to 6.5 m range at time of re-closure, most of the branches within the first 5 m long sawlog would have been formed by then. As inferred by the minimal differential in mean maximum branch diameters at rotation between the stands (c.f., 2.65 cm versus a mean of 2.70 cm for the control and thinned stands, respectively; Table 3), suggest that this extended period of openness did not consequentially affect branch development within this economically-important portion of the stem. Comparing Regimes 2 and 3 against Regime 1, indicated that on the positive side, the PCT treatment (1) shorten the time to stand operability status by an average of 8 years, (2) produced trees of large mean size at rotation (i.e., average increases in mean volume of 32%), (3) increased the percentage of sawlogs produced by an average of 12%, and (4) enhanced overall structural stability (e.g., reducing the height/diameter ratio by an average of 10%). On the negative side, however, the single PCT treatment resulted in lower per unit yields for merchantable volume (average of 12% less), and biomass and carbon production (average of 18% less). Economically, however, the PCT treatments did result in gains in economic efficiency (an average of 88% increase) at the specified rotation age of 80 yr, irrespectively of sawmill type. These economic differences can be largely attributed to the lower product degrade values specified for the thinned stands, and to the assumed reduction in variable costs at the time of harvest arising from decreased harvesting and manufacturing expenses due to increased piece-size,

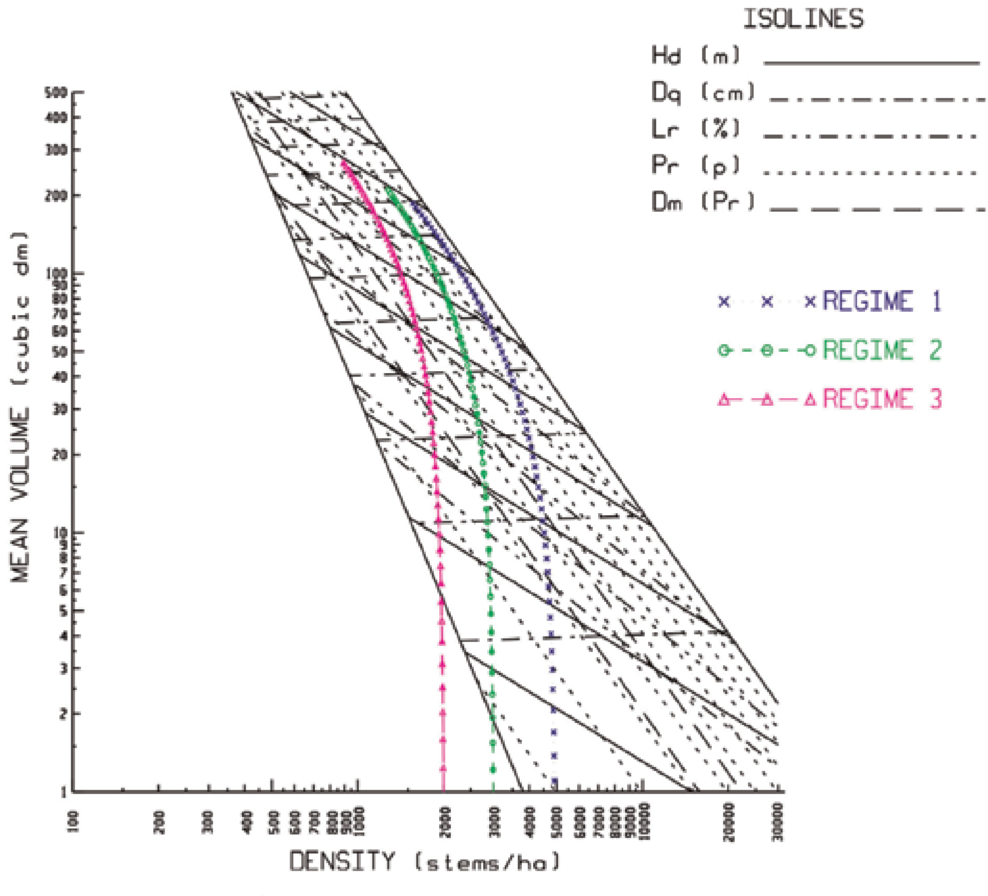

Fig. 2. Dynamic SDMDs for natural-origin upland black spruce stand-types managed under an extensive silvicultural intensity. Graphically illustrating (1) isolines for mean dominant height (Hd; 6-22 m by 2 m intervals), quadratic mean diameter (Dq; 4-26 cm by 2 cm intervals), mean live crown ratio (Lr; 35, 40, 50,…, 80%), and relative density index (Pr; 0.1-1.0 by 0.1 intervals), (2) the self-thinning line at a Pr = 1.0, and initial crown closure line (lower solid diagonal line); (3) lower and upper Pr values delineating the optimal density management window (Dm; 0.32 ≤ Pr ≤ 0.45); and (4) expected 80-yr size-density trajectories with 1 year intervals denoted for 3 user-specified density management regimes for stands situated on a medium site quality (site index = 16).

reduced size variation, and more uniform spatial patterns. In terms of provision of wildlife trees, the number of large standing snags (trees/ha), as approximated by the number of merchantable-sized abiotic trees which died during the last decade before harvest, was 41% less in the PCT stands as compared to the control stand.

Attribute (unit)	Regime 1 - Control	Regime 2 – PCT (thinning yields)	Regime 3 – PCT (thinning yields)
Mean dominant height (m)	19.9	19.9	19.9
Quadratic mean diameter (cm)	18	19	20
Basal area (m^2/ha)	39	35	32
Mean volume per tree (dm^3)	186	216	252
Total volume (m^3/ha)	291	272 (4)	242(6)
Total merchantable volume (m^3/ha)	276	257 (0)	229(0)
Density (stems/ha)	1561	1254 (1943)	876 (2943)
Relative density index (%/100)	0.91	0.81	0.66
Mean live crown ratio (%)	33	34	35
Number of pulplogs (logs/ha)	3149	2345 (-)	1531 (-)
Number of sawlogs (logs/ha)	724	846 (-)	782 (-)
Residual log tip volume (m^3/ha)	53	39 (-)	28 (-)
Bark biomass (t/ha)	19	17 (-)	15 (-)
Stem biomass (t/ha)	188	164 (-)	139 (-)
Branch biomass (t/ha)	6	6 (-)	6 (-)
Foliage biomass (t/ha)	11	11 (-)	12 (-)
Total biomass (t/ha)	224	198 (-)	172 (-)
Bark carbon (t/ha)	10	9 (-)	8 (-)
Stem carbon (t/ha)	94	82 (-)	70 (-)
Branch carbon (t/ha)	3	3 (-)	3 (-)
Foliage carbon (t/ha)	5	5 (-)	6 (-)
Total carbon (t/ha)	112	99 (-)	86 (-)
Chip volume – SM (m^3/ha)	127	110 (-)	90 (-)
Lumber volume – SM (m^3/ha)	149	133 (-)	122 (-)
Chip volume – RLM (m^3/ha)	108	94 (-)	77 (-)
Lumber volume – RLM (m^3/ha)	166	148 (-)	134 (-)
Chip value – SM ($K/ha)	6	6 (-)	5 (-)
Lumber value – SM ($K/ha)	27	27 (-)	26 (-)
Total product value – SM ($K/ha)	33	33 (-)	31 (-)
Chip value – RLM ($K /ha)	6	5 (-)	4 (-)
Lumber value – RLM ($K /ha)	36	37 (-)	35 (-)
Total product value – RLM ($K/ha)	42	42 (-)	39 (-)
Land expectation value – SM ($K/ha)	1.2	2.7	2.6
Land expectation value RLM - ($K/ha)	3.2	4.7	4.3

Table 2. Rotational yield estimates for upland black spruce natural-origin stands subjected to PCT. Values in parenthesis denote yields derived from the PCT treatment (n., a dash line indicates an incalculable value).

Index (unit)	Regime 1 - Control	Regime 2 - PCT	Regime 3 – PCT
Mean annual volume increment (m^3/ha/yr)	3.4	3.2	2.9
Mean annual biomass increment (t/ha/yr)	2.8	2.5	2.1
Mean annual carbon increment (t/ha/yr)	1.4	1.2	1.1
Percentage of sawlogs produced (%)	19	27	34
Percentage of lumber volume recovered - SM (%)	54	55	57
Percentage of lumber volume recovered - RLM (%)	61	61	64
Relative land expectation value – SM (%)	-	138	49
Relative land expectation value – RLM (%)	-	129	36
Duration of optimal site occupancy (%)	9	11	16
Mean height/diameter ratio (m/m)	103	96	90
Mean wood density (g/cm^3)	0.48	0.49	0.49
Mean maximum branch diameter (cm)	2.65	2.68	2.72
Time to operability status (yr)	64	58	55
Time to initial crown closure (yr)	14	14	14
Age of crown re-closure post-PCT (yr)	-	18	22
Number of large standing snags (trees/ha)	332	228	165

Table 3. Stand-level performance indices for density-manipulated upland black spruce natural-origin stands subjected to PCT.

In summary, this specific simulation indicated that PCT resulted in (1) earlier stand operability status, (2) larger but fewer trees at rotation, (3) an increased in the duration of optimal site occupancy, (4) enhanced structural stability, (5) a decline in overall merchantable volume productivity, and (6) production of fewer wildlife trees.

3.2 Elite silviculture: Genetically-improved upland black spruce plantations subjected to PCT and CT

Similar to the PCT treatments within the natural-origin stands, the resultant mean volume-density trajectories for elite treatments are graphically illustrated within the context of the SDMD graphic (Figure 3). Table 4 lists the rotational and thinning yield estimates whereas Table 5 lists the resultant stand-level performance indices. Although self-thinning occurred within all 3 regimes indicating full occupancy had been achieved, the rate of density-dependent mortality increased with increasing planting density. The PCT treatments extended the period of openness by approximately 3 yr, however the effect of branch development was minimal (c.f., 2.65 cm for the control stand versus 2.69 and 2.72 cm for the PCT and PCT+CT stands, respectively). The trajectories also revealed that the thinned stands spent a greater portion of the rotation in the optimal site occupancy zone: 20% and 44% for the PCT and PCT+CT regimes, respectively, versus 12% for the control stand. This suggest that the thinned stands, particularly the stand that received a dual treatment (Regime 3), the rate of carbon sequestration and biomass production was close to an optimal level for a considerable portion of the rotation. Essentially, stands below the zone are not fully utilizing the site and consequently site resources are going unused in terms of forest biomass production (e.g., resource supply exceeds demand). Stands above the zone are over-occupying the site resulting in intensive asymmetric resource competition among local neighbors and subsequent mortality through self-thinning.

Further examination of the SDMD revealed that the size-density trajectories intersected the crown closure isoline slightly above the 4 m mean dominant height isoline. This corresponds to an age of 13 yr for this site quality and represents the target PCT age. The yield-density isolines indicated that the stands were slightly above the 4 cm quadratic mean diameter isoline and the 0.1 relative density isoline, at the time of the PCT treatment. The corresponding interpolated mean volume, density and basal area values were 2.9 dm^3, 2707 stems/ha and 4.0 m^2/ha, respectively. Similarly, the size-density trajectories at the time of the CT treatment were slightly above the 12 m mean dominant height isoline, 14 cm quadratic mean diameter isoline, 0.5 relative density isoline, and the 40% live crown ratio isoline. The corresponding interpolated mean volume, density and basal area values were 77.9 dm^3, 1604 stems/ha and 25.4 m2/ha, respectively. Accordingly, the stands would be candidates for CT treatments based on the guidelines given by McKinnon et al. (2006): CT candidate stands must have been previously managed in terms of density control treatments (e.g., IE with PCT), have a pretreatment basal area of greater than 25 m^2/ha, a mean live crown ratio greater than 35%, and where density-dependent mortality within the merchantable size classes is imminent. In case of the PCT stands, this last requirement was projected to occur at an age of 31 yr.

The mean dominant height at rotation age was 17.2 m for all 3 plantations. Respectively, for Regimes 1, 2 and 3, the rotational values for mean live crown ratio were 33, 34 and 38% and cumulative merchantable volume were 284, 240 and 211 m^3/ha. The CT treatment consisting of removing 35% (8.8 m^2/ha) of basal area at age 30 resulted in a mid- rotation harvest of approximately 39 m^3/ha of merchantable volume. Density-dependent mortality rates within the merchantable size classes of the CT stand was considerably lower than that within both the control and PCT stand during the post-CT period (c.f., 204 stems/ha within the PCT+CT stand versus 710 and 389 stems/ha within the control and PCT stands, respectively, over the 20 yr period). Although, relative to the control and PCT stand, the CT treatment resulted in larger but fewer trees of slightly inferior quality at rotation, the dual treatment did extended period of optimal site occupancy and substantially increased the economic worth of the stand at rotation. Relative to the control stand, the number of large standing snags (trees/ha) at rotation was approximately 39% and 73% less in the PCT and PCT+CT treated stands, respectively.

In summary, relative to the unthinned plantation, the thinning treatments resulted in (1) lower overall productivity in terms of merchantable volume (16 and 26% less for the PCT and PCT+CT plantations, respectively), and biomass and carbon production (8 and 11% less for the PCT and PCT+CT plantations, respectively), (2) extended the time to operability status by 6 and 12 yr for the PCT and PCT+CT plantations, respectively, (3) larger (mean volume) but fewer trees at rotation, (4) increased economic efficiency (36 and 54% less for the PCT and PCT+CT plantations, respectively), and (5) increased durations of optimal site occupancy (8 and 36% more for the PCT and PCT+CT plantations, respectively). With respect to the single core objective of increasing the production of high-value end-products through thinning, the results were not fully supportive. For the 2 mill configurations assessed, the thinned plantations produced lower volumes of chip (13 and 18% less for the PCT and PCT+CT plantations, respectively) and dimensional lumber products (9 and 20% less for the PCT and PCT+CT plantations, respectively). The removal of the merchantable-sized trees during the CT contributed to the decline in sawlogs and associated dimensional lumber volumes at rotation. In terms of product values, the thinned stands produced generally lower monetary values due to the decreased end-product volumes. However the differences were not large and in some cases were nil (c.f., product valves for the RLM configuration for the thinned versus control plantation (Table 4)). The largest benefit from thinning was in terms of an increase in economic efficiency as inferred from the ratio of land expectation values between the control and the treated plantations (Table 5). The lower product degrade values employed and the assumed lower variable costs arising from a more uniform piece-size distribution, largely contributed to this positive economic result.

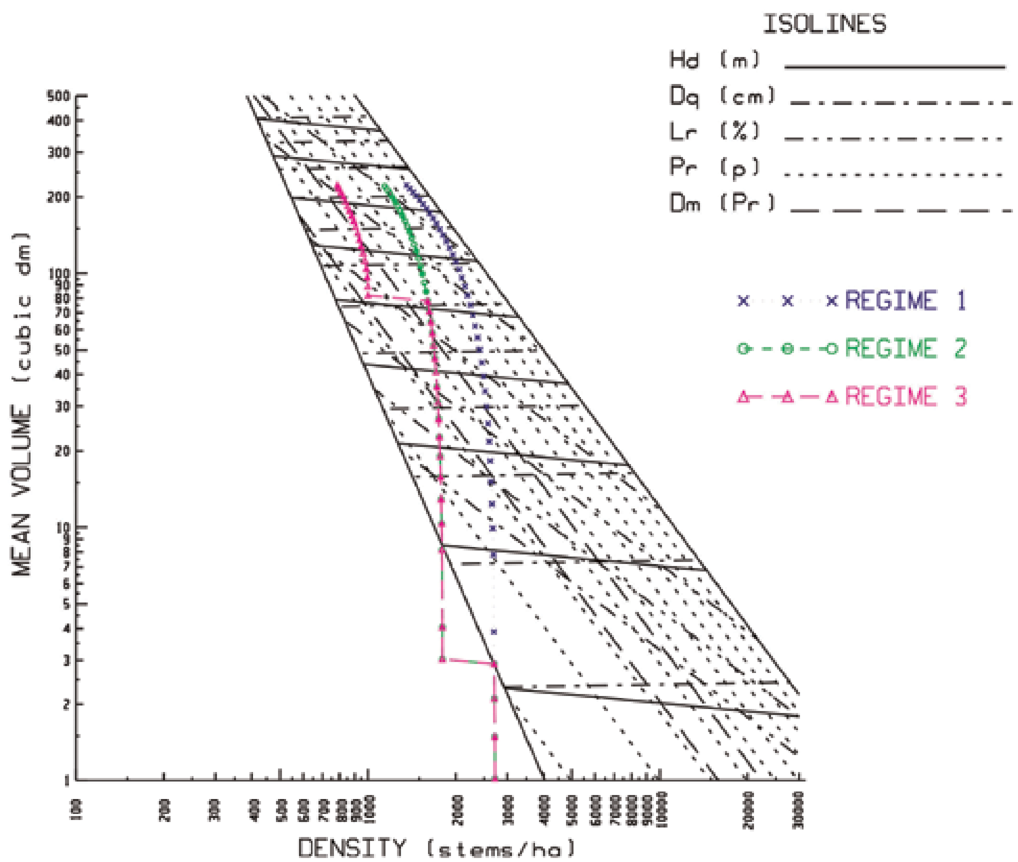

Fig. 3. Dynamic SDMD for genetically enhanced upland black spruce plantations managed under an elite silvicultural intensity. Graphically illustrating: (1) isolines for mean dominant height (Hd; 4-20 m by 2 m intervals), quadratic mean diameter (Dq; 4-26 cm by 2 cm intervals), mean live crown ratio (Lr; 35, 40, 50,…, 80%), relative density index (Pr; 0.1-1.0 by 0.1 intervals); (2) self-thinning line at a Pr = 1.0 and initial crown closure line (lower solid diagonal line); (3) lower and upper Pr values delineating the optimal density management window (Dm; 0.32 ≤ Pr ≤ 0.45); and (4) expected 50 year size-density trajectories with 1 year intervals denoted for 3 user-specified density management regimes for plantations situated on a good site quality (site index = 18).

Attribute (unit)	Regime 1 - Control	Regime 2 - PCT (thinning yields)	Regime 3 – PCT+CT (thinning yields)
Mean dominant height (m)	17.2	17.2	17.2
Quadratic mean diameter (cm)	21	21	21
Basal area (m^2/ha)	46	38 (1)	27 (1,9)
Mean volume per tree (dm^3)	222	227	237
Total volume (m^3/ha)	302	255 (2)	183 (2,43)
Total merchantable volume (m^3/ha)	284	240 (0)	173 (0,39)
Density (stems/ha)	1358	1120 (907)	773 (907,604)
Relative density index (%/100)	0.89	0.74	0.53
Mean live crown ratio (%)	33	34	38
Number of pulplogs (logs/ha)	1982	1667 (0)	1203 (0,464)
Number of sawlogs (logs/ha)	909	824 (0)	643 (0,0)
Residual log tip volume (m^3/ha)	42	35 (0)	25 (0,9)
Bark biomass (t/ha)	18	17 (0)	14 (0,3)
Stem biomass (t/ha)	164	144 (1)	110 (1,21)
Branch biomass (t/ha)	9	8 (1)	8 (1,3)
Foliage biomass (t/ha)	14	15 (2)	16 (2,5)
Total biomass (t/ha)	205	184 (5)	147 (5,31)
Bark carbon (t/ha)	9	8 (0)	7 (0,1)
Stem carbon (t/ha)	82	72 (1)	55 (1,10)
Branch carbon (t/ha)	4	4 (1)	4 (1,1)
Foliage carbon (t/ha)	7	8 (1)	8 (1,3)
Total carbon (t/ha)	103	92 (2)	74 (2,16)
Chip volume – SM (m^3/ha)	123	107 (0)	79 (0,22)
Lumber volume – SM (m^3/ha)	131	120 (0)	94 (0,11)
Chip volume – RLM (m^3/ha)	106	92 (0)	68 (0,20)
Lumber volume – RLM (m^3/ha)	146	133 (0)	105 (0,13)
Chip value – SM ($K/ha)	7	6 (0)	5 (0,1)
Lumber value – SM ($K/ha)	25	24 (0)	21 (0,2)
Total product value – SM ($K/ha)	32	30 (0)	26 (0,3)
Chip value– RLM ($K /ha)	4	4 (0)	3 (0,1)
Lumber value– RLM ($K /ha)	34	34 (0)	29 (0,5)
Total product value – RLM ($K/ha)	38	38 (0)	32 (0,5)
Land expectation value – SM ($K/ha)	3.7	5.5	6.4
Land expectation value - RLM - ($K/ha)	7.3	8.9	9.8

Table 4. Rotational yield estimates for upland black spruce plantations established at fixed IE levels subjected to PCT and CT treatments with genetic worth effects incorporated. Values in parenthesis denote yields derived from the thinning treatment(s) (ordered by time of treatment).

Index (unit)	Regime 1 - Control	Regime 2 - PCT	Regime 3 – PCT+CT
Mean annual volume increment (m^3/ha/yr)	5.7	4.8	4.2
Mean annual biomass increment (t/ha/yr)	4.1	3.8	3.7
Mean annual carbon increment (t/ha/yr)	2.1	1.9	1.8
Percentage of sawlogs produced (%)	31	33	28
Percentage of lumber volume recovered - SM (%)	52	53	51
Percentage of lumber volume recovered - RLM (%)	58	59	57
Relative land expectation value – SM (%)	-	49	74
Relative land expectation value – RLM (%)	-	22	34
Duration of optimal site occupancy (%)	12	20	48
Mean height/diameter ratio (m/m)	72	72	70
Mean wood density (g/cm^3)	0.48	0.49	0.50
Mean maximum branch diameter (cm)	2.65	2.69	2.72
Time to operability status (yr)	35	41	47
Time to initial crown closure (yr)	13	13	13
Age of crown re-closure post-PCT (yr)	-	16	16
Number of large standing snags (trees/ha)	350	214	93

Table 5. Stand-level performance indices for density-manipulated upland black spruce plantations established at fixed IE levels subjected to PCT and CT treatments with genetic worth effects incorporated.

3.3 Extension of the model to address non-timber objectives

Conservation of biological diversity is the cornerstone of sustainable forest management (OMNR, 2005). Although the broader issues of forest-level structural complexity and connectivity, and overall wildlife habitat requirements were assumed to have been addressed at the landscape level during the forest management planning process, density management treatments are expected to affect biodiversity at both the stand and forest levels (Thompson et al, 2003). Specifically, at the stand-level, biodiversity would decline as a direct consequence of the reduction in structural complexity arising from IE, PCT and CT treatments, principally through the (1) establishment of monocultures, application of herbicides and species-specific thinning treatments which would reduce species diversity, (2) regulation of intertree spacing which would result in a decrease in spatial complexity, and (3) truncation of the diameter distribution due to the thinning-from-below treatment protocol which would reduce the degree of horizontal and vertical structural heterogeneity. Employment of improved planting stock would also result in a reduction in genetic diversity. Lastly, the lowering of the intensity of resource competition through IE, PCT and CT would result in a reduction in the rate of self-thinning and hence a decrease in the production of abiotic components (e.g., snags and coarse woody debris).

However, the question remains as to the degree of impact that a reduction in biodiversity arising from density management would have. In an extensive literature review of previous biodiversity impact studies augmented by model projections, Thompson et al., (2003) concluded that (1) the presence of large sturdy and standing snags was most important to the vertebrate population in terms of providing nesting and denning site, (2) the quality of coarse woody debris (CWD) in terms of its decay stage and size were more important than quantity in relation to providing cover, feeding areas, and den sites for wildlife, (3) having a devise vertical structure combined with the presence of fruiting species within the understory was most conducive to the songbird populations, and (4) canopy cover was important for many vertebrate species in regards to avoiding avian predators. However, it is evident that some of these requirements are specific to a given wildlife species and hence are inversely related (c.f., (3) and (4)). Consequently, achieving an optimal stand structure which complies with all the wildlife habitat requirements would be largely illusive. Thus regulating stand densities in order to realize biodiversity objectives will likely involve various tradeoffs.

The modular-based SSDMM can be used to provide direct or indirect structural metrics that address biodiversity objectives. For example, at any point in a stand's development the model provides estimates of horizontal and vertical structure (e.g., diameter and height distributions). Similarly, the degree of canopy closure and crown heights can be inferred from crown closure line or calculated from the live crown ratio isoline as presented in the SDMD graphic (Figures 2 and 3). Estimates of the approximate number, age and size of CWD components produced during stand development can be derived from the model using the density and total volume estimates (Newton, 2006). Once the threshold values for these biodiversity-based structural metrics are explicitly quantified, they could be added to the suite of performance measures. Hence, the SSDMM could be used to determine if a specific crop plan complied with not only volumetric, end-product, or economic objectives, but also biodiversity goals.

For example, consider the plantation scenarios but now with a CWD requirement superimposed. Although CWD requirement has yet to be defined in terms of absolute volumes, sizes and decay classes, it is evident that a CT treatment will remove a substantial amount of the larger-sized trees that would have naturally incurred mortality during the later stage of the rotation. In fact, relative to the control stand, 506 fewer merchantable-sized trees per hectare experienced mortality during the post-CT period. Hence this differential in CWD production is of concern given the importance of CWD to maintaining biodiversity. However, one approach in overcoming this CWD deficit is to leave more of the CT trees on site at the time of the treatment. Specifically, by changing the merchantability thresholds of the trees to be removed, more of the stem can be left behind on the forest floor. The minimum diameter of CWD has been defined as 7.5 cm in Ontario (OMNR, 2010) and hence by decreasing log length and increasing the minimum threshold diameters for both sawlogs and pulplogs, the residual amount of stem volume left on the site will increase.

To demonstrate, the third scenario was re-run with the following modifications: (1) all log lengths were set to 2.59 m; (2) the minimum diameter for pulplogs was increased from 10 to 12 cm; and (3) the diameter defining the merchantable top was set to 7.5 cm. Effectively, this increases the residual stem tip volume left behind given that this volume is defined as the volume between the top of the upper most log removed and the top of the merchantable stem. Table 6 lists a subset of the resultant yields and performance metrics for this modified regime relative to the previous PCT+CT regime where the CWD requirement was not explicitly addressed (Tables 4 and 5).

This comparison reveals that the production of large volumes of CWD via CT did result in a decline in merchantable volume productivity, economic efficiency and operability status. However, the treatment was profitable given that the revenue generated from the approximately 9 m^3/ha of merchantable wood that was removed from the site exceeded the costs of acquiring and processing it. The CT treatment resulted in a substantial increase of relatively large CWD components (varying log lengths with diameters ranging from a minimum of 7.5 to a maximum of 12 cm). This CWD contribution should provide acceptable habitat to various wildlife species, particularly, the pine marten. Although not identical in terms of the volume of CWD produced, this scenario is similar to that proposed by Sturtevant et al. (1996) for pine marten habitat in western Newfoundland: i.e., providing old-growth stand structural attributes through the use of CT to generate downed CWD, which created denning and resting sites, subnivean access for cover, prey access, homeogeothermic regulation, and prey biomass (principally voles (genera *Microtus* and *Myodes*)), for the pine marten.

Index (unit)	Regime 3 – PCT+CT	Difference
Residual tip volume left on site at time of CT treatment (m^3/ha)	24	+15
Total merchantable volume removed from the site via CT (m^3/ha)	9	-30
Net Revenue arising from the CT treatment – SM ($K/ha)	0.1	-0.6
Net Revenue arising from the CT treatment – RLM ($K/ha)	0.4	-1.6
Relative land expectation value at rotation – SM (%)	74	-49
Relative land expectation value at rotation – RLM (%)	34	-38
Time to operability status (yr)	47	+2

Table 6. Subset of CT yield metrics and stand-level performance indices for upland black spruce plantations managed for the production of CWD.

3.4 Utility of SDMD-based decision-support models in forest management

SDMDs have an extensive history of development and use in forest management throughout many of the world's temperate and boreal forest regions. The SDMD developed by Ando (1962) for Japanese red pine (*Pinus densiflora* Siebold and Zucc.) in Japan was the first model to explicitly incorporate the reciprocal equations of the competition–density (C-D) and yield–density (Y-D) effect (Kira et al., 1953; Shinozaki & Kira, 1956) and the self-thinning rule (Yoda et al., 1963), into an integrated model framework. The reciprocal equation describes the relationship between mean tree size (C-D effect) or per unit area yield (Y-D effect) and density at specific stages of development within stands not incurring density-dependent mortality. The self-thinning rule describes the asymptotic relationship between mean tree size and density within stands undergoing density-dependent mortality. These core relationships were derived from empirical results and associated mathematical formulations arising from numerous plant competition experiments conducted during the 1950s and 1960s (e.g., Donald (1951), Kira et al. (1953), Hozumi et al., (1956), Shinozaki & Kira (1956), Holliday (1960), Yoda et al. (1963)). The SDMD is presented as a 2-dimensional bivariate graphic with density on the x-axis and mean volume on the y-axis upon which the reciprocal equations and self-thinning line are superimposed. Ando (1962) used the SDMD to design thinning schedules which would yield a specified quadratic mean diameter at rotation.

Following the successful introduction of the SDMD by Ando in 1962, Tadaki (1963) developed a SDMD for Sugi stands (*Cryptomeria japonica* D. Don.) in Japan and extended the utility of the model by illustrating how the reciprocal equation of the C-D effect could be used to estimate thinning yields. Later in 1968, Ando (1968) introduced a new set of SDMDs for Japanese red pine, Sugi, Hinoki cypress (*Chamaecyparis obtuse* (Siebold and Zucc.) Endl.) and Japanese larch (*Larix leptolepis* (Siebold and Zucc.) Gord.) stands in Japan. Using these new models, Ando demonstrated how they could be used as a decision-support tool in terms of evaluating the potential yield outcomes to various thinning treatments. In order to extend the applicability of the mean size – density relationship represented by the reciprocal equation of the C-D effect to stands incurring density-dependent mortality, Aiba (1975a,b) modified the Ando (1968) SDMD model for Sugi stands by replacing the reciprocal equation of the C-D effect with an empirical-based function where mean volume was expressed as function of both density and diameter.

Acknowledging the utility of the SDMD in forest management and silviculture decision-making, Drew and Flewelling (1979) introduced SDMDs to the forest management community in the Pacific Northwest through the development of a SDMD for coastal Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco.) stands. Since their introduction to the English-based forest science literature, numerous diagrams have been developed and utilized in stand-level management planning. These included SDMDs for Japanese red pine in Japan (Ando, 1962, 1968) and South Korea (Kim et al., 1987), Monterey pine (*Pinus radiata* D. Don.) in New Zealand (Drew & Flewelling, 1977) and Spain (Castedo-Dorado et al., 2009), Douglas fir in Spain (López-Sánchez & Rodríguez-Soallerio, 2009), lodgepole pine (*Pinus contorta* var. latifolia Engelm.) in the western USA (McCarter & Long, 1986; Smith & Long, 1987) and the Pacific Northwest (Flewelling & Drew, 1985), slash pine (*Pinus elliottii* Engelm. var. elliottii) and loblolly pine (*Pinus taeda* L.) in the southern USA (Dean & Jokela (1992) and Dean & Baldwin (1993), respectively), black spruce in the eastern and central Canada (Newton & Weetman, 1993, 1994), teak (*Tectona grandis* L.) in India (Kumar et al., 1995), pedunculate oak (*Quercus robur* L.) in Spain (Anta & González, 2005), Scots pine (*Pinus sylvestris* L.) and Austrian black pine (*Pinus nigra* Arn.) in Bulgaria (Stankova & Shibuya, 2007), Merkus pine (*Pinus merkusii* Jungh. et de Vriese) plantations in Indonesia (Heriansyah et al., 2009), and *Eucalyptus globulus* and *Eucalyptus nitens* short rotation plantations in Southwestern Europe (Pérez-Cruzado et al., 2011).

Analytically, the development of SDMDs has been characterized by a sequence of continuous incremental advancements in which increasingly complex and innovative model variants have been proposed. Acknowledging the paradigm shift in management focus from volumetric yield maximization to end-product recovery and value maximization (e.g., Barbour and Kellogg, 1990; Emmett 2006), and realizing the limitations of traditional SDMDs in addressing these new management objectives, the structural SDMD was introduced (Newton et al., 2004, 2005). Specifically, the structural model incorporated a parameter prediction equation system for recovering diameter distributions within the SDMD model architecture. More recently, an expanded version of the structural model was developed in order to address stand-level volumetric, end-product, economic and ecological objectives. To date, modular-based SSDMMs has been developed for jack pine (*Pinus banksiana* Lamb.) (natural-origin stands and plantations; Newton, 2009), black spruce and jack pine mixtures (natural-origin stands; Newton, 2011), upland black spruce (natural-origin stands and plantations; Newton, 2012a), and lowland black spruce (natural-origin stands; Newton, 2012b). These models were calibrated using extensive measurement data sets derived from hundreds of permanent and temporary sample plots situated throughout the central portion of the Canadian Boreal Forest Region. Consequently, the model and associated software suite (Croplanner) represents an operational and enterprise ready decision-support tool.

Essentially, these modular-based SSDMMs retain the ecological and empirical foundation of the original SDMD models, but in addition, incorporate estimation modules for predicting diameter, height, biomass, carbon, log, end-products and associated value distributions, and fibre quality attributes, at any point during a stand's development. The model allows managers to predict the consequences of a given crop plan in terms of realizing specified volumetric, end-product, economic or ecological objectives. In terms of its ability to forecast productivity, end-product and economic, the consequences of various density management treatments, the modular-based SSDMMs share a number of similarities to some of the existing stand-level density management decision-support models. Among others, these include SYLVER (Di Lucca, 1999) which was calibrated for Douglas fir and other coniferous species for use in western Canada, SILVA which was developed for Norway spruce (*Picea abies* (L.) Karst.) and other conifers and deciduous species for use in central Europe (Pretzsch et al., 2002), and MOTTI (Hynynen et al., 2005) which was developed for Scots pine and other conifers for use in Finland.

The SSDMM model architecture in which yield-density and allometric relationships provide the quantitative linkage among the component modules is readily adaptable in addressing new and evolving forest management objectives, as exemplified in the examples considered in this study. Given the large number of existing SDMDs combined with the transformative shift in management focus from volumetric yield maximization to product diversification, suggests that the modular-based SSDMM platform may have wide applicability in resource management.

4. Conclusion

The objectives of this study were to describe an enhanced stand-level decision-support model for managing upland black spruce stand-types, and demonstrate its operational utility in evaluating complex density management regimes involving IE, PCT and CT treatments. The traditional SDMD modeling approach along with its embedded ecological foundation is retained within the modular-based SSDMM structure. For a given density management regime, site quality, and cost profile, the model provides a broad array of yield metrics. These include indices of (1) overall productivity (mean annual volume, biomass and carbon increments), (2) volumetric yields (total and merchantable volumes per unit area), (3) log-product distributions (number of pulp and saw logs), (4) biomass production and carbon sequestration outcomes (oven-dried masses of above-ground components and associated carbon equivalents), (5) recoverable end-products and associated monetary values (volume and economic value of recovered chip and dimension lumber products) by sawmill-type (stud and randomized length), (6) economic efficiency (land expectation value), (7) duration of optimal site occupancy, (8) structural stability, (9) fibre attributes (wood density and branch diameter), and (10) operability status.

The utility of the model was exemplified by contrasting operationally relevant crop plans using a broad array of performance metrics. Specifically, the likelihood of (1) realizing an early operability objective via the use of PCT treatments within density-stressed natural-origin stand-types, and (2) enhancing end-product value through the use of PCT and CT within plantations, was evaluated. As demonstrated through these simulations, this ecologically-based model enables forest practitioners to rank alternative crop plans in order to select the most applicable one for a given objective. Additionally, the model provides annual and rotational estimates of volumetric, biomass and carbon yields, log distributions, recoverable products and monetary values, and fibre attributes, at both the diameter-class and stand levels. Although the results of these simulations are largely dependent on the input parameter settings (e.g., treatments (establishment densities, thinning treatments, site classes, rotation ages, product degrade values, variable and fixed cost profiles), the results readily illustrates the potential utility of the model in sustainable forest management.

The importance of the model in managing forest resources for the production high value solid wood products, bio-energy feed stocks, carbon credits, and ecosystem services including biodiversity, is explicitly acknowledged in the model's structure and output. Consequently, the model should be of utility as forest managers migrate to a value-added management proposition and attempt to address diverse objectives under varying constraints.

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Sustainable forest management (SFM) is not a new concept. However, its popularity has increased in the last few decades because of public concern about the dramatic decrease in forest resources. The implementation of SFM is generally achieved using criteria and indicators (C&I) and several countries have established their own sets of C&I. This book summarises some of the recent research carried out to test the current indicators, to search for new indicators and to develop new decision-making tools. The book collects original research studies on carbon and forest resources, forest health, biodiversity and productive, protective and socioeconomic functions. These studies should shed light on the current research carried out to provide forest managers with useful tools for choosing between different management strategies or improving indicators of SFM.

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