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Editors

Landscape Ecology in Forest Management and Conservation

Challenges and Solutions for Global Change

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With 73 figures

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Foreword

Like many others, my first exposure to the science of landscape ecology was from the book entitled *Landscape Ecology* published by Richard Forman and Michel Godron in 1986. For me, this was a new and exciting way for looking at the world in which we live. It was obvious to me after reading this book that the science of landscape ecology had much to offer natural resources managers. But it is also important to recognize that a "landscape perspective" has been around for a long time in a variety of sources and in a variety of places. One example is a book published in 1962 by Paul B. Sears, an early ecologist in the United States, entitled *The Living Landscape*. In this book written for a general audience, Sears described with great elegance why a "landscape perspective" is relevant (page 162):

"Compared to the noblest work of human genius, the landscape about us offers endless variety of interest and challenge. It is more than something to look at, it is something to comprehend and interpret. We are inseparably a part of it, and it is equally a part of us. Our destinies are linked, and while Nature will assuredly have the final judgment, modern man has the power to determine whether it will be thumbs up or down."

Aside from the gender bias that was common to that period, modern humanity indeed will be making important choices that will profoundly affect our children and many subsequent generations. Those choices should be predicated on the best available scientific knowledge. The current book edited by Li, Lafortezza, and Chen is another valuable contribution to comprehending and interpreting forested landscapes. It represents the latest work resulting from the bi-annual meetings sponsored by the IUFRO Landscape Ecology Working Party (08.01.02). The strength of this book is in the fact that it reflects the experience and knowledge gained by scientists in 15 different countries. It also provides a rich source of international literature.

It would be naive, however, to think that all we need to cure our challenging environmental and human problems is to do good science. Humanity has to recognize what Sears stated so well in his book – "We are inseparably a part of it, and it is equally a part of us." Until this linkage is clearly established in the minds of humanity, our future is uncertain. Perusing the current book suggests that both the science of landscape ecology and its application have come a long way. This book is worthy of a place on our bookshelves and it should not be collecting dust. But we need more. We need to recognize that our destiny is inexplicitly linked to that of those landscapes in which we live, work, play, raise families, and, above all, depend on for our very existence.

Thomas R. Crow
Fort Collins, Colorado, USA

Preface

Landscape ecology, as an independent research field, has been developed rapidly over the past three decades, largely due to the effective applications of theories from other ecological research fields in a spatially explicit manner that endorses the development of new concepts and methodologies; advanced methods and technology related to the geographical information systems (GIS) that integrates, synthesizes, and manipulates geo-referenced information in an efficient way; fast-developed information technology (IT) that provides necessary computing power in implementing the research at large spatial and temporal scales; increasing availability of spatial data sets, especially from the aero photography and remote sensing (RS) techniques; and the practical needs from the industries, regulatory agencies, and communities and societies. Nowadays, the theories and concepts of landscape ecology are relevant not only for natural systems including climatic and environmental systems, but also for anthropogenic systems including social systems, economic systems, and coupled natural and human systems. The behaviour of resulting complex systems is hardly handled efficiently, except for the mathematical modeling approach. Thus, landscape models have become test fields for exploring the logical consequences of the interactions among different theories and concepts and this, in turn, reinforced the fast development of landscape ecology.

Forest landscape ecology has reached a relatively mature stage for applications to real forest management challenges and issues. Many published books on landscape ecology have been focusing on addressing theoretical, conceptual, and methodological concerns, which provide a solid foundation for its applications to assist forestry policy development and forest management decision-making. This book attempts to focus on more specific issues and/or challenges in forest management and land-based multi-purpose management in the changing global environment.

Forests across the world provide living environments, services, and life necessities for human, wildlife, and other organisms to sustain their generations. However, the increasing footprint from human activities on unmanaged forest landscapes has altered normal ecosystem processes under natural conditions. Consequently, forest ecosystem dynamics are much more complicated to understand as a consequence of the interaction between human activities and natural processes. The impacts of global change have added more layers on top of coupled human-natural forest dynamics. The questions of how the global changes, especially climate change, could impact forest landscape dynamics and their management have become important challenges that forest managers, researchers, and professionals are facing. We consider these as both challenges and opportunities for landscape ecologists and practitioners to be able to address the question: how could landscape ecology research provide answers and solutions to forest management?

Forest management in a broad sense can have three main components: natural disturbance, habitat, and resource management, with each operation in any of the components can have an impact on the other components. The level of resource utilization is perhaps the only variable that humans can control to balance economic development and social, ecological, and conservational needs. Human's utilization of forest resources through harvest and land-use change has resulted in the reduced and fragmented forest lands and, in turn, the changes in wildlife habitat, biodiversity, productivity, old growth forests, environmental conservation, and other non-timber values including ecosystem goods and services. As a result, increasing attention has been paid to forest resource management with decreasing availability of forest lands and degrading quality of wood supplies. To contribute useful solutions to the forest management-related issues, landscape ecologists and researchers need to have a better understanding of the approaches, methods, procedures, and regulations involved in the forest management practice.

Understanding regional forest dynamics over space and time is crucial in forecasting the wood fibre supply. At the landscape scale, however, the critical issues are how the forest resource availability and habitat treatments could be influenced by natural and anthropogenic disturbances and their management. Natural disturbances such as fire, insect, disease, and wind can have profound impacts on forest dynamics as well as the quality of the resulting wood supply. Anthropogenic disturbances such as harvest can have an additive effect on forest landscapes and thus the sustainability and spatial distribution of forest resources. The mechanisms and processes of these disturbances need to be well understood for making informed management decisions.

Our expectation through this volume is to provide updated information on the approaches, procedures, and methods in practical forest management, which were different from those occurring decades ago. Research progresses in the three components of forest management and the development of decision support tools/systems driven by the spatially explicit landscape models toward solving the challenging issues in forest management.

This book consists of four parts: **Part 1** includes three chapters on landscape ecology and forest management, aiming at providing a conceptual framework and general background of contemporary forest management practices and procedures, challenges, and the research needs in a changing globe from a forestry and forest science perspective and a brief summary of what could be contributed from landscape ecology research toward solutions in forest management. **Part 2** is composed of five chapters on modeling disturbance and succession in forest landscapes, with a focus on the management of natural disturbances, especially forest fire and related research topics, through spatially explicit model development and applications. **Part 3** includes four chapters on emerging approaches in forest landscape conservation, which focus on the management and conservation of wildlife habitat and biodiversity and discuss how the zoning process can be improved through developing a forest network system as well as the forest landscape fragmentation-related issues. **Part 4** contains five chapters on practicing sustainable forest landscape management, which focus on the management of forest resources and related issues including applications of landscape and habitat suitability models, the effect of abandonment, the loss of biodiversity in South America, and decision support technology for achieving sustainable forest management.

The book is a collection of knowledge and experience from 15 different countries and provides complementary information to existing international literature in this field in terms of forest management planning and problem-solving on large-scale issues from a long-term perspective. In addition, this book is designed to serve as a reference book for providing materials for higher education purposes, in that more and more universities are offering landscape ecology-related courses through their undergraduate and graduate programs in natural resources, agricultural and rangeland, forestry, environmental sciences, etc.

The editors are happy to see a new trend and a number of senior scientists encouraged their students and technicians who bravely took the responsibility of first author and/or corresponding author. This is a powerful way of training highly qualified personnel for the future study and this will contribute to the rapid promotion of the IUFRO Landscape Ecology Working Group.

This book is the third publication in a series of contributions from the activities of the IUFRO Landscape Ecology Working Group (08.01.02). Most of the chapters of this book are authored by participants of the 2008 IUFRO Landscape Ecology Bi-Annual Conference held in Chengdu, China, hosted by the Chinese Academy of Forestry (CAF), on September 16–22, 2008, including some other interested experts who participated in this conference. The conference was the biggest in number of participants and countries in the history for this Working Group. The success of the conference largely relied on the enthusiastic participation and professional contribution as well as support from many organizations, including the USDA Forest Service, the NASA Land-Cover/Land-Use Change Program (LCLUC), the Institute of Applied Ecology of the Chinese Academy of Sciences, Fudan University, the Northern Global Change Program of USDA Forest Service, the University of Toledo, the CSIS of Michigan State University, the Higher Education Press, the *Journal of Plant Ecology*, the IUFRO Landscape Ecology Working Group, the CAF, the IUFRO Urban Forestry Working Group, the International Association of Landscape Ecology (IALE), the Sino-Ecologists Association Overseas (Sino-Eco), and the Sichuan Academy of Forestry (the local host). The success of this conference also depended on the strong logistic support provided by the ChuangWei Hong Company and the volunteers (Bixia Chen, Jessica Schaefer, Fei He, and others). We thank people of the Higher Education Press and Springer for their consistent support in considering this book.

We also appreciate very much the valuable and timely reviews from Devendra Amatya, João Azevedo, Huiquan Bi, Jan Bogaert, Kimberley Brosofske, Enrico Caprio, Mauro Centritto, Reinhart Ceulemar, Liding Chen, Robert Corry, Mark Ducey, Almo Farina, Alberto Gallardo, Eric Gustafson, Shongming Huang, Hong Jiang, Ranjeet John, Pekka Kauppi, Bob Keane, Habin Li, Zhenqing Li, Changhui Peng, Ajith Perara, Soung-R Ryu, Santiago Saura, Sari Saunders, Rob Scheller, Conghe Song, Henrich Spiecker, Ge Sun, R Talbot Trotter III, Chuankuan Wang, Mingliang Wang, Xiaohua Wei, Jian Yang, and Pat Zoner. Finally, this publication would not be available without the tireless drive and support of Dr. Bingxiang Li of the HEP.

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Part I
Landscape Ecology and Forest Management

Chapter 1 Managing Forest Landscapes under Global Change Scenarios

Chao Li*, Jianwei Liu, Raffaele Lafortezza and Jiquan Chen

Abstract

The increasing footprint from human activities on unmanaged forest landscapes has altered ecosystem processes under natural conditions and the climate change impact will add one more layer on top of the human-natural coupled forest ecosystem dynamics. How climate change could impact forest landscape dynamics has become one of the emerging challenges humans face today. This is also an opportunity to find out how landscape ecology research could contribute to addressing these issues. This chapter begins with the concepts, scope, and trends in forest management, followed by the linkages and interactions between different components of forest management. The level of resource utilization is probably a major variable that humans can regulate in achieving the goal of balanced decision-making to satisfy the needs from social, environmental, and economical concerns. The key factors in determining the level of resource utilization include forest growth and yield prediction, and uncertainties associated with natural disturbance regimes. With a good understanding of the above factors over space and time, the models in landscape ecology can contribute significantly to the climate change impact assessment and mitigation strategy development because climate change will influence both natural disturbance regimes and the growth rate of trees.

Keywords

Forest management, landscape ecology, climate change, landscape dynamics.

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1.1 Introduction

The increasing footprint from human activities on unmanaged forest landscapes has altered normal ecosystem processes under natural conditions over the past several decades. Consequently, forest ecosystems are more complex due to the interactions of the human activities coupled with natural processes (Liu et al. 2006, 2007). The impacts of global change include climate change, economic and market globalization, induced business consolidation, and industry transformation. Rapid technology development will add more layers on top of coupled human-natural forest ecosystems. The questions of how the global changes, especially climate change, could impact forest landscape dynamics and their management has become one of the important challenges that forest managers, researchers, and professionals face today (Crow 2008). They are both challenges and opportunities for landscape ecologists and practitioners as to how landscape ecology research could contribute to providing answers and solutions to these questions. This chapter aims at providing an overview of background information, challenges in forest management, and how landscape ecology research can help to solve the complex issues in forestry.

Forests across the world provide living environments and life necessities and services for humans, wildlife, and other organisms to sustain their populations. Forests are the primary producers of the earth's ecosystems and absorb solar energy from sunlight and carbon dioxide through photosynthesis to produce the essential mass (e.g., oxygen and glucose) and energy for life on earth. Forests also provide a wide variety of habitat types for different species of wildlife communities. For example, old growth forests contribute to biodiversity conservation and aesthetics (e.g., recreation and eco-tourism). Recent initiatives on the global carbon cycle further suggest that forests' carbon storage and sequestration capacity are crucial in the global carbon budget. From an economic perspective, forests are known as important resources for raw materials such as providing wood and pulp products, securing employment opportunities in the manufacturing industry, supplying biomass for bio-fuels or bio-energy, and bio-refinery development. Consequently, the challenge of balancing economic development and social, ecological, and environmental conservation has emerged as a new budding issue in forest management.

Increasing attention has been paid to forest resource management with decreasing availability of forest lands. Prior to modern industry, forests covered about half of the earth's surface, yet only less than one third of that area remains as forest cover (Food and Agriculture Organization 1993). Historical forest losses are largely due to the massive forest clearances for agricultural land in early cultures. In North America, timber harvest has been the primary reason due to the needs of production such as fibre, pulp and paper, and bio-energy. Forest resources management can play a vital role in balancing the wood fibre demands and harvest operations through the determination of the regional annual allowable cut (AAC) for forest resource utilization.

Improvement through sound management and planning can thus contribute significantly to achieving sustainable resource development and environment conservation.

Understanding regional forest dynamics over space and time is vital for forecasting the wood fibre supply. In forest succession research, the traditional Clementsian climax theory (Climents 1916) has been a “universal law” from a single equilibrium viewpoint that characterizes a regional mature forest status without significant natural disturbances. This theory has been supported by a lot of research including those of forest measurements, in which patterns of tree volume and biomass growth are usually described as having a mono-increasing sigmoid shape curve as a function of stand age (Avery and Burkhart 1994) and gap dynamics (West et al. 1981). However, increasing reports on age-related forest decline have also been documented (e.g., Gower et al. 1996; Ryan et al. 1997, 2004; Kirongo and Mason 2003). This phenomenon, coupled with new evidence on possible causes with process-based investigations, is underway to understand the biophysical constraints on forest development. The possible causes include increased respiration, reduced nutrient supply, increased allocation to non-woody components, and decreased gross primary production (GPP). The implication of alternative forest management, nevertheless, will essentially influence the values of our forests. At the landscape scale, however, the critical issues are how the forest resource supply, landscape fragmentation, wildlife habitat, and biodiversity can be harmonized by including other natural and anthropogenic disturbances for sustainable development.

Without natural disturbances, forests are assumed to grow following a sigmoid pattern over time. This has been extensively studied in the field of growth and yield and supported by a massive amount of literature, either theoretical or applicable. Current models predicting stand dynamics are largely region-dependent, based on site-specific relationships between volume (merchantable), diameter at breast height (DBH), tree height (H), tree age, stand density, site index (SI), taper factors of tree species, basal area, and mortality rate.

Natural disturbances such as fire, insect, disease, and wind can have a profound impact on forest dynamics. In Canada, for example, fires can have negative or undesirable effects on public health and safety, property, and natural resources from a socio-economic perspective, while they also play positive roles in the maintenance of forest ecosystem integrity, species diversity, and conservation of water and nutrients. According to the statistics of the Canadian Interagency Forest Fire Centre (CIFFC), the national average fire occurrence is about 8,000 times per year, with an average area burned of about 2.5 million ha per year. This is coupled with an annual suppression cost of $300-500 million. Here lightning fires represent 45% of all fires and 81% of total area burned. Additionally, 3% of fires are greater than 200 ha in size but represent 97% of burned area. Forest insects also play a major role in the decomposition of organic matter (i.e., on carbon cycle). While insect outbreaks are natural, normal, and initiate secondary successions and help to rejuvenate stands, the death or weakening of trees during an outbreak can cause significant economic losses. Finally, forest diseases, either biotic agents (or living organisms) or abiotic agents (or environmental factors), can make a negative impact on forests. For example, *Armillaria* root rot remains a major agent responsible for root diseases in Canada.

This chapter is aimed at providing an overview of background information, challenges in forest management, and how landscape ecology research can help to solve the complex issues in forestry. In the following section, we will first summarize the concepts, current research, and challenges in forest management. We argue that the level of utilization is the key variable that humans can modify toward multiple and optimal use of forest wood fibre products while socio-economic concerns can be properly addressed. Section 1.3 describes the challenges in global change that forest management must take into account. Section 1.4 presents a perspective that landscape ecology research can contribute toward potential solutions in forest management. This chapter will primarily focus on the professional aspect because it is probably the most suitable description of developing decision support systems (DSS) that links the landscape modeling approach to assist in the decision-making process in forest resource management. However, the social and environmental concerns should also be incorporated into these DSS. This chapter will end with a section of concluding remarks.

1.2 Forest management

In this section, we describe forest management definition and related research fields in forest science, components of forest management, and harvest planning in Canada and the management objectives.

1.2.1 Forest management and science

Forest management is one of the most commonly used terminologies in forestry and forest sciences. It can be defined across a spectrum of technological detail. At one end of the spectrum, forest management can be defined in a non-technical and very broad sense, as long as people (e.g., forest managers) think about the future of the forests. At the other end of the spectrum, it can be defined in a highly technical sense, requiring a wide range of expertise and skills including a good understanding of biological and ecological processes, knowledge of wildlife and their habitats, appreciation of forests' environments (e.g., rivers and wetlands), the long-range viewpoint of a planner, the patience of a labour negotiator, the skills of an administrator, the alertness, flexibility, and all-range resourcefulness of a successful business executive, and a genuine sense and feeling for the forest as an entity (Davis et al. 2002).

Forest management as a general terminology includes components of forest resource management, natural disturbance management (e.g., fire, insect, disease), wildlife habitat management, etc. However, the exact meanings and scopes of concern of this terminology can vary with different groups of people. Leuschner (1984), for example, described four different definitions. In a broad sense, forest management "integrates all of the biological, social, economic, and other factors that affect management decisions about the forest". Based on this definition, to make an informed decision requires knowing almost everything and a wide range of research activities will be necessary to achieve this goal. In a narrow sense, forest management deals primarily with silviculture and the biological management of the forests. This definition has been widely used in many forest management texts, especially in earlier times. From a forest industry perspective, forest management refers to any decision needed to operate a forest on a continuing basis. This definition in fact includes not only forest resource management, but also other considerations such as human resource management, mechanical engineering, business and market impact. From a professional perspective, forest management is the study and application of analytical techniques to aid in choosing management alternatives that contribute most to organizational objectives. This definition is basically a combination of economic and biological management.

This chapter will primarily focus on the professional aspect because it is probably the most suitable description of developing DSS that uses landscape modeling approach to assist in the decision-making process in forest resource management at broader spatial and temporal scales. However, the social and environmental concerns should also be incorporated into the DSS concept.

Other research fields related to forest resource management include forest mensuration, statistics, forest inventory, operational research (OR), and the applications of high technology such as remote sensing (RS), global position systems (GPS), and geographical information systems (GIS). Forest mensuration has been a traditional research field in forest sciences (Avery and Burkhart 1994) with a goal of characterizing the physical dimension of forest conditions primarily at individual tree or stand level through permanent sampling plot (PSP) and temporal sampling plot (TSP) techniques. The monitoring variables at different time periods are treated as important data sources for growth and yield modelling; the yield tables are used as basic information sources for estimating current and future wood fibre production.

Forest inventory is the main information of forest conditions available for forest management decision-making and in landscape scale research. Different types of forest inventory data at different scales exist in a variety of research and operation programs. For instance, operational forest inventory data is commonly used in the resource management planning processes of government forest management agencies and forest industries. In Canada, such inventory data is generated from a standard procedure including aerial photography and interpretation, field sampling design and plot data acquisition, and statistical data analysis to formulize forest growth equations and yield tables by stand types. Due to the complex procedure and large quantity of field sampling requirement, compilation of this type of inventory data relies heavily on resource availability. Depending upon the size of forests for management, this type of forest inventory usually takes a long time to complete.

The inventory data for the Forest Management License (FML) #1 of Manitoba (MB) in Canada is a good example showing the utilization of data collection and its application. The total size of this FML is 889,471 ha and about two thirds is considered as productive and potentially productive forestland. The forest inventory generation for the FML was based on the 1997 aerial photos and took five years and over $20 million to complete. During this period, 700 polygons were selected for field sampling that contains three plots for each polygon. The size of a polygon ranges from less than 1 ha to larger than 390 ha. In each plot (with a size of 100 m^2), DBH and H of all of the trees with a DBH >7.1 cm were measured, resulting in a total of about 26,000 trees for developing the forest inventory. Other variables have also been recorded in the field including site conditions, understory vegetation, and disturbance type and history. Though the investment and workload are overwhelming, this type of forest inventory provides thus far the best information on forest conditions.

While operational forest inventory is mainly at the regional level, national forest inventory can be obtained through aggregating these regional forest inventory at a coarser spatial resolution. For example, Canada's National Forest Inventory (CanFI) is at a spatial resolution of 100 km^2 for most provinces (Penner et al. 1997). The aggregated CanFI and its applications are most useful for national statistics, forest policy development, and reporting to various domestic and international organizations. Other forest inventory data also exists for specific purposes and is usually associated with research programs targeting specific objectives such as old growth condition, status of biodiversity, environment, and wildlife habitat.

With rapid IT development, locations of sampled trees and plots are being accurately determined using GPS and the forest inventory can be brought into GIS for various analyses (e.g., forest wood fibre production and analysis, harvest scheduling, Asia-Pacific Forestry Commission 1999). Furthermore, development in RS has made the forest inventory standardization possible and indirectly promotes the expansion of forest inventory to include remote areas and areas where currently no inventory is available. The operational forest inventory is generally presented at the stand or polygon and landscape scales for strategic and tactical harvest planning in large areas. Additionally, objective-oriented research programs have also scaled down using RS imageries with a high resolution such as the light detection and ranging (LiDAR) technology

(Wulder et al. 2000). This scaling down approach aims at providing more detailed and accurate information at the individual trees and stand scales (i.e., toward small spatial scaled forest management).

1.2.2 Components of forest management

Despite the diverse discussion on forest management, three main components can be identified as major contents: resources, natural disturbance, and habitat management. Resources management refers to harvest planning in both strategic and tactical senses. Natural disturbance management includes management of fire, insect, and disease. Wildlife habitat management includes old growth forests, biodiversity conservation, landscape aesthetics for recreation and ecotourism, and ecosystem goods and services.

All three foci of forest management are connected with each other and no single one can produce benefits in all aspects of ecosystem function and services. An operational action applied to and based on the management principle will affect the consequences of other actions (Table 1.1). The resource management, using harvest as a tool, will reduce wood fibre availability, biodiversity, fuel loading, living biomass, and carbon storage, increase landscape fragmentation, reduce/increase connectivity, and increase fuel breaks and dead biomass. The disturbance-based management would increase the wood fibre supply, biodiversity, old growth habitat, fuel load, living biomass, carbon storage, and decrease dead biomass, and produce mixed effects on landscape fragmentation and connectivity depending upon the type and size of the disturbances to be mimicked through management. The habitat-focused management, meanwhile, can increase the wood fibre supply, biodiversity, old growth habitat, fuel load, living biomass, and carbon storage, and has mixed

Table 1.1 Effects of different management on ecosystem function and service

Ecosystem function and services (examples)	Major components of forest management		
	Resources	Disturbances	Habitat
Wood fibre supply	−	+	−/+
Biodiversity	−	+	+
Old growth habitat	−	+	+
Fragmentation	+/−	IS*	−/IS
Connectivity	+	IS	+/IS
Fuel load	−	+	+
Fuel breaks	+	−	IS
Living biomass	−	+	+
Dead biomass	+	−	IS
Carbon storage	−	+	+
Carbon release	−	−	IS

*IS, insignificant.

effects on landscape fragmentation, connectivity, fuel breaks, dead biomass, and carbon fluxes.

One example showing the complex outcomes of three different management lies in harvesting activities and fire hazards (i.e., reducing fuel load and thus the probability of hazard fire), suggesting that disturbances management decision-making will be affected. A combination of reduced harvesting and enhanced natural disturbances management would have a positive impact on maintaining biodiversity and old growth forests, so that the habitat management decision-making will be affected. A combination of enhanced natural disturbances and habitat management could result in the increase of the future wood supply, living biomass, and carbon storage, thus the resources management decision-making will be affected.

Due to the interactions between the three major components, forest management as a whole can be seen having the structure of an interconnected web. Consequently, an important question raised here is: what is the key variable that humans can modify or control and that could have a profound influence on the overall dynamics of forest ecosystems, thus contributing to the balance among economic, social, ecological, and environmental development? The identification of this major variable would facilitate a coordinated and efficient forest management decision-making process.

We propose to treat resource utilization as the main variable in forest management, with a goal of sustainable development. On one hand, over harvest or if the level of resources utilization is too high, land owners may gain economically in short-term periods; however, society and environment could show concerns about unsustainable manner of management, lost of biodiversity, reduction of old growth forest area, and too much carbon being transferred to other pools. On the other hand, under harvest or if the level of resource utilization is too low, land owners may lose market opportunities, reduce forest renewal and carbon sequestration, and increase disturbance risks. However, old growth forests area can be increased and biodiversity can be better maintained. Reflected in the practical resources management, these are questions of a harvest planning process, AAC determination and harvest blocks allocation, and are related to the determination of the best utilization strategy being full, multiple, and optimal (Li 2009).

1.2.3 Harvest planning process in Canada

Forest harvest planning in Canada has experienced three major periods: a traditional management period before 1980, an integrated resource management period (1981-1995), and an ecosystem-based management period (1996-present). During the traditional forestry period, the priority goal was to maximize economic benefits through enhancing the human capacity of timber utilization. In this period, the capacity was generally limited by the technology on how to get the trees harvested efficiently and transport the timber to mills for processing. The market demand played a key role in determining harvesting methods and amounts. When this capacity had been developed to a point that exceeded sustained yield (i.e., the harvested stock is equal to the stock that can grow), sustainable forests became an important issue that brought up the concept of AAC in order to regulate the level of harvesting. Sustainable forest management later added additional constraints on protecting social and environmental benefits from the forests. Consequently, how the forest harvest planning process can cope with this trend and requirements has presented a serious challenge to contemporary forest resource managers and professionals.

In MB, Canada, for instance, a three-level (strategic, tactic, and operational) systematic analysis of the harvest planning process was conducted. At the strategic level, the goal is to determine the theoretical maximum sustainable harvest levels that can be constant on the land base over the planning horizon of 200 years. At this stage, the sustainable harvest level is calculated to meet the forest management policy requirements such as uninterrupted fibre supply from the land base and the operational constraints such as defined timber utilization standards, riparian zone protection, minimum harvest age, and forest regeneration delay.

For tactical planning, wood production is determined by the harvest blocks and harvest schedules that are derived from the strategic level planning. At this stage, the harvest blocks and harvest schedules are allocated/mapped-out under the spatial considerations such as flow fluctuation of wood supply, sizes of cut-blocks, spatial linear distance for grouping harvest-blocks, and green-up delay over the planning horizon of 20 years.

At the operational level, both five-year and annual plans are laid out according to the provincial guidelines (MB Conservation 2007). The operating conditions such as contingency logging area, minimizing road construction and access, maintaining core area and old forests, tree retention for wildlife in the harvest block, and forest renewal practices have been integrated into its annual plan (Tembec 2008).

The strategic and tactical analyses are formulated according to the provincial forest policies, which are operational guidelines and harvesting practices followed by the forest industries. Therefore, they are considered to be Provincial "Base Case" wood supply analyses. The Remsoft Spatial Planning System (RSPS) developed by Remsoft, Inc. (2006) was used to perform this analysis. The RSPS consists of two main software packages: "Woodstock" and "Stanley". Woodstock produces inventory projections, long-term harvest schedules, biodiversity and wildlife habitat evaluations, compliance certification standards, etc. Woodstock was used in the analysis to determine the optimal sustainable harvest level at the stragetic level in accordance to the stated objectives, actions, and constraints. Stanley is a simulation model that allocates the harvest blocks spatially at the tatical level according to the harvest schedule from the Woodstock. The objective of the Stanley simulation is to find the best fit or configuration of polygons (cut-blocks) to meet the Woodstock harvest schedule for the first 25 years. In doing so, the Stanley must take into consideration the setting of spatial constraints on harvest openings (cut-blocks), maximum block size, green-up delay, and spatial linear distance.

Protecting wildlife habitats and old growth forests from social and environmental demands can be maintained and satisfied through this planning process. Woodland caribou (*Rangifer tarandus caribou*), for example, have been classified as being at risk across Canada (Canadian Council of Forest Ministers 2009). The factors leading to the decline of the caribou population consist of forest resource utilization including harvesting and other activities associated with regional economic development such as roads, pipelines, and transmission corridors. These human activities generally result in the habitat loss, degradation, and fragmentation of the caribou. MB has been involved in a national woodland caribou monitoring study to determine the size and location of its populations (MB Conservation 2005). Radio collars and satellite and global positioning have been used to track the caribou's spatial distribution and to locate habitat core areas for a better landscape design and species conservation. Habitat Suitability Indices (HSI) were calculated by attributing scores of suitability to factors considered to be of importance to the wildlife species. The woodland caribou were assigned a score from 0 (unsuitable) to 1 (most suitable) based on the forest cover type and its age class. This was used in the analysis to evaluate their food and cover habitats on all available land areas, including closed and restricted areas, buffer areas, and protected areas that were removed from harvest consideration. For example, the reduction of the calculated AAC has been laid out for the FML #1 of MB, which had a reduction of 1.1% in softwood and 1.5% in hardwood through tactical level planning and a further reduction of 2% in softwood and 5% in hardwood through wildlife habitat protection.

1.2.4 AAC determination and harvest blocks allocation

The AAC determination and harvest blocks allocation are critical components of a strategic and tactical forest harvest planning process. The AAC determination is generally the result of a widely applied wood supply analysis. Allowable cut is the amount of timber considered available for cutting during a specified time period — usually one year. It is the amount of timber that the forest manager would like to have cut and thus is a target or guideline the manager attempts to "reach" (Leuschner 1984). The calculation of the AAC can be based on either area control or volume control.

After the method of area control, equal areas or areas should be cut annually or periodically. This requires cutting the same number of ha each year, in the simplest case. Therefore,

$$AAC_{Area} = A_{Total}/R_{Harvest} \tag{1.1}$$

where AAC_{Area} is the annual allowable area cut, A_{Total} is the total area of forests under management, and $R_{Harvest}$ is the designed harvest rotation. The volume of the AAC_{Area} can be estimated by looking up the appropriate yield table and multiplying by the number of hectares. The AAC_{Area} estimation becomes more complex if the hectares in the forest have different productivity levels, because the cuts of volume could fluctuate significantly in different years and create problems in even wood flow for a manufacturing plant or an even cash flow as a management objective. Consequently, in the practical forest management, the area control method generally needs to be modified for equal productivity by using the mean yield per ha, which is simply the mean weighed by the number of ha in each site class or

$$\bar{Y} = \left(\sum_i Y_i A_i\right) \Big/ \sum_i A_i \tag{1.2}$$

where $\bar{Y}$ is the mean yield per ha for the forest, Y_i is the yield per ha in the ith site class, and A_i is the number of ha in the ith site class. The area control method is easy to understand and calculate. With a specific rule of harvest such as "harvest the oldest stand first", the area to be harvested can be readily identified. However, large fluctuations in harvested volume using AAC_{Area} may cause serious problems from a commercial viewpoint. Therefore, the area control method must be combined with some type of volume control method when applied to unevenly aged stands.

After the method of volume control, annual or periodical cuts should have equal volumes. The calculation can be based on one of several formulas and this volume is then cut each year or in a period of time. The formulas include the Hundeshagen's Formula, the von Mantel's Formula, the Meyer's Amortization Formula, the Austrian Formula, and the Hanzlik Formula.

One of the main advantages of volume control is that some estimates can be made with very few data. For example, the von Mantel's Formula needs only an estimate of total growing stock (that can be made from an extensive timber cruise) and rotation age. This estimation technique can be applied as a rough first approximation or when better data is simply unavailable. It can also be a useful overall guide and first step toward regulation. However, the formulas requiring little data can be imprecise and inaccurate. Therefore, a combination of area and volume control methods is usually applied in practical forest management.

The AAC determination has nowadays become a standard and relatively mature method with the help of some commercial software packages such as the Remsoft Spatial Planning System (Remsoft, Inc. 2006) and Patchworks (Spatial Planning Systems 2009). When the calculation method was simplified (i.e., based on an ideal forest condition that allows trees to grow without any natural disturbance event consideration), the results might generate an overestimated AAC. This has raised serious concerns among forest managers and professional planners in Canada because of biased (often overestimated) AAC, which could lead to the overharvest of existing forests and cause problems in sustainable resource development. In practice, the AAC will be re-calculated when a catastrophic disturbance event causing larger than 10% of land bases is being altered. Li et al. (2005) investigated the issue of whether fire regimes could have a significant impact on the AAC determination. They simulated spatial forest dynamics under two scenarios of fire regimes with and without fire management and found that the AAC under a fire regime influence could be significantly lower than that under an ideal forest condition (i.e., no fire disturbance at all). Based on the simulation results, Li et al. (2005) suggested that the goal of fire management in the study area was controlling the size of annual burned area to be no larger than 1,200 ha. Other considerations in harvest planning include wildlife habitat protection, biodiversity and old growth forests conservation, etc.

Having determined the AAC for a region, the issue then becomes how to allocate the AAC spatially within the region. Forest management agencies could have a set of harvest rules such as a minimal age of tree or stand that can be cut, and spatial adjacencies of cutting blocks. In most available commercial software packages, this was done by a random selection and combination from all eligible stands for harvest, which satisfies the harvest rules. While the advantage of this approach is its flexibility in providing a large number of choices to forest managers, the disadvantage is that no optimal solution can be identified. From a perspective of science and technology, this is a question of spatial optimization and some solutions have been documented in the literature, which can be incorporated into the commercial software packages as well as the research models in landscape ecology.

1.2.5 Full, multiple, and optimal wood fibre utilization

The forest wood fibre utilization strategy is essential in determining the efficiency of using the available wood fibre supply. The full wood fibre utilization is referred to when not only the best quality wood fibre is used, but all quality classes of wood fibre are used. To implement this strategy, forest managers need to know the spatial distributions of different quality classes of wood fibre supply in their regions. This information may or may not exist in current forest inventory. However, it could be derived from existing forest inventory through the relationships between wood quality classes and the variables such as tree species composition, site index, and other physical site conditions. Furthermore, the spatiotemporal dynamics of the distributions of different quality classes can be affected by the changes in forest succession stages and natural disturbances, by which wood fibre supply in higher quality classes can be changed to lower quality classes or can even lose entire value used in given end products. This has presented new challenges for forest researchers in terms of providing methods and tools to predict different quality classes of the wood fibre supply over space and time (Li 2009).

Modelling landscape disturbances can be refined to provide the spatiotemporal information on different quality classes of wood fibre production. For instance, a number of landscape fire regime models (Keane et al. 2004) are able to simulate the impact of fire disturbances in terms of fire frequency, fire size distribution, and fire severity over space and time. These models are by far the most advanced landscape dynamic models that can be refined to meet the informational needs from forest resource managers.

The multiple wood fibre utilization is when the use is not limited to wood and pulp and paper products, the wood fibre is also used for the biomass production of bio-fuels and bio-refinery, for the potential carbon offset credit, and for other non-timber values. To realize the multiple utilizations, all possible values from forests need to be taken into account that not only limit the forest products, in which economic values can be estimated in a relatively straightforward manner. Biomass production for bio-fuels, bio-energy, and bio-refinery has been a highly emphasized use of lower quality classes of wood fibre. For example, salvage harvest of the trees infested by mountain pine beetles has been considered for bio-energy usage. Forests being used for potential carbon offset credit in the international and domestic trading systems have also been valued considering the soaring unit price. Other non-timber values of forests include wildlife habitats in the hunting and gaming industry, landscape aesthetics for the recreation and ecotourism industry, biodiversity for human needs of food, medicine, shelter, and other consumption products, and for the functioning of ecosystems and critical ecosystem processes that moderate climate, govern nutrient cycles and soil conservation, control pests and diseases, and degrade wastes and pollutants, ecosystem goods and services such as nutrients and hydrology that provide essential necessities for forest health and integrity, either from ecological valuation methods by a cost of production approach, or from economic valuation methods by the exchange value of ecosystem services, or from integrated dynamic approach that deals simultaneously with the above two in a balanced way (Winkler 2006).

A common “currency” is required for the integrated dynamic approach that balances the ecological and economic valuations and the economic market value in dollars can be one option of the common “currency”. With these valuations for multiple potential usages, a net benefit for each site can be estimated through the reduction of total costs associated with each potential usage of wood fibre following the marginal value concept in forestry economics (Pearse 1990):

$$M = R - C \tag{1.3}$$

where M is the marginal (or net) value, R is the revenue or value creation from a given end use of wood fibre, and C is the costs associated with the value creation.

Consequently, the specific end usage corresponding to the maximal M

value will be the best usage among all possible end uses of wood fibre.

The optimal wood fibre utilization is to match the right fibre to the right product at the right market time. This strategy emphasizes the match between the end products and their best fit of the wood fibre attributes and quality classes, as well as the timing of production for given end products. For example, using a high-quality class of wood fibre supply for a low-quality class of end products will contribute less in reaching the fullest potential of the wood fibre supply. An important point in the optimal wood fibre utilization is taking the market conditions into account. Well-known fluctuations in the market, sometimes dramatic, mean that the forest products' demand and price can display favourable or unfavourable conditions alternatively so that the production in a given type of forest product needs to be adjusted from time to time.

Ideally, this full, multiple, and optimal wood fibre utilization strategy can ensure that all quality classes of wood fibre are used most potentially and all possible value creation options are considered. To implement this strategy, some concepts and methods from other fields probably need to be introduced, such as value chain (analysis) and global value chain from the field of business administration as well as mathematical programming in the field of operations research.

1.2.6 Management objectives and future forest management

The best forest management decision might largely be determined by the management objectives, which are ultimately decided by the landowners and stakeholders of the forests. Across the landscape, patterns of land ownerships vary from country to country suggesting that management planning needs to be spatially adaptive. In Canada, most forest lands are publically owned and managed by provincial governments through forest management licenses to different forest industries. Consequently, governments can develop guidelines to regulate resource management planning, while the cooperation between forest industries and government jointly decide strategic harvest planning for the industries to implement. Close cooperation among these interested parties is the key for us to achieve our common objectives. Through the harvest planning process, collective decisions are made by the forestland owners and stakeholders with various social, economic, and environmental considerations.

The objectives of forest management may also change over time for a given region where different jurisdictions exist. Hence the best management solution for various objectives can vary significantly. For example, if the objective is more focused on old growth habitats aiming at biodiversity, the best solution would be a longer forest harvest rotation and a lowered AAC. However, if the objective is focused on carbon sequestration and a high mean annual increase (MAI) of forest wood fibre supply, the best solution would need to be a shorter forest harvest rotation and higher level of AAC (Li et al. 2008). Other management objectives may result in the best solutions between these two extremes. Nevertheless, all of the possible objectives of forest management are bound to satisfy an essential requirement of forest sustainability.

Forest sustainability is no doubt an integral part of forest management and policy but its exact meaning is still in discussion for a consensus. The scope and emphasis of forest management have evolved from changes in conceptual value of the forests into all ecosystem functions and services. For landscape research, the focus on forest dynamics has shifted from individual tree level to stand level and landscape level, from inventory projection to include natural disturbance impact, from non-spatial to spatial, from timber value only to include non-timber values, from single wood fibre use to multiple usages, from volume-based to value-based management, and from canopy tree only to include understory vegetation. These shifts are essential in the enhanced understanding of forest dynamics, to the changes in societal and people's conceptual focuses on forest value, and in global and domestic conditions in environment and business.

1.3 New challenges in a changing globe

Forest management has been experiencing tremendous challenges due to the unforeseen and sometimes unfavourable forest products market and changes in global finance and business networks. Many of the challenges are probably not new, but they have been further complicated by the emerged properties of forest ecosystems from increased human activities, climate change, and restructuring of global business networks. Under such changes, forest sectors need to justify their priority ranking of regional issues and concerns. Nevertheless, the following issues appeared essential in limiting the capability of forest managers, professionals, and practitioners in addressing the present challenges:

(i) Climate change impact and adaptation: Climate change has both positive and negative impacts on forest ecosystem functions and services, through two primary mechanisms. One is to directly affect tree growth rates and another is to alter natural disturbance regimes. There is a general lack of empirical evidence of how different tree species respond to changing climate, making our forecasting of future forest conditions difficult. More attention needs to be paid toward our knowledge based on tree responses to the changing climate.

(ii) Natural disturbance impact: Historical fire records are incomplete in many countries and regions and the understanding of the dynamics of natural disturbances remains sporadic. For example, physical science-based fire behaviour research results are being used in predicting fire growth in short-term dynamics. However, the fire behaviour research is still not adequately used for long-term fire management. Scenario-based fire regime models are not suitable for climate change-related research because the simulated fire dynamics do not respond to changes in climate. More attention needs to be paid to process-based fire regime models that can produce climate variable-sensitive results.

(iii) Forest inventory: The operational forest inventory is essentially based on the aero photo interpretation and the ground truth sampling and updated with timber cruise sampling data. The wood volume-based inventory can be projected into the future according to the growth and yield equations that represent average growth patterns of different tree species over their lifespan. For implementing the full, multiple, and optimal wood fibre utilization strategy to meet demands from social, economic, and environmental considerations, other information will be needed such as valuation of available wood fibre supply based on various end forest products, non-timber values, and fibre attributes. Landscape ecologists are in a unique, favourable position to be able to contribute to this by synthesizing the relationships between variables of site conditions and fibre attributes from the research results of wood sciences.

(iv) Scale issue: Ecosystems are hierarchically structured from global to continental, national, regional, provincial, landscape, stand, tree, fibre, cell and genotype levels. Forest dynamics are often referred to as the tree level, although the underlying processes are linked at all hierarchical levels (Inouye 1999; Millennium Ecosystem Assessment, 2003). The information flow upwards and downwards constitutes the scale issue in ecological research. Our primary concerns in forest resource management are from the tree to stand to landscape levels. At each scale, the variables used to characterize structural and functional properties can be different. Information aggregation or scaling-up from tree level to stand level and to landscape might be straightforward if every tree within the stand and landscape are measured. However, this is neither economically possible nor ecologically desirable in the real world. It is well known in the hierarchy theory that lower-level processes are constrained by processes operating at higher levels (Allen and Starr 1988). Therefore, a sampling design must be applied by including a higher-level summary based on lower-level detail.

(v) Landscape fragmentation and loss of old growth forests: It is widely recognized that landscape structure has been altered significantly by human activities responsible for the landscape fragmentation and loss of old growth habitats. The old growth forests are generally accompanied by rich biodiversity, high landscape aesthetic values, rich ecosystem goods and services, and quality habitats for rare wildlife. The challenge to forest managers is to incorporate these knowledge bases into their systems.

1.4 Landscape ecology contributions

Landscape ecology is almost at the exact right position for the solutions to challenges forest management is facing nowadays. This is due to: (i) combinations using computer modelling and spatially explicit research approach using a GIS platform, (ii) assembling available information according to the geo-references of different information sources such as RS images, aero photos, plot data, and tree measurement, (iii) applications of all useful methods from different fields, (iv) reconstruction of natural and anthropogenic influenced forest ecosystem dynamics through process-based simulation models, and (v) producing useful simulated data for forest managers and specialists of other research fields for further analysis.

(i) Developing landscape models for a better understanding of forest dynamics: Current and future forest management decision-making needs spatially explicit forest dynamics information about where, when, and what wood fibre would be available, requiring spatially explicit models. Landscape models are good tools to integrate and synthesize available information with georeferences, primarily because traditional experimental designs (e.g., manipulations) cannot be applied at broader spatial and temporal scales. However, existing models need to be refined to include more detailed forest growth descriptions including how they respond to the changes in climate and environment variables, natural disturbances and their interactions, and how forest management options might affect all of the outcomes.

(ii) Developing models and tools to support decision-making: Landscape models simulating or predicting forest conditions under given management options or operations can serve as the core engine of DSS. The reliability and accuracy of model simulation results will have a profound impact on the DSS performance. A DSS usually consists of models of forest dynamics, user interfaces to organize input and output data and how the management operations are to be enforced on the forest dynamics, analytical procedures of simulation results, and visualizations of final outputs from different aspects. The landscape ecologists can play a significant role in this aspect.

(iii) Working together: Effective and successful forest management cannot be achieved in its own profession and needs close collaborations with researchers in other fields such as management sciences. Strategic thinking emphasizes a long-term perspective as business networking, applications of value chain (analysis) (see <http://www.quickmba.com/strategy/value-chain/>) and global value chain (see <http://www.globalvaluechains.org/concepts.html>) concepts, and local and global optimization. From a technical aspect, systems engineering, operations research, and mathematical programming had developed useful tools in this regard; however, more emphases should be placed on adding social, environmental, and market conditions to balanced decision-making. These models and tools will be needed to facilitate roundtable discussions among forest land owners and stakeholders.

1.5 Conclusion remarks

The interaction between natural forest ecosystems and human activities results in the complex dynamics we observe. The challenges forest managers and researchers are facing today are essentially to define and determine the best forest management operations and their appropriate levels that can benefit both current and future generations in terms of meeting social, economic, environmental, and managerial requirements.

To achieve this goal, the first requirement is a better understanding of forest dynamics over space and time under various conditions including global change. This understanding will enable forest managers and researchers to evaluate the wood fibre supply for potential usages, which, in turn, enables the determination of the full, multiple, and optimal regional wood fibre utilization strategy that can maximize the realization of the potential. Landscape ecology research has the potential to make these things happen or to be realized by working with experts and researchers from other research fields.

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Chapter 2 Landscape Ecology Contributions to Forestry and Forest Management in China: Progresses and Research Needs

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Abstract

This chapter presents an overview on historical and current forestry and forest management in China. Although China's natural forests had greatly reduced over the past several centuries due mainly to agricultural development, over-exploration and wars, there has been a sustained growth in total forest area and volume for several decades partly because of the implementation of several national key forestry programs aiming at biodiversity conservation and sustainable forestry development. China's forest resource today is still insufficient because of low quality and productivity, and inadequate forest management. The major problems of forest management in China include deficiency in linking forest management with end usage, inadequate forest health management, lack of integrated forest landscape management, and unbalanced consideration on economy over environment. Forest management must address increasing concerns on challenges and emerging global issues, of which climate change is identified as the most severe threat. To tackle the existing problems and cope with uncertainties in changing environmental conditions with climate change, landscape ecology can play a major role in facilitating sustainable forest management (SFM) by providing theories and management tools for forest restoration, biodiversity conservation, land and water resource management and forest landscape planning. Forest management practices that consider spatial heterogeneity, pattern-process, disturbance regime, scale and spatial-temporal context of forest landscapes beyond forest boundary are increasingly adopted by forest

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researchers and managers in China. However, more research is needed to enhance long-term forest ecosystem monitoring, develop cross-scale and multiple-purpose forest management guidelines, improve landscape decision support systems, and formulate integrated ecosystem management policies and practices so that forest landscape management can be adapted to climate change and landscape sustainability can be strengthened.

Keywords

Forest management, climate change, landscape ecology, forest conservation, forest landscape management, integrated ecosystem management, landscape decision support systems.

2.1 Introduction

Forest is a major component of terrestrial ecosystems and provides important ecosystem services such as ecological functions and wood and numerous other products that significantly contribute to human well-being. Forests, like many other types of vegetation, have faced great challenges of environmental changes, with human disturbances as a main driving force in the past centuries. Over the past 50 years, forests in China have experienced unprecedented rapid and extensive changes, due largely to rapidly growing demands for food, fresh water, timber, fiber, and fuel (Ma, 2005). Climate change, e.g., increasing frequency of extreme events of dry and hot periods, is expected to exert significant impacts on forests over the next 100 years or so (IPCC, 2007).

Climate change will cause geographical shifts in distributions of individual tree species and forest types by altering the spatial and temporal patterns of temperature and precipitation, two of the most fundamental factors to determine the distribution and productivity of trees and forests. These impacts are not limited to trees themselves, but also to the whole forest ecosystems and the associated biota. The changing climate will also affect the occurrence, timing, frequency, duration, extent, and intensity of disturbances such as fire, insect and disease, hurricane, drought, and wind storm, which shape forest ecosystems by influencing their composition, structure, and functional processes (Dale et al., 2001; Lynch, 2008). The forest sector needs to assess the short-term and long-term impacts of climate change on trees and forests, identify their adaptive potentials, and find ways to improve forest vitality and resilience to cope with global change. There is therefore a clear need to integrate adaptive strategies into current forest management, especially in fragile landscapes and forests under severe threats by both human activities and climate change.

Forest is a major stabilizing component of natural landscapes, protecting soil, water, and households, reducing hazard risks of floods and landslides, and mitigating global climate change by carbon sequestration. Forest ecosystem services depend critically on the area and spatial distribution of forests across landscapes. Many issues of global importance, such as climate change, land use and land cover change and water resource alteration, are of great relevance to forests; furthermore, the consequences of these factors to forests, forestry and human well-being manifest mainly at the landscape and regional scales. Seeking solutions and establishing scientific basis to tackle many of the forestry-related issues should therefore be based on enhanced knowledge of landscape ecology and associated application technologies in terms of GIS, remote sensing, and decision support systems (DSS).

Forest management in the context of global change has become more complex and cannot be approached only at stand level and by the forestry sector alone. Instead, forest management should be implemented with cross-sectoral cooperation at the landscape level. Embedded in the concept of forest landscape management is the recognition of interrelationships between land use, sustainability of landscape resources such as desired goods and services, and diversified adaptive options for different stakeholders to cope with global change and land use conflicts. Sustainable forest management (SFM), which is guided by explicit goals, executed by policies and protocols, and monitored and assessed by defined indicators and standards, has been widely advocated and become the mainstream of the world forestry development. In the last two decades, aided by the fast developing spatial information technology in terms of computer, GIS, and remote sensing, landscape ecology has been increasingly applied to SFM practices (Diaz and Apostol, 1992; Otto, 1996; Schlaepfer, 1997; Wilson and Baker, 1998). Faster and more extensive implementation of the SFM paradigm should greatly reduce anthropogenic impacts and minimize negative climate change threats.

The objectives of this chapter are two-fold: to provide an overview of forestry, forest management, and challenges from climate change in China, and to describe the contributions and future research needs of landscape ecology to achieve SFM. Section 2 focuses on historical and current status of China's forestry and forest management. In this section, we identify the past problems and illustrate the progress made in China's forestry over time, along with different policies and needs from various periods of national economic development. Section 3 describes the major challenges China faces today in forestry and forest management, especially those under the climate change conditions. Following a summary of landscape ecology research that has contributed to the solutions to these challenges in Section 4, a number of future research needs is listed in Section 5. Landscape ecology has a major role to play in coping with climate change and facilitating SFM by providing theories and management tools for forest restoration, biodiversity conservation, land and water resource management and forest landscape planning. Adaptive forest management through a participatory and cross-sectoral approach should be promoted to ensure landscape biodiversity, health and sustainability at multiple scales.

2.2 China's forestry and forest management

Over the past 50 years, forestry development in China have experienced unprecedentedly rapid and extensive changes more than in any comparable period of time in human history, largely to meet rapidly growing societal demands. Objective of forest management has been changing from primarily focused timber production to multiple forest goods and services. Forests are currently managed as an ecosystem as whole for providing timber and non-timber products, and regional economic development, employment, human-living environment amelioration, and cultural and spiritual services as well. There is a series of ongoing key forestry programs in China, which leads to fundamental changes in the social demands for forestry focusing on ecological improvement, ecological security and ecological culture.

2.2.1 History of forestry and its mission

Historically, China had rich forest resources and biodiversity due to its vast geographical areas and highly diversified environmental conditions, which sustained various forest ecosystems ranging from boreal forests in the north to tropical rain forests in the south. Forests have been providing a large quantity and varieties of material resources for China's social and economic development, as well as China's civilization. However, natural forests in China had been greatly reduced over time, particularly in the past several centuries, due to agricultural development, over-exploration, and years of wars. For example, China's forest coverage was estimated to have decreased from 64% in 2000 BC to 10% in 1949 (Ma, 1997; Fan et al., 2008). Since the founding of P. R. China in 1949, the forestry development has experienced a zigzag process characterized by three distinct phases of reduction, rehabilitation and development of forest resources.

Phase I: Focusing on timber utilization. From the 1950s to the end of the 1970s China's forestry focused primarily on timber utilization. This phase was guided by the traditional forestry concepts and exemplified by extensive exploitation of forest resources. In order to meet the needs of national economic development, the priorities of forestry were to secure supply of timber and rehabilitate the country's timber-production capacity from war-induced damages. Under those circumstances, forests were regarded primarily as economic resources, forestry was regarded as a key industry of the national economy, and forestry sector was regarded as an industrial sector. Forest management in this period centered on increasing timber production and emphasized enhancing human-aided stand regeneration after clear-cut.

Phase II: Balancing timber production and ecological improvement. From the end of the 1970s to the late 1990s, the rapid forestry development, especially the implementation of the Three-North Shelterbelt Development Program, ushered China's forestry into a phase with equal emphases on timber production and ecological improvement. It was not a simple coincidence that this phase overlapped closely the first two decades of China's reform and opening-up during which profound economic and social changes took place. While promoting timber production, China gradually intensified its efforts in protecting forest resources, conducted large-scale afforestation and greening campaigns, and initiated ecology-based forestry programs to control soil and water erosion, to protect and improve environmental conditions, and to increase forest resources. The strategic forestry objective was to enhance both forest ecosystems and forest industry systems at the same time. In this period, forest management was mainly conducted at the forest stand level with emphasis on stand productivity, biodiversity conservation, and water and soil protection.

Phase III: Emphasizing ecological improvement. From the late 1990s up till now China's forestry development has entered into a new period characterized by strong emphases on sustainable forestry development and ecological benefits. With rapid progress in economic reform and opening up of the country, the Chinese Government recognizes the importance of balancing the three dimensions of ecological, social and economic benefits in forestry development. In 2003 China released the Resolution on Accelerating Forestry Development, which defines the national strategy for forestry development that focuses on ecological improvement, ecological security and ecological culture. The ecological awareness of the whole society in China has been significantly enhanced, which leads to fundamental changes in the social demands for forestry. For example, a series of key forestry programs have been implemented, including the Natural Forest Protection Program (NFPP), the Conversion of Cropland to Forest Program (CCFP), the Sandification Control Program for Areas in the Vicinity of Beijing and Tianjin (SCP), the Key Shelterbelt Development Programs (SDP), the Wildlife Conservation and Nature Reserves Development Program, and the Forest Industrial Base Development Program (FIBDP). In this period, forest management is often conducted at landscape or region scales, highlighting the roles of forest in biodiversity conservation, carbon and hydrological cycling, and mitigation of global climate change. Landscape ecology and associated spatial technologies are increasingly used in forest management.

2.2.2 National programs of forestry initiatives

Since the Rio Summit in 1992, the Chinese Government has been improving the policies, regulations and laws on sustainable forest development, encouraging people from all walks of life to get involved in forest ecological improvement through the implementation of the national key forestry programs. The following is a brief description about the scope and status of each program.

The Natural Forest Protection Program (NFPP): The NFPP, started in 1998 and fully implemented in 2000, covers large areas of China: the upper reach of the Yangtze River (including the provinces of Yunnan, Sichuan, Guizhou, Chongqing, Hubei, and Tibet Autonomous Region), the upper and middle reaches of the Yellow River (including the provinces of Shanxi, Gansu, Qinghai, Ningxia Hui Autonomous Region, Inner Mongolia, Shaanxi, and Henan), the northeastern China (including Inner Mongolia, the provinces of Jilin and Heilong Jiang), and the Xinjiang Uygur Autonomous Region of western China. As a strong measure of the program, timber harvesting has been banned completely in the upper reaches of Yangtze River and Yellow River, and greatly reduced in the northeastern China and the Xinjiang Uygur Autonomous Region. The targets of afforestation are 9.06×10^6 ha and 3.67×10^6 ha for the two regions of Yangtze River and Yellow River, primarily by means of logging moratorium to allow for natural regeneration. Nevertheless, a new policy is envisioned that would allow for some forest management operations such as thinning and limited commercial logging when appropriate.

The Conversion of Cropland to Forest Program (CCFP): The CCFP is to deal with soil and water erosion on hilly areas by afforestation in these marginal agriculture lands. CCFP covers nearly 20 provinces. The program plans to restore 1.467×10^7 ha for forest management, of which 7.452×10^6 ha is in the Yangtze River tributaries and southern China and 7.125×10^6 ha in the Yellow River tributaries and northern China. The program also includes afforestation on 1.733×10^7 ha of barren mountains and lands suitable for forest vegetation, with 7.511×10^6 ha in the regions of the Yangtze River and southern China and 9.822×10^6 ha in the regions of the Yellow River and Beijing.

The Desertification Control Program (DCP): The DCP is to reduce the frequency and intensity of sandstorms for areas adjacent to Beijing and Tianjin. The implementation of SCP began in 2000 with a total planned area of 4.58×10^7 ha in 75 counties of Beijing, Tianjin, Hebei, Shanxi, and Inner Mongolia. By 2007, a total area of 6.694×10^6 ha had been treated as potential sources of airborne particles; a total area of 5.684×10^6 ha had been protected from grazing; the number of the ecological migrants who moved from the area severely affected by desertification to the other areas less affected by desertification was 116,000. During the period of 2001-2005, the coverage of forests and grasslands within the program area increased by 10-20.4%, while the dustfall declined by 15.8%. The large area of desertification control (ca. 960 million ha) should help to mitigate the impact of climate change on ecosystems and the well-being of people in China.

Key Shelterbelt Development Programs (SDP): The SDP have been widely implemented in the Three-North regions and the middle and lower tributaries of the Yangtze River in China. SDP include six sub-programs that cover all major river systems, costal lines, and vast mountain and plain areas in China (Table 2.1). With the full financial support and wide-range participation, these key shelterbelt programs have achieved great success. For example, the Three-North SDP, the largest shelterbelt program in China, have received world-wide recognitions. In 2007 and after 30 years of construction, the total area of reforestation has reached 23.74 million ha, the forest coverage has doubled and increased to 10.51%, the timber stock increased from 72 million m^3 to 130 million m^3, and the erosion-pro land areas in the Loess Plateau have been reduced by 40%.

Table 2.1 Brief descriptions of the China's Shelterbelt Development Programs (SDP)

Shelterbelt Program	Areal Extent	Program Target
The Fourth Phase of the Three-North Shelterbelt Program	Covering 590 counties in 13 provinces of Northwest, North, and Northeast of China	6.30×10^6 ha for tree planting; 1.26×10^6 ha for air-sowing; 1.94×10^6 ha for natural regeneration via hill closing
The Second Phase of the Yangtze River Shelterbelt Program	Covering 1,033 counties in 17 provinces	6.87×10^6 ha for tree planting; 6.29×10^6 ha for shelterbelt improvement
The Second Phase of Costal Shelterbelt Program	Covering 220 counties in 11 costal provinces	6.8×10^5 ha for tree planting; 6.2×10^5 ha for natural regeneration via hill closing; 6.0×10^4 ha for air-sowing
The Second Phase of the Zhujiang River Shelterbelt Program	Covering 188 counties of 6 provinces	2.28×10^6 ha for tree planting; 1.0×10^6 ha for shelterbelt improvement
The Second Phase of the Taihang Mountain Greening Program	Covering 112 counties of Hebei, Shanxi, Henan provinces and Beijing	1.46×10^6 ha for tree planting; 4.5×10^5 ha for shelterbelt improvement
The Second Phase of the National Plain Greening Program	Covering 944 counties of 26 provinces	4.2×10^5 ha for tree planting; 7.3×10^5 ha for shelterbelt improvement

The Wildlife Conservation and Nature Reserves Development Program: This conservation program is to protect biodiversity of genes, species and ecosystems by establishing 2,500 nature reserves. In addition, new conservation measures will be developed, such as hunting-free areas, reproduction bases, and wild plant cultivation bases. By the end of 2006, more than 2,000 nature reserves had been established, covering a total area of 1.21 million km^2, which accounts for about 14% of the total terrestrial area in China. China has also established more than 400 centers for natural resource conservation of wild plants and genes, and more than 160 botanical gardens or arboretums. These measures preserved more than 60% of the China flora with more than 1,000 rare and endangered plants being effectively protected.

The Forest Industrial Base Development Program (FIBDP): The FIBDP aims at enhancing timber production by planting fast-growing and high-yielding trees in most favorable areas of southeastern and northeastern China. FIBDP covers 886 counties in 18 provinces with a plan to afforest 6.18×10^6 ha and to improve 7.15×10^6 ha of low-productivity plantations. Forest plantations can complement natural forests and other land uses across the wider landscape. Thus, forest plantations as an important renewable resource will continue to grow in importance and to increase in China. However, forest plantations must be carefully distributed and properly managed in order to ensure positive economic and environmental effects on natural landscapes. For example, establishing forest plantations in areas previously occupied by natural or semi-natural forests in China should consider the possible significant loss of habitat for a wide range of species and potential increase in risks of biodiversity decline, soil degradation, pest and disease outbreaks and fire occurrence (Liu and Li, 1993; Liu et al., 1998a; Sheng, 2001; Whitmore, 2008)

The implementation of these six key forest programs nationwide has generated a great momentum for sustainable growth of forest coverage and timber stock and for improvement of forest quality and stand structure. According to the 6th national forest resources inventory taken in 2003, China has 8.21% of forest coverage, with 175 million ha of forested lands (i.e., 4.5% of the world's total) and 12.456 billion m^{3}of timer (i.e., 3.2% of the world's total) (Fig. 2.1). Of these forest resources in China, plantations account for 53.257 million ha and 1.505 billion m^3.

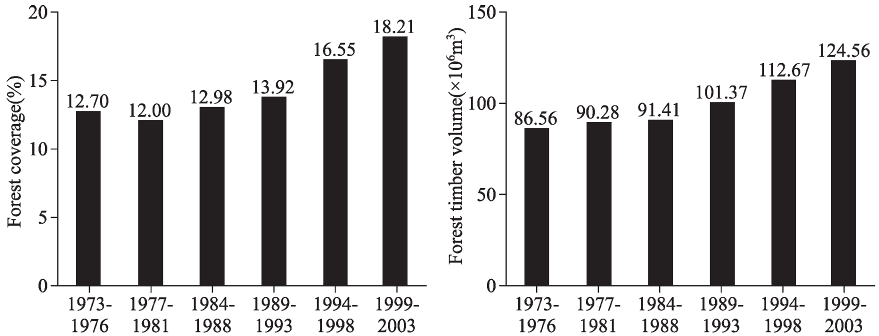

Fig. 2.1 Changes in forest coverage and stock in China during 1973-2003 (Xiao et al., 2005).

Although outstanding achievements have been abtained in forestry development in recent years, China still faces great challenges of insufficient forest area and stock, and low productive forest resources because of the long-term severe disturbances and over-logging. The area ratios of coniferous to broad-leaved to mixed forests are 47:50:3. Monoculture is the predominant practice with poplar being the primary tree species in the north and the Chinese fir or Masson pine in the south. The forests are primarily young stands (67.85%) with low volume growth per unit area. The average canopy closure is 54% with an average diameter at the breast height (DBH) of 13.18 cm. The average annual growth rate is 3.55 m^3/ha, and the average stock is 84.73 m^3/ha (i.e., lower than the world average of 100 m^3/ha).

Currently, China's forest resource cannot meet the national needs for wood and other forest products, which inevitably leads to the large amount of timber import. Given the annual timber consumption of 550 million m^3, the available timber volume in China can last for only about 10 years without import. The gap between the timber supply and demand is estimated to be 300 million m^3. In 2005 China imported an equivalent of 73 million m^3 of timber. The future forest resource used for timber production mainly comes from forest plantations. Although China ranks first in the world in the existing plantation area (53.2573 million ha), its stock volume is only 1.5045 billion m^3, which accounts for 12.44% of the entire forest stock volumes (stock per unit area at only 46.59 m^3/ha). The reasons for such low values are the poor management and the low site index.

2.2.3 Key issues for forest management in China

2.2.3.1 Deficiency in linking forest management with end usage

The main objective of forest management is to enhance forest productivity and ecosystem services to satisfy social, environmental, and economic needs for forest products. In this regard, clear and long-term vision for forest management is critical to the competitiveness and sustainability of forestry and associated industries. However, the interconnectedness of the forest production to the end uses of forest resources has often been overlooked in China. For example, plantations use only a few tree species, but their final utilization is not clearly defined. In addition to the traditional objective of using forest ecosystems as renewable resources (e.g., timber, biomass, water supply and quality), non-economic functions of forest, such as biodiversity, recreation, education, carbon sequestration and aesthetics, have become increasingly important. Such changes in societal perspectives have had profound implications for forest management (Spieker, 2002): sustainable forestry requires that forest management emphasizes multi-functions of forests, such as production of timber, pulp for paper, bio-fuels, maintenance of wildlife habitat and watershed health, and carbon sinks to combat climate change. These desirable end-products and goals must be defined clearly and explicitly in future forest management plans to ensure sustainable forestry development. Technological innovations and scientific advancements should contribute to new, sustainable, ecological and competitive products for forestry-based industries and to enhancement of ecosystem services critically needed in the face of increasing anthropogenic effects on environment.

2.2.3.2 Inadequate forest health management

Due to past long-term and large-scale anthropogenic disturbances, the existing natural forests in China are largely distributed in the remote mountain areas, in fragmented landscape, and/or in degraded secondary successional stages (Chen et al., 1994). Many natural forests in China have been put under protection to restore productivity, biodiversity, and ecosystem stability; but, the time required may be long because of their low resilience to disturbances. At the same time, China has the largest acreages of plantations in the world, and these plantations are mostly composed of single tree species and simplified stand structure and are more vulnerable to disease and pest infection, forest fire, extreme climatic disasters and even soil degradation (Xu, 1992; Sheng, 2001; Zhou and Sheng, 2008). In general, most of China's forests have low resilience in terms of low ecological stability and productivity to disturbances because of inadequate forest health management. Forest health management may need to consider manipulating forest composition, structure, and diversity to enhance ecological functions at genetic, species, ecosystem and landscape levels and to improve forest resilience to natural and anthropogenic disturbances.

2.2.3.3 Lack of integrated forest landscape management

For SFM, according to ecosystem based management, forest management objectives should be clearly defined before the management takes place (Wintle and Lindenmayer, 2008). In China, the current forest landscape regardless of geographical regions is likely to be a mix of primary forest, managed forest plantation, secondary forest and degraded forest lands interspersed with extensive areas of other agricultural land and rangeland, non-forest land-uses, due to the rapid land use and land cover change that leads to deforestation, forest degradation and forest fragmentation. At the same time, there are many more people living in these landscapes at present than in the past. Therefore, there is clear need to address multifunctional forest landscapes management to attain a large number of beneficial functions and services to human beings, which go far beyond agriculture and silviculture production (Foley et al. 2005). However, forest managers rarely consider the impacts of forest management on fisheries and aquatic biodiversity and downstream wetlands due to lack of an integrated forest landscape management. In addition, non-commodity outputs as well as a wide array of ecosystem functions considered to be indispensable properties of forest landscapes have been poorly recognized. Forest managers are being challenged by the necessity to consider all relevant landscape functions in forest landscape management. Integrated forest landscape management should take account of many factors, including interests of key stakeholders, the nature of the physical landscape, the resources available, the existing institutional and land tenure arrangements, and the prevailing land-use policy framework. Decision supporting system for integrated forest landscape management is needed to develop to facilitate sound forest landscape management decisions for ensuring landscape sustainability.

2.2.3.4 Unbalanced consideration on society, environment, and economics

Society has placed increasing demands for forest planners to balance diverse resource management objectives (Kneeshaw et al., 2000; Schulte et al., 2006). With adoption of SFM that emphasizes integrated land-use planning strategies with social, economic, and ecological dimensions (Lämås and Eriksson, 2003), China has invested heavily in the six key forestry programs (see discussions above) to expand forest resources and to improve eco-environmental conditions, especially in western China over the last twenty years. However, these programs were designed separately and independently, and implemented on a sectoral basis and in accordance with the individual program's objectives and legal constraints of each agency in charge. This requires extensive coordination and systematic integration in order to develop effective strategies of ecosystem management, identify land use patterns for multiple objectives of resource sustainability and ecosystem health, and explore solutions to location-specific environmental problems. At the moment, the expected overall synergic effects in terms of forest resource expansion and environmental improvement are yet to be fully achieved.

2.3 Challenges and emerging global issues in forestry

2.3.1 Coping with uncertainties of climate change

In the global context, forestry has become an important concern of many international conventions and a key component of human sustainable development. Climate change is one of most serious threats to forestry in China, and adaptive forest management must be implemented in order to reduce any negative effects resulting from climate change. The forest management today aims at maintaining ecosystem integrity and stability while achieving multi-functions of forests, such as timber, biofuels, carbon sequestration, water, and biodiversity. In response to challenges and emerging issues, there is now increasing demand for knowledge-based and process-oriented approaches to deliver customized management options for sustainable forest management.

Forests in China are under threads of changing climate (Bu et al., 2007; Leng et al., 2008). Changes in temperature and precipitation regimes have the potential to gradually affect forests in terms of forest structure, spatial distribution, growth and productivity. Some effects from rising temperature and increasing precipitation may be positive for forest growth and productivity (Liu et al., 1998b; Fang, 2000), while others (e.g., increased fire occurrence and pest and disease outbreaks) may be negative (Tian et al., 2003; Zhang et al., 2005; Wang et al., 2007). In addition, changing climate can affect hydrological process and water yield of forested watersheds, as well as the downstream water availability for both people and wetland ecosystems (Minshal, 1988; Poff and ward, 1989; Poff, 1996; Sun et al., 2008). The climate extremes can be highly detrimental to forest ecosystems (e.g., 21 million ha of forests damaged by the ice-storm occurred during the early spring of 2008 in southern China). Therefore, forest management needs to consider the uncertainties of climatic change and its effects on forests and environments in order to enhance the positive effects while reducing the negative effects.

2.3.2 Developing desicion suport systems for SFM

Traditional forest management in China focused primarily on tree planting and harvesting (Qu and Zhou, 2000), without the help of management tools to plan for whole-system based silvicultural operations. The "new information technologies" are changing the way of how forest management is conceived and applied by allowing for easy access and effective use of information and knowledge, thus enhancing participation and collaboration in decision making based on multiple objectives and functions of forestry. The forest management today is knowledge-based and process-oriented. There is now increasing demand for web-based DSS's to deliver customized management options for sustainable forest management (Jose and Keith, 2006; Jiang, 2008). One example is REMSOFT (Li et al., 2008), which can generate options for multi-purpose management planning to facilitate SFM.

2.3.3 Managing forest ecosystems at multiple scales

The traditional forestry focused on managing stand structure, growth process, and productivity at the stand level. With increasing understanding of the structure, functions and services of forest ecosystems, the forest management today aims at maintaining ecosystem integrity and stability while achieving multi-functions of forests, such as timber, biofuels, carbon sequestration, water, and biodiversity (Deng, 1998). Therefore, SFM should consider diversity and stability of forest ecosystems at genetic, tree, stand, ecosystem and landscape levels, and fulfill ecological, social, and economic functions at local, regional, country and global scales. Landscape planning and design to maintain regional landscape heterogeneity and diversity and to enhance landscape sustainability and resilience to disturbances must be at the core of SFM (Huang, 2004).

2.4 Contributions of landscape ecology to forest management and conservation

Landscape ecology plays a key role in facilitating the research development to address emerging issues of global forestry. Landscape ecology provides both theories and tools for forest management and planning, which enable land managers to assess the impacts of rapid and broad-scale changes in the environment (Turner et al., 2001). The concepts and theories of landscape ecology have helped change the traditional visions and ways of managing forest lands from stand level to landscape scale or even to regional scale. Forest management plans that combine the concepts of spatial heterogeneity, pattern-process, scale and spatial-temporal context of forest landscapes within a region are being developed and implemented by forest managers in China. The demands for sound adaptive management strategies should stimulate further development of theories and methods of landscape ecology, while the applications of landscape ecology in various forest landscape types under varying intensity of human disturbances should provide excellent experiment sites for landscape ecology research.

2.4.1 Ecological restoration

Forest restoration is to reestablish forest cover to produce economic products or restore ecological functions in areas where forests have been destroyed (Choi, 2007). In many cases, forest restoration is synonymous to reforestation and afforestation. The paradigm of ecological restoration considers the changing environments, global change, in particular, and emphasizes the maintenance of ecosystem functions and processes, rather than simply re-assembling the past floras and faunas (Choi et al., 2008). Ecological restoration has also shifted its focus from local degraded sites (or ecosystems) to landscapes, placing the emphasis on the roles of size and spatial configuration of forest patches in the targeted forest landscapes (Naveh, 1994; Be11 et a1., 1997; Aronson et a1., 1998; Schuller et a1., 2000). In addition, effective restoration of degraded ecosystems or landscapes requires the restoration of the natural disturbance regimes and the removal of artificial disturbances at the sites (Kuuluvainen and Aapala, 2005).

In recent years, ecological restoration in China has become a top management priority with increasing focus on ecosystem functions and biodiversity conservation as the national key forestry programs are being implemented (Liu et a1., 2003; Kong et a1., 2004). Many restoration projects are either completed, underway, or planned in China. For instance, Guan et a1. (2003) proposed the idea of constructing ecological safety pattern strategically to promote restoration of degraded landscapes. Long et a1. (2001) established an index system composed of water and soil erosion rates, forest cover, biomass and other variables to assess landscape change and its ecological consequences. Guo and Zhang (2002) analyzed distribution patterns and dynamic changes of landscape elements during the forest landscape restoration process in Guandishan Mountain. Kong et al. (2004) investigated how slope and elevation affected forest landscape restoration in the burned areas of Da Xingan Mountains in Northeast China. However, the new paradigm of ecological restoration has not been incorporated fully into the forest management planning in general and into most of the restoration projects in particular.

2.4.2 Biodiversity conservation

China is one of the countries with the richest biodiversity in the world. For example, China has approximately 6,481 species of vertebrates, accounting for 10 percent of the world total, and over 30,000 species of vascular plants with 17,000 species being endemic plants, ranking the third in the world (Zhang, 2002). However, massive deforestation (including timber harvest) during the 1950s and 1980s resulted in a huge loss of biodiversity associated with natural forests. Biodiversity is closely related to landscape change, such as patch dynamics and habitat fragmentation. Thus, species conservation should consider the integrity and diversity of ecosystems (habitat) and landscapes, with more emphases on the landscape approach than the species approach, because species distribution pattern, ecological processes, and their relationships operate at multiple scales and manifest at the landscape scale (Wu, 1992; Otte et al., 2007). It is likely to be the dynamics of patch mosaics per se that may hold the key to conservation of species diversity (Pickett and Rogers 1997). Biodiversity conservation should allow species to be adapted to variations in habitat types and patch configurations created by natural and anthropogenic disturbances to guarantee their survival in the changing forest landscapes.

Conserving large natural vegetation patches, protecting riparian zones and river corridors, and reducing habitat fragmentation by establishing stepping stones of suitable habitat are all the key landscape planning principles for biodiversity conservation (Forman, 1995). These principles are being used in China to create patch networks for biodiversity conservation in the context of global climate change. For example, Chen et al. (2000) and Lu et al. (2003)

assessed the landscape suitability for giant panda conservation in the Wolong Nature Reserve. In addition, Liu et al. (2001) examined the impacts of establishing the Wolong Nature Reserve on giant panda conservation and found that habitat loss and fragmentation have been unexpectedly intensified within the reserve. All these case studies provide knowledge for identifying suitable habitats and designing future reserves for giant pandas. Future forest landscape management in China should use landscape ecological principles to address issues about biodiversity conservation and adaptive management under potential climatic change conditions.

2.4.3 Forest eco-hydrology

Forest can help maintain and regulate hydrological processes, one of the most important services provided by forest ecosystems. Vegetation dynamics and spatial distribution of forests are largely controlled by climate and soil characteristics, whereas vegetation may affect climate by modifying the radiation, momentum, and hydrologic balance of the land surface (Foley et al., 2000). There is increasing concern with fresh water supply from forested watersheds because of the potential effects of climate change on forest cover. However, the current watershed hydrology in China is concerned mainly with land use/cover change on hydrological processes. Research is greatly needed to integrate the complicated interactive relationships between climate change, forest vegetation dynamics and hydrological processes at large landscape and regional scales. The roles of landscape structure or pattern change in watershed hydrological processes are poorly understood (Lin et al., 2004; Suo et al., 2005; Li et al., 2006). Studies showed that the vegetation composition in terms of forest, shrub and alpine meadow could affect the amount of water yield in a watershed (Jiang et al., 2004; Liu et al., 2006) and that the annual mean runoff coefficient and evapotranspiration (ET) may be closely related to landscape structure of watersheds (Li et al., 2006; Jiang et al., 2004; Liu et al., 2006, 2008). Optimization of landscape structure could improve utilization of water resources especially in semi-arid and arid regions (Lin et al., 2004; Li et al., 2006). In addition, changing landscape patterns in upper-stream forests may have severe consequences to down-stream hydrology and, thus, the ecological integrity of downstream ecosystems. Optimizing spatial pattern of forest vegetation by means of combining hydrological models with habitat models may meet critical needs for addressing hydrological issues at large scales. In recent years, the effects of climate change on either hydrological processes (e.g., precipitation, snow cover, snow melting) or forest vegetation dynamics are increasingly manifested in the upper Mingjiang River, the southeastern extension of the Tibet Plateau. An analysis of NDVI (Normalized Difference of Vegetation Index) indicated that vegetation activity showed great improvement over the period of 1982-2003, leading to the

40% increase in ET and consequently reduction in runoff (Sun et al., 2008). Sustaining and restoring watershed health (e.g., water supply and quality, stream integrity) must be a top priority of forest management because it is an integral part of hydrological processes.

2.5 Research needs for forest landscape management

Theories and approaches of landscape ecology are of highly importance to forest management by optimizing landscape planning for forest resource management and forest biodiversity conservation. Forest landscape management will change our traditional vision and way to manage forest lands from stand level to landscape scale for meeting multi-objectives of ecosystem services and diverse land use patterns. Forest management combining with spatial heterogeneity, pattern-process, scale and spatial-temporal context of forest landscape within a region even beyond forest boundary will further facilitate future forestry development and forest landscape management. Development of forest landscape management decision making tools appropriate to targeted forest landscape planning or special purpose is an important approach for addressing forest landscape management questions about how land use and forest ecosystem services can be optimized to achieve landscape sustainability. Current research needs of forest landscape management should be explored in response to the new emerging fields focusing on ecosystem rehabilitation and biodiversity conservation, carbon forestry, urban forestry, forest-based biomass energy and wetland protection.

2.5.1 Long-term forest ecosystem monitoring

Long-term monitoring must be the cornerstone of successful SFM. High quality ecological monitoring data, together with simulation modeling, provide the necessary knowledge of potential effects of climate change and forest management practices on forest ecosystems and various ecological services (Scheller and Mladenoff, 2005). It may be necessary to redesign monitoring systems in the context of climatic change in some cases. The monitoring of short-term carbon and green house gas (GHG) fluxes in forests should be incorporated into the long-term forest ecosystems studies. New monitoring indicators also need to be developed and included in the current monitoring systems to recognize the importance of climate change impacts on forest management. In addition, long-term data about natural disturbance regimes are the prerequisite for SFM in any attempts to enhance forest ecosystem resilience.

2.5.2 Cross-scale and multiple-purpose forest management

Paradigm change must take place at all levels of forest management hierarchies. Forest ecosystems are open systems operating through all the components linked in an interacting network of ecological processes (e.g., flows of energy, nutrient and water) cross scales (Jentsch et al., 2002). Therefore, forest management should aim at maintaining the complex biodiversity, healthy ecological processes, and reliable ecosystem services, and seeking the appropriate balance between biodiversity conservation and resource utilization. To achieve such integrated and comprehensive objectives, forest ecosystem management should use the concepts and principles of landscape ecology to develop appropriate spatial planning tools and DSS's to meet the long-term and multi-functional objectives, including biodiversity conservation, water and soil protection, carbon sequestration, ecotourism and ecosystem services.

2.5.3 Landscape decision support systems

Landscape dynamics models are necessary and useful to assess the effects of forest management and climate change scenarios on forest (He and Mladenoff, 1999; Scheller and Mladenoff, 2005, 2008). Many complicated issues and scenarios can only be evaluated with the help of comprehensive DSS's. However, most ecological models used in China were developed outside China, and such introduced DSS's may be of limited values and applications (Liu et al., 2006). Therefore, it is necessary to modify the introduced models or develop new models specifically designed for China to meet the requirements of the specific landscape configurations and management objectives, especially those models for assessing effects of forest management and for carbon and GHG accounting.

2.5.4 Fragile forest ecosystem management and protection

The threat of climate change to fragile forest ecosystems is the most serious problem in managing forest resources. The sensitivity of forests in China to climate change varies with regions, ecosystems and climatic factors. For instance, mangrove forests are highly sensitive to the sea-level rise maybe caused by climatic warming; forests in semi-arid and arid regions are vulnerable to changes in pattern and amount of precipitation; forested wetlands are susceptible to variability in hydrological regimes in both wetland and upland forests. Recent studies have also found that the transitional zones between forest and other ecosystem types may be more vulnerable to climatic change (Neilson, 1993; Allen and Breshears, 1998; Loehle, 2000; Noss, 2001). Therefore, improving health, stability and resilience of fragile forest ecosystems should be a top priority in forest management planning and forest landscape restoration, in addition to preventing forest degradation, fragmentation, and alien species invasion to maintain biodiversity and ecosystem functions.

2.5.5 Adaptive forest management

As responses to climatic change, the distributions of some tree species in China may move northward and up in elevation and, as a result, new assemblages of species may emerge (Xu et al., 1997). At the same time, the extreme climate events (e.g., hot spot, severe drought) may have harmful impacts on forest ecosystems directly by damaging forests and trees or indirectly by altering patterns of pest and disease outbreaks and fire occurrence. Conventional forest management strategies in China are unlikely to be able to cope with the uncertainties associated with climate change and to meet growing needs for ecosystem services and forest products. It is crucial that adaptive forest management should be developed for current and future forest landscapes under different climate change scenarios. Implementation of SFM through the adaptive management process can contribute to the reduction of negative environmental, social and economic impacts on forest and forestry caused by climatic change. Adaptive forest management must consider principles of landscape ecology and disturbance theory and integrate multiple objectives, including improving land productivity and ecosystem health, enhancing ecosystem services such as water and soil protection, biodiversity conservation, and carbon sequestration, facilitating and coordinating development of effective policies, programs, and actions, and ensuring sustainability of all social economic benefits from forest ecosystems. For example, since landscape fragmentation induced by human activities is likely to have more serious impacts under climate change conditions on forest biodiversity conservation, establishing patch and corridor network is essential to facilitate migration of plant and animal species under potential future climatic conditions. Adaptive forest management must also assess the possible changes in natural disturbance regimes induced by climate change through simulations with spatially explicit and process-based landscape models. Thus, research is needed to develop a system of adaptive forest management strategies that suits unique characteristics and situations in China. One aspect of such research is to educate land managers and the public about the principles and process of adaptive forest management, such as getting all shareholders involved in the planning phase. Major changes in attitudes must take place before adaptive forest management can become the new paradigm in China.

2.6 Concluding remarks

China has gained great achievements over the last 20 years in sustaining continuous growth of forest areas and volumes, improving biodiversity conservation, and developing effective strategies and policies of forestry and forest ecosystem management. One example of the achievements is the successful implementation of the national key forestry programs. The three-phase history of changing forest management focuses also exemplifies the great progress in China's forestry. However, China's forestry is still insufficient with large areas of forests being of the poor quality and low productivity, which poses great challenges in meeting the increasing demands for desirable goods and services. Effective forest management may help resolve the problems. To be successful, new forest management strategies must be based on better understanding of emerging global challenges and issues and reorganization of climate change as the most severe threat to future forests and forest management.

Given the complexity and uncertainties of climate change, adaptive forest management must be developed to combat current and future implications of climate change to forest ecosystems. Adaptive forest management based SFM principles can help prevent or at least contain forest degradation, enhance forest resilience, and reduce negative environmental, social and economic impacts on forestry and forest ecosystems posed by climatic change. A key component of adaptive forest management is to define operational guidelines to carry out the goals and objectives, in which landscape ecology can play a major role. Landscape ecology provides theories and management tools for forest restoration, biodiversity conservation, land and water resource management, and forest management planning to help cope with climate change and facilitate SFM. For example, forest management should consider ecological patterns and processes, disturbance regimes, spatial heterogeneity, and forest landscape configurations at multiple spatial-temporal scales to achieve comprehensive ecological, social and economic benefits. Forest researchers and managers in China should advocate and adopt the adaptive management approach to ensure sustainability of landscape biodiversity, health, and functions and processes.

In addition to policies, guidelines, and strategic planning, forestry research must lead the way to sustainability of forest resources. To alleviate impacts of climate change and anthropogenic disturbances on forest ecosystems in the future, cross-disciplinary research activities are needed, including long-term forest ecosystem monitoring, improvement of forest productivity and ecosystem services, decision-support system development and applications, and adaptive forest management that integrate multiple objectives at multiple spatial and temporal scales. The emerging global issues also require strong international collaborations such that research can be advanced in broad scopes and at fast paces to help decision-makers and land managers cope with the rapidly changing environment and associated problems in forest management. It is critical that close relationships between science and policy (i.e., policies based on science while science used explicitly to address policy-related issues) can be established to assure and promote significant research contributions to SFM and climate change mitigation and adaptation.

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Chapter 3 Issues Facing Forest Management in Canada, and Predictive Ecosystem Management Tools for Assessing Possible Futures

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Abstract

Forestry has been changing throughout its history in response to changing needs of human populations and changing supplies of forest resources and values to satisfy these needs. Canadian forestry has undergone a series of changes that reflect much of the global pattern of the change in this human activity, and considering the extent and diversity of Canadian forests, they are now amongst the best managed in the world. However, change continues in the face of continuing challenges and environmental, social and economical issues. Some of these are discussed briefly in this chapter. We also describe one contribution to the resolution of some of these issues in forestry: hybrid simulation, ecosystem management models that span from individual trees (for complex mixed stands) to landscapes of various sizes. The family of models that is briefly described is based on the FORECAST model. Emphasis is given to the LLEMS landscape model.

Keywords

Canada, forestry, issues and challenges, ecosystem modeling, FORECAST, LLEMS.

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3.1 A brief history of forestry in Canada

As in most forested countries, forestry in Canada is continually changing. In contrast to many countries the history of this change is relatively short. A significant proportion of the forest in the northern parts of many Canadian provinces remains in a relatively "natural" (pre-European colonization) condition, whereas many forests in the south have been significantly changed. This reflects timber harvesting, clearance for agriculture, fire control, a reduction in the historic influence of our "First Nations" (the pre-European inhabitants of Canada), and, in some areas, oil and gas exploration. Any discussion of current Canadian forest management and possible directions for future change should be framed by an understanding of this history and its variation across this vast country. The objective of this chapter is to consider some of the key issues facing Canadian forestry, and to explore one example of the type of ecosystem-based decision support tool that we believe is a pre-requisite for achieving sustainable forest ecosystem management in the context of these issues. A review of other types of forest models can be found in Messier et al. (2003).

Before looking briefly at Canada's forest history it is useful to consider the more general patterns of development in the relationship between people and forests that have occurred at various times and places around the world. Early human societies simply exploited forests for a wide variety of values — both timber and non-timber. Such exploitation[1] was sustainable because the rate of utilization was less than the rate of natural replacement. This reflected one or more of: low human populations, low per capita demand for the resource or value, low levels of technology that limited the rate of consumption, or the rapid renewal of the resource or value by natural processes (Salim and Ullsten 1999). As populations and the power of technology increased, human utilization of forest resources began to exceed the supply and rate of renewal, at which time the exploitation became non-sustainable and the supply declined. Human response was either to become nomadic to secure resources from uninhabited areas, or, where this was not possible, to invoke taboos and belief systems (e.g. religious edicts) to protect local forests. Alternatively, remote forest areas were colonized by force. Much of the European colonial period involved exploitation of timber in other countries for ships to maintain trade and military superiority. When these mechanisms failed to ensure desired future supplies, forest management evolved (Winters 1972).

Early forest management has always tended to be politically or administratively organized, rather than be based on recognition of the spatial and temporal diversity in the ecological character of forests. It tended to be based on regulation of an inventory of existing forest values, rather than manage-

[1] We define *exploitation* as the utilization of a resource or value without any overt action to ensure the future supply of that resource or value, either ignoring resource renewal or relying on natural processes to accomplish it.

ing the ecosystem processes that ensure their renewal. The failure of such non-ecological approaches to forest sustainability generally leads, after various periods of time, to an ecologically-based approach. One of the main values delivered by early, "administrative" forest management was generally wood for fuel, timber for industrial purposes, or wood fibre for non-solid wood products. The main emphasis at the start of the subsequent stage in the evolution of forestry (*ecologically-based forestry*) was also on tree growth and wood yield. However, as "modern" societies have developed a renewed interest in other forest values (e.g. wildlife, potable water, non-timber forest products, aesthetics, recreation, many of which were valued by earlier societies), and added concerns about biodiversity, carbon budgets and climate change effects, the need to manage ecosystem processes at both stand and landscape scales to sustain multiple values has become apparent. This leads to the beginnings of what forestry really should be — *ecosystem management*. However, this historical pattern of development from exploitation to ecosystem management has rarely developed in a linear manner, being commonly interrupted by conflicts and social changes, and there are many impediments to true ecosystem management.

How has the development of forestry in Canada reflected this generalized pattern? Many of the First Nations of Canada lived in and depended on forests for their resources and survival. They had a well developed, experience-based knowledge of the forest values they depended on, and practiced either sustainable exploitation or passive/active management of these largely non-timber values. They also practiced warfare to ensure access to forest-based and other resources. Their management involved family or tribal ownership of hunting and gathering areas, while active management involved the use of fire to clear forests, maintain wildlife habitat, promote hunting and food gathering, and/or protect themselves against wildfire and enemies. This continued for thousands of years (Drushka 2003). The arrival in Canada of Europeans brought diseases that decimated many First Nations and resulted in a loss of experienced-based wisdom because of the lack of written language. The acquisition of guns and metal tools altered the First Nations' ability to harvest wildlife and trees, but this was generally balanced by the reduction in their populations. The reduction or elimination of First Nations' use of fire resulted in significant changes in forests in historically fire-dominated areas (MacKay 1979). However, it was the arrival of Europeans that initiated significant and frequently unsustainable exploitation of Canadian forests (Drushka 2003).

Because the colonization of Canada by Europeans began in the east — the closest to Europe — the history of human-induced forest change is longer there than in the west, and the changes in the forests are most apparent (Craig 1988, Frelich 2002). Logging of pine and spruce for ships masts and timbers, and of lumber for export to timber-starved Europe (where forests had been decimated by war, industrial harvest for fuel, and clearing of forests for agriculture) significantly altered the species composition and age-class structure of eastern Canadian forests, and is thought to have changed the historical natural disturbance regimes associated with insects and fire. This European impact on Canadian forests began in the 1700's in the east, but gained momentum in the 1800's and the early 1900's. Major forest harvesting in British Columbia (BC) did not begin until after World War I, although the gold rush in 1858-1865 brought in 33,000 miners, many of whom found jobs in forestry after the gold rush was over. European settlers on the southern coast of BC in the wake of the gold rush initiated logging of coastal old growth forests in the latter 1800's, and the construction of the transcontinental railway initiated forest clearing and harvesting across BC's southern interior at about the same time. The early industrial logging was driven largely by growing demand for high quality "old growth" lumber in the US Pacific Coast, and export of wood products to the US has continued to be the major driver of forest harvesting and a major contributor to the economy in Canada.

3.2 Canada's lands and forests

Canada is the second largest country in the world, with the third largest area of forest in the world after Russia and Brazil. Canada spans 41 degrees of latitude and 87 degrees of longitude, and has a forest area of 402.1 million ha — 10% of the world's forests and 30% of the global boreal forest (Natural Resources Canada) (Table 3.1). Canada's forests range from temperate deciduous and semi-Mediterranean forest, through savannah, dry and wet temperate conifer forest (including temperate rainforest on the west coast), to subalpine conifer, boreal mixedwoods (deciduous and conifers mixed) and wet and dry coniferous boreal forest. Canada's west coast supports about 25% of the world's extent temperate rain forest (MacKinnon 2003). The western province, British Columbia, accounts for a majority of Canada's total ecosystem diversity — climatic, geological, soils, topographic and natural disturbance regimes, and more than 50% of most measures of the biological diversity associated with this physical diversity. BC has 60% of Canada's vascular plant species, 75% of the bryophyte species, 70% of the bird species, 80% of the mammalian species, and over 60% of the Canadian insect species (Pojar 1993).

Forestry in Canada is mostly under the jurisdiction of the governments of its ten provinces and three territories. The federal government has a network of research centers across the country that are responsible for national inventories and research concerning forest protection, trade, economics and other topics that transcend provincial boundaries, but the actual management of forests for multiple values is provincial/territorial (Table 3.1). Canada also has a network of 14 model forests representing the major forest regions of the country (Natural Resources Canada 2008). Canada's forests are divided into 12 forest regions, each with sub-regions. These represent major climatic and physiographic subdivisions of the country. There is also a national system of "ecological (biophysical)" land classification (Table 3.1), but forest management at the provincial/territorial level generally employs one of several types of ecosystematic classifications as the ecological foundation for silviculture and stand management (e.g. the biogeoclimatic (BEC) classification of BC (Table 3.1)). The BEC system is one of the most advanced ecosystem classifications specifically designed as the foundation for forest management anywhere in the world, something that reflects the high degree of physiographic, climatic, edaphic, geological and biotic diversity in this western province, and the public ownership of 95% of the forest. In most Canadian provinces some type of ecosystem classification is legally required as the basis for silvicultural and harvesting decisions, which are supported by detailed ecological guidebooks and manuals based on the BEC or similar systems.

Table 3.1 List of some electronic resources about forest management, policy and research in Canada (Websites last accessed on June 15, 2009).

Organization	Topic	URL address
Canadian Forest Service	Forest Research Centers in Canada	http://cfs.nrcan.gc.ca/
	Depository of literature on Canada's forests	http://bookstore.cfs.nrcan.gc.ca
Canadian Model Forest Network	Model Forests in Canada	http://www.modelforest.net/cmfn/en/
Canadian Forestry Association	Climatic and physiographic regions of Canada	http://www.canadianforestry.com/html/forest/forest_regions_e.html)
Natural Resources Canada	Statistical data of the forest sector in Canada	http://canadaforests.nrcan.gc.ca/?lang=en
Nature Serve Canada	Canadian National Vegetation Classification System	http://www.natureserve-canada.ca/en/cnvc.htm
Alberta Sustainable Resource Development	Forest Management at provincial level: Alberta	http://www.srd.alberta.ca/
BC Ministry of Forests and Range	Forest Management at provincial level: BC	http://www.gov.bc.ca/for/
	BC Biogeoclimatic Ecosystem Classification	http://www.for.gov.bc.ca/hre/becweb/
New Brunswick Ministry of Natural Resources	Forest Management at provincial level: New Brunswick	http://www.gnb.ca/0078/index-e.asp
Ontario Ministry of Natural Resources	Forest Management at provincial level: Ontario	http://www.mnr.gov.on.ca/
Sustainable Forest Management Network	Research consortium on SFM	http://www.sfmnetwork.ca/html/index_e.html

Most of Canada's forest land is publically owned, the balance is divided between more than 450,000 private owners. One of the major differences between forestry in eastern and western Canada is differences in the ratio of public to private ownership of forests. In Prince Edward Island 96% of the timber is cut on private lands, compared with 11–12% in the three western provinces. Newfoundland, the most easterly province, does not fit this trend, with only 3% of the timber harvest from private ownerships (Rotherham 2003). The regional variation in ownerships is a major factor determining variation in forest practices in Canada and throughout North America (FAO 2009). Large areas of government-controlled forest land make the application of ecologically-based regulations easier than where forest ownership is in a large number of small private parcels. However, property rights in public forests can be a major impediment to the development of ecosystem management tenures (Tedder et al. 2002). The diversity of land ownership, forest history, cultures and politics across Canada interacts with the ecological diversity, the components of which were noted above. This requires a region-specific, landscape-specific and stand-level ecosystem-specific approach to forest management, and results in region-specific forest issues in addition to those issues that are common to all of Canada's forests.

3.3 Issues facing forestry in Canada today

Forestry in Canada has advanced to the stage of ecosystem-based management (EBM), but has not yet succeeded in proceeding to true ecosystem management (EM). This reflects the fragmentation of management responsibility for different values on public forest lands between different government agencies, the small size and limited ownership rights on many private forest lands, and the tenure structures on public lands that limit revenue-generating management to timber-related values. All other values are a limitation on that objective. The lack of integrated overall planning (with value trade-off and scenario analysis) and management for all values over landscapes of appropriate extent renders true ecosystem management currently beyond our reach. There are examples where this is not true or only partially true — such as in some community and municipal forests and in Canada's model forests, but it is true for the main forest area, despite the very desirable emergence of ecosystem-based management as a goal across much of the country. Despite great advances towards sustainable forest management and our current status as having probably the best overall management of forests of comparable size and ecological diversity in the world, forestry in Canada faces numerous issues, including, but not limited to, the following (not in any order of priority):

3.3.1 Lack of recognition of the role of natural disturbances

Most of Canada's forests are disturbance-driven: historically by fire, insects and/or wind, and, over the past 200 years in the east and the past 100 years in the west, by timber harvesting (Suffling and Perera 2004). Most of our forests have had some degree of human influence since trees first colonized bare ground following the retreat of the last ice age. The combination of human and non-human disturbance regimes has created landscape level biological diversity in addition to that determined by the diversity of physical environments. By maintaining a shifting mosaic of stand conditions and ages, disturbance regimes have sustained ecosystem productivity, wildlife habitat and biodiversity. Some of the natural disturbances have been altered in scale, frequency and severity by human action firstly by First Nations use of fire (Vale 2002), and secondly by reductions in wildfire over the past 50 to 100 years (Brown and Sieg 1999). Fire control has also been blamed as a major contributor to a vast outbreak of the mountain pine beetle that has decimated more than 7 million ha of lodgepole pine forest in British Columbia (Taylor et al. 2006). Fire control had increased the age of these forests, and, in conjunction with several years of mild winters, had created ideal habitat and survival conditions for this bark beetle. Extensive outbreaks of insect defoliators are a feature of eastern and boreal forest, and there are suggestions that selective removal of certain tree species over the past 150 years may have increased the severity of these outbreaks (Shore et al. 2006).

The key issue related to forest disturbance is the opposition by many environmentalists, and as a result by a large segment of society, to logging. Many insist that this human-made disturbance is bad and that forests should be managed with the lowest levels of disturbance possible, or not at all. Although some of the concerns of environmental groups are justified (such as extensive clearcutting in areas where the historical scale of disturbance has been at a much smaller spatial scale), the non-disturbance view does not respect the ecology of many of Canada's forests, nor does the attempt to replace clearcutting by partial harvesting everywhere. Increasingly, forestry in Canada seeks to balance the desire to emulate the ecosystem effects of historical natural disturbances (landscape patterns and stand characteristics) that have been altered in their frequency, severity and extent by contemporary society, with the rejection by the public of the visual and short-term ecosystem consequences of disturbances that are in fact needed to sustain long-term diversity, productivity and aesthetics. The short-term visual consequences of natural disturbance emulation are frequently interpreted as "ecosystem damage" rather than a necessary part of the long-term ecology of desired values. There is a need for decision support and communication tools that can demonstrate the potential consequences for Canada's forests of deviating significantly from historical disturbance regimes. For a discussion of disturbance ecology, see Attiwill (1994) and Perera et al. (2004).

3.3.2 Need to include climate change effects on forest planning

The need to include strategies of adaptation to and mitigation of climate change effects is widely recognized in Canada (CCFM 2008 (Table 3.1)). Although there have been many estimates of the effects of climate change on Canada's forests, we cannot yet predict with confidence the long-term consequences of climate warming (Redmond 2007). The major effect may be increases in fires and insect epidemics, and possibly some forest disease issues (Bergeron and Flannigan 1995). There will undoubtedly be direct effects on seed production, regeneration, and tree physiology, and in flat topographic areas climatic zones may move significant distances (Hamann and Wang 2006). Recent research has documented migration of tree species in the US (Woodall et al. 2009) and increased rates of tree mortality in the Pacific Northwest (van Matgem et al. 2009), and bioclimatic models have become a widely used tool for assessing the potential responses of species ranges to climate change (Beaumont et al., 2005). However, in more mountainous topography, the effects of climate on determining forest composition are strongly modified by aspect, slope, slope position and soil moisture and fertility. As a consequence, changes in local climate may have rather more subtle effects on the spatial distribution of plant communities than those suggested. The complexity of the interactions suggests that some of the more dramatic pronouncements about "bioclimatic envelope" shifts may only have validity in flatter areas (Hamann and Wang 2006). These authors are revising their predictions concerning ecological zone shifts in British Columbia as a consequence of reducing the error associated with the bioclimatic models they use (Tongli Wang, UBC, pers. com.). Some researchers have criticized the "bioclimatic envelope" approach because it does not represent biotic interactions, evolutionary change and species-dispersal strategies and limitations (Pearson and Dawson 2003). After all, our long-lived species have survived through major climate shifts over the past millennium, suggesting that their ranges may be less sensitive to climatic change than suggested. Climate effects on trees may have more to do with seed production and recruitment of seedlings than with mortality of mature trees, resulting in considerable time lags in changes in tree species distributions. These effects could be explored through ecological models (Nitschke and Innes 2008, Blanco et al. 2009). Until we understand more about climate change effects and their potential variation in different parts of the country, it is difficult to develop coherent forest policy with respect to climate change. One of the tools needed to explore this important and complex topic is ecosystem management models that represent key ecosystem processes, and the effects of climatic variables on these processes.

3.3.3 Need to include carbon budgets in forest management plans

West coast "old growth" forests contain some of the highest stores of carbon of any terrestrial ecosystem. With the public preoccupation with carbon storage rather than with understanding total carbon budgets, the prevailing public opinion is that we should reserve all our old forests as carbon stores, allow younger forests to become older, and protect them from logging, fire and insects (Cannell 1995). Unfortunately this very complex question is generally oversimplified, and results from stand-level carbon inventories and budgets in one type of forest are often extrapolated uncritically to very different types of forest. An essential aspect of carbon budget analysis is the recognition of the temporal scales associated with the changes in forests as sinks, neutral or sources of atmospheric carbon, and there is often inadequate consideration of landscape-level budgets and budgets over longer time spans, although there are encouraging advances in this field (Trofymow et al. 2008). The assumption is frequently made that old forests continuously sequester more carbon, whereas in reality the capacity of forests to act as a net carbon sink generally declines with age as ecosystem respiration begins to equal or exceed primary production and the nitrogen cycle slows down; there is debate over the age at which this occurs (Buchmann and Schulze 1999). Similarly, while several analyses have shown that some undisturbed old-growth forests have significantly greater total quantities of organic carbon than younger managed stands, net annual carbon fixation rates in managed young stands are consistently higher than in old, unmanaged stands (Smithwick et al. 2002). Most in-stand studies in northern temperate forests show that harvesting and replacement of old forest by productive young forest are carbon neutral or slightly negative (Thornley and Cannell 2000) and do not result in significant losses of soil carbon following harvesting (Yanai et al. 2003).

If product replacement (e.g. use of wood instead of steel and concrete — materials that have a greater carbon "footprint") and fossil fuel displacement (unused wood fibre converted to biofuels) are accounted for, harvesting old forest and replacement by younger forest makes a positive contribution to the issue of climate warming. However, the analysis is sensitive to assumptions about long-term storage in the resulting wood products, which led some earlier studies in the US to conclude that the best carbon strategy there is to retain old forests (Harmon et al. 1996).

The issue related to carbon in forests is the tension between advocates of storage vs. advocates of carbon budgets and sequestering — using forests as carbon pumps, not static/declining stores. The debate is complicated of course because old forests offer many other values besides their carbon functions, and these justify the reservation of certain old forests. Also, a focus on carbon stores in many of Canada's forests ignores the periodic release of stored carbon by wildfire (Kurz et al. 2008a, b). What is needed to compliment the work by the Canadian Forestry Service on national carbon budgets (Kurz and Apps

2006, Kurz et al. 2008a) is to drive these budgets by stand-level ecosystem process models linked to life cycle analysis models that track post-harvest carbon storage and fossil fuel displacement.

3.3.4 Lack of understanding of a variety of key ecological concepts

Sustainability, ecosystem resilience, stability and integrity, and biodiversity and "old growth" are all concepts that are part of the foundation of sustainable forest management for multiple values, without an understanding of which forest policy and conservation strategies may fail to meet their objectives and may not satisfy the public's expectations. Inadequate or conflicting understanding of these complex concepts and their operational applicability by some politicians, policy makers, resource managers, researchers, environmentalists and the general public hinders progress. Similarly, the need for site and value-specific management that is based on ecosystem sciences, and the need to practice ecosystem management rather than the management of individual values based on their individual ecologies are not yet widely recognized and accepted in Canada. Ecosystem "health" and "integrity" are discussed in Kimmins (2004). Sustainability is discussed in Nemetz (2007), and forest sustainability in Kimmins (2007).

An interesting recent development in ecology is the growing use of statements such as "ecosystems are complex adaptive systems", and terms such as "emergent properties" (system properties that cannot be deduced from knowledge of individual system components) (Anderson 1972). It seems to us that this reflects the realization on the part of ecologists who had previously focussed on levels of biological organization below that of the ecosystem (individuals, populations or communities) that nature cannot be understood, explained or predicted at these levels without considering their place in ecosystems. This is not new: the great debate about density dependent vs. density independent regulation of populations that raged between animal population ecologists in the mid 20th Century was largely the result of the failure to recognize that ecosystems are complex, that population processes vary with physical environments, and that the future of any level of biological organization can only be successfully predicted in the context of the next true level of integration above: the ecosystem in terms of ecology (Huffaker and Messenger 1964, and Rowe 1961). Population futures are not predictable in particular ecosystems outside of a consideration of all key ecosystem determinants of population dynamics in those ecosystems (see discussion of multi trophic level regulation of populations in Sinclair et al. 2000 and Tscharntke and Hawkins 2002).

The concept of ecosystems as "complex adaptive systems" entails a redundancy. Ecosystems are by definition complex (Tansley 1935), and they are the "emergent property" of the combination of biological communities and environmental factors. The attribution of the individual-level and species-level concept of "adaptive" to ecosystems is also troubling; ecosystems are neither "born" nor do they die; they do not have a physiology that can acclimate, nor a frequency of genotypes that can become adapted through natural selection. The concepts of ecosystem extirpation or extinction would appear to be a stand-level that refer only to the biota of the ecosystem or to individual species. Stand-level ecosystems are embedded in landscapes of various spatial scales, and have a physical as well as biotic component. It is the meta-populations, meta-communities and meta-ecosystems (the landscapes) that truly define ecosystem function, pattern, stability and resilience (however these are defined), rather than the small scale, local subcomponents thereof. Accompanying these semantic and conceptual difficulties is the variation in the concept of resilience (Holling 1973), which sometimes is defined in the same work as "inertial stability" and sometimes as "elastic stability" (e.g. Puettmann et al. 2009, which uses both definitions). In reality, resilience as inertial stability is a term that refers mainly to the biotic components of the ecosystem because many of the physical and chemical components remain relatively unchanged after disturbance that causes significant biotic change. Resilience as elastic stability involves both biotic and physical/chemical ecosystem processes as ecosystems develop post-disturbance.

Our concern over the possible misuse of such terms is that those concepts that have validity at lower levels of biological organization but not at the ecosystem level may be adopted as the ecological foundation for the management of whole ecosystems and will disappoint us accordingly (Kimmins 2008; Kimmins et al. 2005, 2008). It is time to marry useful stand-level biology and ecology to our understanding of landscapes, and use knowledge of ecosystem dynamics in the face of successional processes and disturbance (Attiwill 1994; Frelich 2002; Perera et al. 2004) rather than develop a new and often redundant set of nomenclature and theory. To explore this topic we need ecosystem-level models that can also be run at population or community levels to investigate the importance for prediction of modeling at the ecosystem level (Kimmins et al. 2008).

3.3.5 Lack of recognition of the complexity of ecosystem-level issues

Politicians, the general public, many environmental groups and frequently forest managers are simply not equipped to recognize, understand and deal with the social and ecological complexity of forestry. As Bunnell (1999) said "forestry is not rocket science — it is much more complex". William of Occam (the source of "Occam's Razor" — a fundamental tenet of science) noted six centuries ago that "theory, explanations and actions should be as simple as possible, but as complex as necessary", a thought echoed by Albert Einstein more recently — "theory should be as simple as possible but not simpler". Society, science and forestry have all been slow to embrace this important need to recognize and account for complexity.

A problem is an issue that does not get solved; an issue that gets solved quickly is not a problem; problem issues often persist because they are complex and only simple solutions are offered (Kimmins 2008). The current discussion of "complex adaptive systems" is a welcome recognition of ecosystem complexity, but its implementation should be rooted in mainstream, ecosystem-level science of ecosystem function, temporal dynamics and spatial diversity of ecosystems. In Canadian forestry we need policy and management decision support tools that can address complexity at the ecosystem level and marry biophysical forecasts, based on an adequate representation of forest ecosystem complexity, to social desires and needs.

3.3.6 Lack of a landscape perspective and a sufficiently long time scale

There remains a preoccupation with stand-level conditions over short time spans, especially in the minds of the public and certain environmental groups, but also involving some researchers, resource managers and others (Kimmins et al. 2005). The major issues in Canadian forests are generally landscape level and long term, yet there is public antipathy towards licensing forest management institutions to manage large, public forested landscapes over management-relevant and ecologically-relevant time scales. Landscapes are shifting mosaics of changing stands, so clearly the stand level is important. However, the key issues facing forestry transcend stands, and often involve substantial landscapes. Some examples of operational forestry at larger spatial scales in BC can be found in the application of the BEC system in management (Mah and Nigh 2003) or the implementation of management based on natural-disturbance emulation (DeLong 2007). Again, the need is to use ecosystem-based decision support tools that can span ecologically-relevant time and spatial scales (linking stand-level process models with large landscape models), and communicate to a variety of stakeholders our best science-based forecasts (educated guesses) as to the possible outcomes of alternative approaches to forest management.

3.3.7 Using inadequate support tools and predictive models

If all factors affecting the future development of forests remained as they were in the past, traditional, "historical bioassay" population, stand, and landscape models that reflect the past based solely on experience would be the best type of decision-support tool. However, with changing public expectations for multiple values, changing management methods and changing climates, the inflexibility of simple, experience-based, tree population tools renders them of questionable utility. They should be combined with knowledge-based tools to permit a plausible range of forecasts for changing and uncertain futures for multiple values. Where they are used to support the management of ecosystems for multiple values, they should be ecosystem-level. Models that predict only a single value (e.g. timber or tree growth) based on a single limiting factor (e.g. crown space, light) may be useful for forests in which the selected limiting factor is the only important one and where a single value is the object of the modeling. They are of relatively little value in systems in which there are multiple limiting factors expected to change in the future, and where forecasts must be made for multiple values and permit value trade-off analysis and ecosystem management scenario analysis.

3.3.8 Incorporating public opinion into forest management policies

Fortunately for Canada's forests and their many values, forestry in this country has advanced from an almost totally timber and economics focus to the inclusion of multiple values — cultural, social, biological, environmental and even spiritual — as objectives of management. This evolution has been driven by professional foresters, researchers and environmental groups, and over the last 30 years a consensus has emerged that the public should be involved in the management of Canadian forests (Hamersley and Beckley 2003). The general public has long felt that important issues were not accounted for in forest management, and often the political barriers that frustrated the efforts of foresters and scientists to change traditional forestry required the public action of environmentalists (Wagner et al. 1998). Recent experience has shown that incorporating public opinion and local knowledge can lead to better management decisions, reduced conflict, and greater compliance with sustainable forestry regulations (Hamersley and Beckley 2003). Creation of ecological reserves, parks, reserves of "old growth", wildlife reserves and improved riparian management have contributed to biodiversity objectives, and the sustainability of multiple values has been improved by a policy change to emulate in our landscape harvesting patterns the mosaics of stand ages and composition resulting from past natural disturbance. Not all of these positive aspects of change have occurred everywhere. Sometimes governments are slow to change policy. Sometimes forest companies are slow to recognize the need to change management practices if they wish to retain a social license to operate. Similarly, not all the pressures from environmentalists have been positive, and in some case these have interrupted the evolution of forestry and returned it to an earlier, less desirable stage (Drews 2008). As noted by Aldo Leopold (1966; p 263) "The evolution of a land ethic is an intellectual as well as emotional process. Conservation is paved with good intentions which prove to be futile, or even dangerous, because they are devoid of a critical understanding either of the land, or of economic land use." Public opinion has to be informed by clear and well-defined indicators of the sustainability of desired values, and about the current state of forest management with respect to these indicators. As Hebert (1999) has stated "Sustainable forest management is really just an attitude. It begins with wants and values, is driven by people and eventual policies, and is fine tuned by science". Spies and Duncan (2009) demonstrated the dangers of forestry and conservation strategies driven by incomplete information and understanding. To assist in communicating choices to various publics and other stakeholders, the ecosystem-level decision support tools mentioned above and later in this chapter need to be linked to advanced, interactive visualization systems (e.g. Sheppard and Harshaw 2000) .

3.3.9 Lack of adequate tenure systems

Hardin (1968) formalized what many people have learned from experience: that unregulated use of a commons leads to unsustainable exploitation. Such over-use of resources has always been the progenitor of forestry. To be sustainable, forestry must be planned, regulated and practised over ecologically relevant spatial (landscape) and temporal scales (minimum of one tree crop rotation or cycle of stand-replacing natural disturbance; preferably several). In the administrative stage of the development of forestry, regulations lack an ecosystem-level understanding of the ecology of resource renewal and sustainability, and in the end they fail. This leads to ecologically-based forestry which, if linked to an ecosystematic land classification (e.g. BC's biogeoclimatic classification (Table 3.1)), may develop into ecosystem-based management (EBM). EBM differs from true ecosystem management (EM) in that no single agency manages all ecosystem components, structures, processes and values. In theory, EBM can become EM by creating a single management plan for the entire ecosystem in a defined forest area, in which a balance of values is managed by using value trade-off and scenario analyses, allowing for a shifting mosaic of conditions and values across the landscape over time. In reality, there are frequently two major impediments to this transition on public forest land: the tenure system and the failure to use multi-scale, ecosystem-level, and management decision support tools. Forest tenure systems vary widely, but in Canada they are generally limited to timber harvesting tenures only, with non-timber values being a constraint on timber objectives. They almost never license the integrated management of ecosystems for a balance of values that varies over time in any place, and varies from place to place according to the local ecosystem and desired values. Long-term tenures (rotation length or longer) and area-based tenures of ecologically-relevant size have a much higher probability of encouraging stewardship and sustainability of multiple values than short-term tenures and tenures of very restricted spatial extent, but they are not supported by public opinion and often not by environmental groups. Long-term involvement of resource managers within a particular forest area and a single management structure that actively manages all values and their tradeoffs in time and space has a much greater potential to satisfy public expectations for the management of public forests than the restrictive (in time and space and limited to timber management) tenures that generally regulate how public forests are managed today in Canada. Multiple agencies, multiple management/harvesting institutions and no long-term attachment to "place" greatly reduce the possibility of good management compared with alternative institutional arrangements. There are no guarantees of course. Good or bad management can be observed under a variety of tenures — lengths, types and sizes.

3.3.10 Recognizing the role of international trade and economics

Canadian forestry is primarily an export activity — most forest products are traded out of the country, and these products are an important part of our balance of trade and the national economy. The current global economic downturn has reduced the demand for wood products, especially in the US — Canada's major market for forest products (Cashore 1998). This is creating great economic hardship in many forest-dependant communities across Canada (Zhang 2001). "Good" forestry which sustains the diversity of values desired by society is generally more expensive than "bad" forestry. Exploitative harvesting with no management investment is often the most profitable, at least in the short run. It may be difficult to sustain the levels of ecosystem-based forest management that have been achieved in Canada if the economic downturn persists for long. However, promoting the use of certified products, especially by big buyers such as corporations and government contractors, can help to move public opinion towards support for sustainable, ecologically-based forest management.

3.4 How can Canadian forestry respond to these and other issues? One way is ecosystem management modeling

There is no simple answer to this question. It will depend on the issue, the type of forest concerned, and the time and spatial scales at which the issue is addressed. However, it is clear that many of these challenges are social and political rather than solely biophysical (Spies and Duncan 2009). It is unlikely that the solutions offered will be effective unless they are founded on the best available science, but unless biophysical scientists become more successful at transmitting their science in an understandable and policy-relevant manner, science will continue to be more for science sake than for the solution of problems (Kimmins et al. 2005; Kimmins 2008). A major problem of science is that most of it is conducted at scales of complexity, space and time that are far removed from the issues society faces. To better integrate our current scientific knowledge we can combine our knowledge of the past with our understanding of the present ecosystem structures and processes to develop hybrid experience-understanding decision support tools that are able to project possible futures for the variety of forest values desired by society.

A major risk in modeling is dealing with uncertainty. It is not possible to predict the future with certainty, so modeling in forestry finds its major value in the ranking of alternative scenarios and value tradeoffs and considering possible forest futures — necessary forecasts for policy and practice decision making — rather than making firm predictions. Despite the challenges posed by uncertainty, we continue to use models based on the best available knowledge and understanding because decisions have to be made in spite of uncertainty. Ecosystem-level, multi-value management models offer the best way of dealing with ecosystem complexity, and with changing and uncertain futures for which we have no experience. In order to reduce this uncertainty while maintaining simulation credibility, the use of hybrid models is becoming increasingly popular in forest research and management. Following this approach our research team has developed the FORECAST family of ecosystem-level simulation models.

3.4.1 Hybrid simulation models

Modern forest management calls for managing the whole forest ecosystem at stand and landscape scales, and the dominant trend in Canadian forestry is the emulation of the ecosystem consequences of natural disturbances (Perera et al. 2004). This trend supposes a good understanding of the major processes and interactions between ecosystem components. Simulation models can organise the complexity of information and data into a coherent tool for analysing systems at these various scales (Messier et al. 2003). Many management strategies are undertaken at spatial or temporal scales that make replication extremely expensive if not impossible, therefore modeling is a necessary alternative to empirical experimentation. Also, using forecasts derived from mechanistic simulation models allows forest managers to predict the possible impacts of management alternatives without causing potentially negative effects in real forest ecosystems.

Several ecosystem-level models have been developed (Messier et al. 2003) but in this chapter we focus only on one approach. Process-based models use available scientific knowledge to link several ecosystem variables through equations (Korzukhin et al. 1996), but the difficulty in getting the right coefficients to calibrate those equations can result in unrealistic or unreliable predictions. In contrast, statistical models based on field data usually produce good forecasts if the future management and environmental conditions are similar to those of the past. However, they do not have explanatory powers and therefore cannot be used to explore ecological interactions, generate predictions in areas outside the range of the model's experience base, or predict for futures that are expected to be significantly different from the past (Kimmins 2004). To reduce the shortcomings of both these types of models while maintaining their strengths, hybrid models have been developed. Autecology, population ecology, and community ecology are necessary for the understanding of ecosystems, but they are not sufficient for long-term prediction about ecosystem function. Consequently, these hybrid models should be at the ecosystem level. A more detailed analysis of the philosophy behind hybrid predictors is given in Kimmins et al. (1990, 1999).

3.4.2 The basics of the FORECAST approach: Brief description of the model and its simulation capabilities, and extensions to both landscape and individual tree spatial scales

Since the beginnings of our modeling approach (about 35 years ago), the need to be able to address many of the issues discussed above has resulted in the development of several models, branching out from the earliest one, FORCYTE, and its successor FORECAST (Fig. 3.1). FORECAST is a management-oriented, stand-level, non-spatial forest growth and ecosystem dynamics simulator. It was designed to accommodate a wide variety of harvesting and silvicultural systems and natural disturbance regimes in order to compare and contrast their effects on forest productivity, stand dynamics and a series of biophysical indicators of non-timber values. In FORECAST, empirical input data are used as the basis form which to estimate the rate at which key ecosystem processes (e.g. efficiency of light capture, nutrient cycling, and nutritional regulation of growth) must have operated to produce observed trends in ecosystem productivity and biomass accumulation (see Kimmins et al. 1999, Seely et al. 1999 for further details). These data are entered into "setup" input files and then are processed by the "setup" programs to create the simulation rules and estimates of process rates used to drive the mechanistic, process-based ecosystem-level component of the model. They include (but are not limited to): (i) photosynthetic efficiency per unit of foliage nitrogen based on relationships between foliage nitrogen, simulated self-shading, and net primary productivity after accounting for litterfall and mortality; (ii) nutrient uptake requirements based on rates of biomass accumulation and literature- or field-based measures of nutrient concentrations in different biomass components on different site qualities; (iii) light-related measures of tree and branch mortality derived from canopy structure input data in combination with simulated light profiles; and (iv) competition among simulated species for resources. The inclusion of these processes provides FORECAST with the capacity to address many of the modeling capabilities discussed above. In addition, in order to address the challenge of simulating climate change effects on forests, direct representation of climate (temperature and water balance) is also included in the version FORECAST-Climate, which includes a new module to allow for simulation of climate change effects on ecosystem processes and values. In order to address different issues at different scales, we have extended FORECAST into several additional applications (Fig. 3.1).

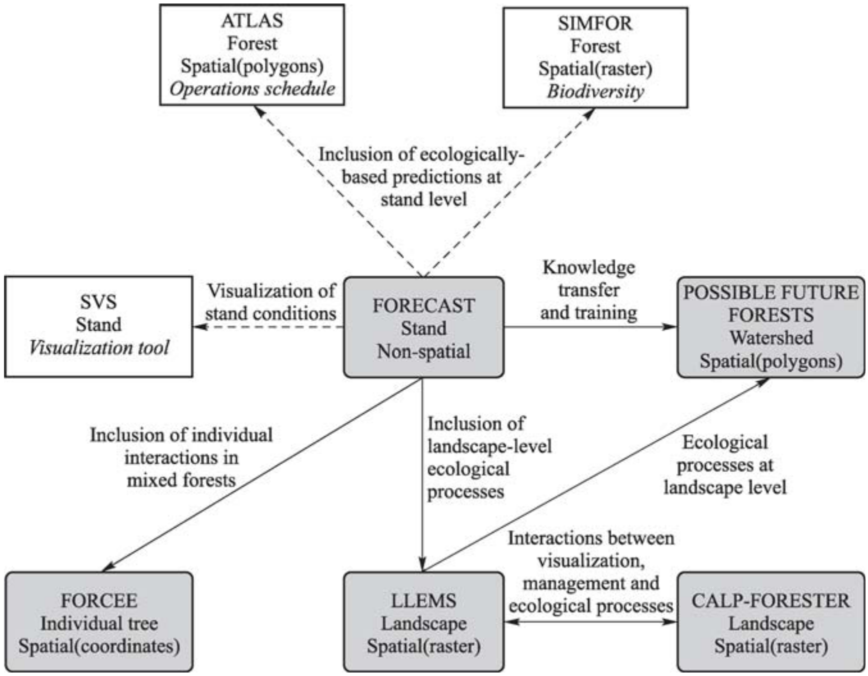

Fig. 3.1 The different models developed by the Forest Ecosystem Simulation Research Group at UBC (boxes with dark background) were developed from FORECAST in order to address the issues indicated by the arrows, and FORECAST output can also be linked to other models.

Ecosystem-level, stand models cannot address all the issues of modern forestry. Landscape-level sustainability and the spatial configuration of ecological processes and the effects of forest management thereon are becoming increasingly important in forest management and conservation. In response, FORECAST has been extended to a spatially-explicit landscape model. This can simulate plant development and ecosystem processes in user-defined interacting grid cells (that can be as small as 10 m $\times$ 10 m) within a framework that can accommodate up to 2 million cells (for a total area of 2,000 ha for 10 m $\times$ 10 m cells). In this Local Landscape Ecosystem Management Simulator (LLEMS), cells are clustered into polygons at the start of a run on the basis of a series of attributes (vegetation structure, density, species composition, age, soil condition, and others). This greatly increases the speed of the simulation and permits a high degree of ecosystem process simulation to be applied across the landscape. As the simulation proceeds, individual cells may get transferred to other polygons as the developing vegetation changes light and soil conditions, and as ingress of understory species and tree regeneration changes the plant community. Management actions such as harvesting, planting or fertilization also cause a subdivision of affected polygons to maintain them within user-set levels of heterogeneity. Cells are updated annually or on shorter time steps. This approach permits very detailed spatial process ("bottom-up") simulation over relatively large areas ("top-down") as well as maintaining simulation flexibility in the face of management or natural disturbance. Natural regeneration can be simulated as a consequence of seed production, dispersal and wind effects within and between polygons. Management actions such as harvesting, planting or fertilization also cause a subdivision of affected polygons to maintain them within user-set levels of heterogeneity.

As additional values such as visual quality of landscapes and inclusion of public input into management plans become increasingly important, we have created CALP-Forester as an interface for LLEMS (Fig. 3.2). This tool provides visual output to accompany ecologically-based predictions, and it can facilitate the involvement of stakeholders in management planning by making it easier to visualize possible forest future conditions under different management regimes. LLEMS is well suited to many forestry applications in ecosystem management since it can represent trees, shrubs, herbs (and bryophytes if needed), the independent management of each of these plant life forms and species within life form, site-level management treatments, and the actions of herbivorous animals. It can represent the interactions between trees and understory, and the ecosystem effects of fire, wind or insect epidemic. LLEMS lends itself well to landscape pattern analysis, including carbon budgets or issues of fragmentation and connectivity of wildlife habitats. LLEMS is linked to an interactive 3-D visualization of landscapes of up to 2,000 ha (larger if cell size is increased) with which to communicate the outcomes of the simulation (Fraser et al. 2007).

LLEMS has been developed to address landscape-level issues that are becoming important in sustainable forest management, such as wildlife management and alternatives to clear cutting. Variable retention management (Franklin et al. 2002) is becoming an increasingly popular method employed by forest managers to address non-timber management objectives. The re-

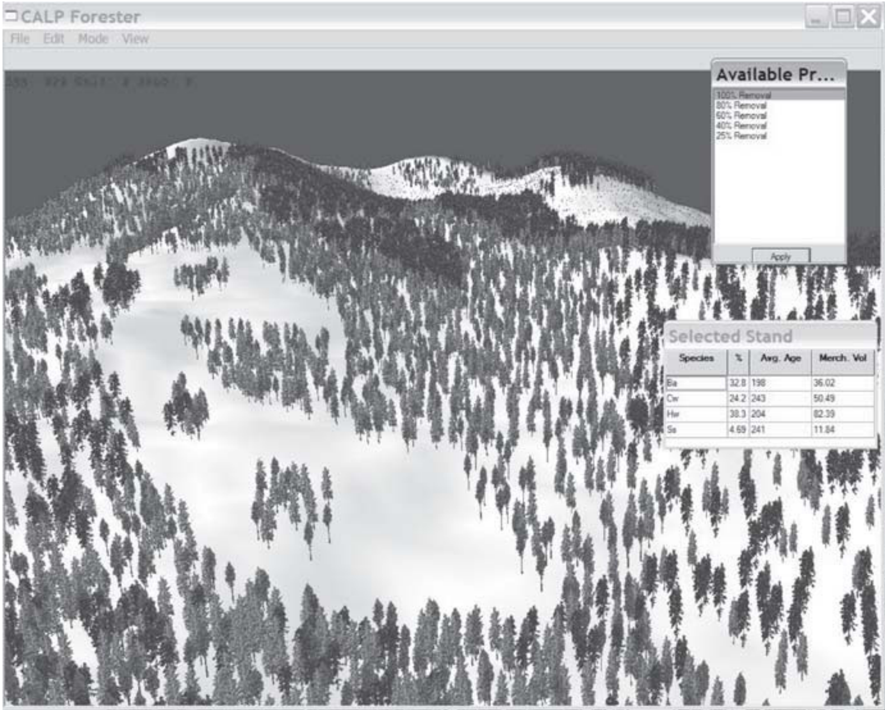

Fig. 3.2 Interactive visualization of LLEMS output: CALP-Forester.

tention of individual trees or groups of trees within a block is intended to maintain structural complexity, to provide habitat for wildlife and to reduce the negative aesthetic impact of timber harvesting (Burton et al. 1999). While it has shown promise in achieving these goals in the short term, the long-term implications of variable retention are largely unknown, and models are needed that can represent the ecological foundations of this new system. Seely (2005) used LLEMS in the coast of British Columbia to study the impact of different levels of variable retention on traditional growth and yield variables (height, volume, biomass) and on several indices of habitat suitability (understory cover, below canopy light levels, number of snags and decomposing logs), demonstrating the capabilities of LLEMS for landscape-level simulation. This use of LLEMS as a decision-support tool for wildlife management has been upgraded with a new module to assess the impacts of management activities at the large cutblock or watershed scale on spatial and temporal patterns of wildlife habitat supply (Seely et al. 2008).

Another important issue in forestry that we have already commented on is the connection between economics, ecology and natural resources values. In order to explore this interaction, we have developed a watershed model, Possible Forest Futures (PFF), designed to simulate small watershed issues that involve the stand-level but with spatially-explicit representation of the interactions of stands. Such issues in landscape ecology include road construction, harvesting schedules, landscape pattern and riparian forest management, among others. Similar to LLEMS in terms of polygon structure and interaction (but using polygons rather than pixels and minus the detailed light profiling), PFF includes a hydrology model and can track road development. The model also includes extensive output concerning economic costs and benefits, productivity, carbon budgets and values of other social and environmental variables important in modern forestry, the variation of which over time can be examined graphically for individual stands and for the entire watershed. PFF can prepare rotation-length movies of different landscape scenarios for subsequent analyses. Because it is an ecosystem management model that can simulate most aspects of landscape-level issues in forest management, PFF

Table 3.2 Some applications of FORECAST for different forest management issues at different scales.

Research area	Temporal and spatial scale	Ecsosystem type	References
Soil organic matter as indicator of sustainability of forest management	Multi-rotation, stand-level	Coastal Douglas-fir forests	Morris et al. (1997) Seely et al. (2002)
Assessment of the two-pass harvesting system	Stand-level, single rotation	Boreal mixedwoods	Welham (2002)
Sustain or improvement of long-term productivity	Stand-level, multiple rotation	Sub-boreal lodgepole pine	Wei et al. (2003)
Assessment of multi-objective management strategies	Landscape-level, multiple rotations	Boreal mixedwoods	Seely et al. (2004)
Links between different model approaches	Landscape-level, multiple rotations	Boreal mixedwoods	Seely et al. (2004)
Study of yield decline and tree-understory interactions	Stand-level, multiple rotations	Sub-tropical Chinese-fir plantations	Bi et al. (2007)
Productivity across multiple short rotations	Stand-level, multiple rotations	Hybrid poplar plantations	Welham et al. (2007)
Site-specific validation of FORECAST	Stand-level, single rotations	Coastal Douglas-fir plantation	Blanco et al. (2007)
Regional validation of FORECAST	Landscape-level, single rotation	Sub-boreal mixedwoods	Seely et al. (2008)
Complexity needed in ecologically-based management models	Landscape-level, multiple rotations	Sub-tropical plantations and sub-boreal forests	Kimmins et al. (2008)
Landscape effects of forest management for bioenergy	Landscape-level, single rotation	Sub-boreal mixed and planted forests	Flanders et al. (2009)

can also be used to examine land use patterns that include mixtures of forest management, forest reserves and other land uses such as agroforestry and agriculture, and because it is an ecosystem management model, it can simulate most aspects of landscape-level issues in forest management at the small to medium watershed scales.

Continuous-forest-cover forestry and stands with complex vertical and horizontal structure have been embraced by the public as the right way to manage forests. This is a response to the aesthetic and other consequences and to the even-age structure of stands resulting from clear-cut or shelterwood silvicultural systems. While low disturbance systems fall within the natural range of variation for some Canadian forest types, they are not characteristic of many forests that have developed with stand replacing natural disturbance in which even-age and monoculture are natural, sometimes temporary but sometimes persistent conditions. Models are needed urgently that can explore the consequences of accepting this public pressure and changing the fundamental disturbance ecology of many forests. Of necessity, such models must represent the key ecosystem processes that are being altered. In response to this need we have developed FORCEE, an individual tree, spatially explicit model, in which the spatial coordinates are known for each individual tree, and any configuration of tree distribution and density can be represented. The model simulates nutrient cycling, light profiles, and patterns of litterfall for each tree, and their effects upon growth of adjacent trees and understory. Rules for the simulation of plant growth and interactions are derived from the FORECAST model and applied to individual trees with additional input data on individual plant dimensions in different competitive environments. As a derivative of FORECAST, FORCEE can examine the limiting factors of light, water and nutrients.

3.5 Conclusions

Forestry in Canada has advanced from unregulated exploitation, through timber-focused administrative forestry, to ecosystem-based forestry in a remarkably short period, considering the size, the ecological diversity of its forests and its social and political diversity. Change in forestry comes slowly because the political and administrative structures and legislation controlling it and the investment structure financing it have an inertia that takes time to adjust. In the minds of the public, change is even slower than it really has been because the visual consequences of past policies and practices are very persistent on the landscape. In fact, under pressure from the public (that built on frequently unrecognized earlier work by academics, researchers and foresters conducted before there was an active environmental movement in Canada), change has been active for at least three decades. Forestry in Canada today is amongst the best in the world, the most rooted in ecosystem sciences, and the most regulated relative to the size and diversity of the country. Many challenges remain, and our forestry remains a significant distance from what it could be. In order to assist all the sectors involved in development of landscape-level ecosystem management, appropriate modeling tools must be developed, tested and used. Given the ecological diversity as well as the complexity of forest ecosystems, diverse forest modeling approaches are needed to address the diverse questions of forest management in Canada, and linking them together will be one of the challenges for future ecological research.

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Part II
Modeling Disturbance and Succession in Forest Landscapes

Chapter 4 Challenges and Needs in Fire Management: A Landscape Simulation Modeling Perspective

Robert E. Keane*, Geoffrey J. Cary and Mike D. Flannigan

Abstract

Fire management will face many challenges in the future from global climate change to protecting people, communities, and values at risk. Simulation modeling will be a vital tool for addressing these challenges but the next generation of simulation models must be spatially explicit to address critical landscape ecology relationships and they must use mechanistic approaches to model novel climates. This chapter summarizes important issues that will be critical for wildland fire management in the future and then identifies the role that simulation modeling can have in tackling these issues. The challenges of simulation modeling include: (i) spatial representation, (ii) uncertainty, (iii) complexity, (iv) parameterization, (v) initialization, (vi) testing and validation. The LANDFIRE project is presented as an example on how simulation modeling is used to support current fire management issues. Research and management needs for successful wildland fire-related simulation modeling projects will need (i) extensive mechanistic research programs, (ii) comprehensive databases, (iii) statistical validation methods and protocols, (iv) software and hardware research, (v) modeling science explorations, and (vi) extensive training. Models will continue to play an integral role in fire management but only if the science keeps pace and managers are poised to take advantage of advances in modeling.

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Keywords
Ecological modeling, spatial dynamics, landscape ecology, mechanistic simulation, parameterization.

4.1 Introduction

Fire management in the United States faces a number of challenges in the next century. Seventy years of fire exclusion policies implemented under successful fire suppression programs have resulted in areas with increased canopy and surface fuels, especially those that historically experienced frequent fire, that now require extensive fuel treatments to reduce fire hazard and restore ecosystems (Ferry et al. 1995). In some areas, extractive land management practices, such as grazing and timber harvest, along with exotic species invasions have created novel ecosystem and fuels characteristics that may also require innovative pro-active fuel and ecosystem treatments. Meanwhile, human development is expanding into the nation's wildland extending the wildland urban interface thereby making fire fighting difficult, heightening the risk of loss of property or life from wildfire, and increasing the need for intensive fuel treatments (Radeloff et al. 2005). Fire suppression costs are spiraling upwards, along with the economic and social costs of treating fuels to reduce high fire severity. New fuel treatment technologies, such as mastication, are finding favor in fire management because they are less risky, easier, and cheaper to implement, but their impacts on ecosystems remain unknown (Agee and Skinner 2005). Above all, future climates are predicted to be warmer and drier resulting in substantial increases in fire size, severity, intensity, and frequency (Cary 2002, Running 2006, Westerling et al. 2006). Curiously, these same fire-prone forests are being proposed for storage of carbon from the atmosphere even though they will probably burn long before they can be effective carbon sinks (Sampson and Clark 1995, Tilman et al. 2000). Fire management will need to develop new policies, strategies, and tools to meet these future challenges and ensure the sustained health of US landscapes (GAO 2007).

Simulation modeling will be one of the most important tools for fire management in the challenging future by providing an effective, standard, and objective context to evaluate management actions and ecological change (Lauenroth et al. 1998). Models can be used to simulate effects of alternative treatments to determine the most effective fuel reduction or ecosystem restoration strategy (Miller 2000). Novel treatments can be simulated to determine resultant short- and long-term effects on a diverse array of ecosystem elements (Ryu et al. 2006). Fire hazard and risk can be simulated to prioritize areas for treatment and to design the most effective treatment prescriptions (Keane et al. 2008). Simulation can also be used to approximate historical landscape conditions that can then be used as reference for ecologically based landscape prioritization and planning (Wimberly et al. 2000). Predictive landscape models can also be used to update broad-scale digital maps and design future sampling strategies for assessing change. Fire behavior and effects models can be integrated to simulate wildfire spread and resultant fire severity to determine if wildfires are providing ecological benefits (Keane and Karau, 2010). Mechanistic landscape models can be used to explore fire, climate, and vegetation interactions and to quantify fire regimes in space and time (Keane et al. 2003, Neilson et al. 2005). Most importantly, mechanistic simulation models can help predict potential fire dynamics in future climates to provide fire management critical information to mitigate adverse effects (McKenzie et al. 2011).

This chapter discusses the challenges of using simulation modeling in fire management. An introduction to simulation modeling is presented that will lay the groundwork for understanding most material in this chapter. Then, the challenges of using simulation in fire management applications are discussed and an example of the use of simulation in fire management is presented. Last, research needs, future direction, and possible solutions are summarized to ensure that simulation modeling becomes an effective fire management tool in the future because traditional fire management approaches will be severely tested in a warmer climate (Flannigan et al. 2008).

4.2 Simulation modeling in fire management

Simulation is an important tool to predict fire behavior and effects for a wide diversity of fire management applications. However, the terminology used to describe fire models can be confusing and inconsistent.

4.2.1 A simulation modeling primer

Three types of variables are generally used in models: *state* variables describe the central entities being simulated, *flux* variables represent the processes that change the state variables, and *intermediate* variables are used to compute flux variables. For example, tree leaf carbon might be a state variable and a flux variable would be the carbon lost each year to leaf fall. In general, four tasks are involved in simulation modeling. *Initialization* involves assigning starting values to the state variables. *Parameterization* concerns quantifying those model parameters that are used in algorithms that compute state, flux, and intermediate variables. *Validation* entails testing the model to ensure that the results are realistic and quantifying the accuracy of results to estimate uncertainty (Rykiel 1996). Lastly, *sensitivity analysis* involves modifying parameters and algorithms to determine their influence on model results (Cariboni et al. 2007).

Model approaches can be described as empirical, mechanistic, stochastic, and deterministic. *Empirical* models are created from extensive data often using statistical modeling techniques. Examples include the Australian fire behavior model (McArthur 1967), the CONSUME fire effects model (Ottmar et al. 1993), and the growth and yield model FVS-FFE coupled to a fire and fuels extension (Beukema et al. 1997). Empirical models are accurate but limited in application to the conditions represented by the data. *Mechanistic* models simulate biophysical processes using universal physical and chemical relationships, therefore mechanistic models are applicable to a wide range of domains, but they are often complex, which often results in instability, inaccuracy, and difficulty in parameterization. *Stochastic* models contain numerical relationships that use probability distributions, which often require repeated runs to quantify the variability in results. *Deterministic* models contain mathematical equations that represent important processes resulting in output that often does not vary for a particular set of inputs. In reality, most models contain a diverse mixture of these four approaches, especially long-term landscape models. For example, a landscape fire succession model can simulate fire using a mechanistic function, seed dispersal using stochastic algorithm, tree growth using mechanistic biophysical equations, and fire effects using deterministic decision trees (Keane et al. 2004).

Two groups of simulation models are used in fire management. *Fire behavior models* simulate the physical combustion processes of wildland fire such as spatial growth, rate of spread, fireline intensity, and flame length. These are strategic models for real-time, operational use under wildfire conditions or planning applications to describe fire hazard. Examples of these models include the mechanistic fire model of Rothermel (1972) that is implemented into the BEHAVE software for point evaluations and into FARSITE for spatial applications, and the empirical McArthur (1967) Australian fire spread model. *Fire effects models* simulate direct and indirect effects of fire on ecosystems (Reinhardt et al. 2001). Direct or first order fire effects include fuel consumption, tree mortaility, and smoke production (Ottmar et al. 1993, Reinhardt et al. 1997), while second order or indirect effects include vegetation development, erosion, and fire regimes (He et al. 2008). Fire effects models include all those ecological simulation models that contain any representation of wildland fire from the stand-level gap models that simulate individual tree growth, mortality, and regeneration to the landscape fire succession models that simulate ecological processes, such as fire regime, in a spatial domain (Keane et al. 2004). This chapter mainly covers landscape level fire effects simulation modeling which nearly always contains embedded fire behavior models.

4.3 Technical challenges in fire management modeling

There are assorted difficulties and dilemmas encountered by modelers as they build various computer programs for fire management that span from how to design a model to how to use the model correctly.

4.3.1 Model design

The chief challenge facing modelers is to build fire simulation models using a mechanistic approach such that causal processes are linked to ecosystem responses so that new, unforeseen results can be generated (Pacala and Tilman 1994, Rastetter et al. 2003). Properly designed mechanistic models are quite robust in terms of scope and application so that their simulated consequences, such as responses to climate change, become emergent properties of the model rather than predetermined results generated as a consequence of parameterization (Peng 2000). The major challenge is to design robust fire models around detailed algorithms that use flux variables to represent important physical relationships and interactions between dynamic, readily quantifiable inputs. Unfortunately, research has quantied a fraction of the major physical relationships in a handful of ecosystems, so simulation design compromises are always made to account for the limited state of knowledge (Keane et al. 2010). Furthermore, those processes with intrinsic uncertainty, such as fire ignition, may always need to be modeled stochastically because of their inherent complexity and cross-scale influences. It is important that modelers identify the plausible extent of mechanistic design using available literature and existing models and explicitly recognize these bounds in the results.

Future fire models must also be designed to be spatially explicit to address complex scale issues (Peters et al. 2004). This means that the modelers must explicitly incorporate spatial relationships in model design and implementation. One-dimensional (1D) or point models, such as BEHAVE (Andrews and Bevins 1999) and FOFEM (Reinhardt et al. 1997), may have limited use in the future because they can represent fire behavior at only one scale. Future fire behavior models, especially research-oriented models, must be multi-dimensional in space and time to ensure that those processes that occur at one scale and location are affecting processes that occur at other scales and locations (Gardner et al. 1991). This approach has many obstacles for implementation including lack of sufficient data across appropriate scales, high demand for computer resources, identification of the appropriate scales for simulation (e.g., selecting the right pixel size), spatial autocorrelation in fire activity (Magnussen 2008), and specification of the proper spatial extent. However, explorations of spatial interactions are the only way to comprehensively simulate fire behavior and effects across landscapes (Keane et al. 2010).

It is important that the spatial scales represented in the model are reconciled to the ecosystem processes that they represent and to the time scales at which these processes operate (Waring and Running 1998). Tree regeneration dynamics, for example, may require a spatially explicit, annual seed dispersal model and a simulation of reproduction phenology at a daily time step to properly reflect climate interactions of tree life history.

Another challenge is that future simulation models must be able to simulate the complex interactions of state and flux variables across scales (Rastetter et al. 2003, Urban 2005). Dynamic feedbacks and cross-scale interactions will allow the prediction of novel ecosystem responses and interactions between climate, vegetation, and disturbance which are likely to lead to nonlinear model behavior and cause important phase transitions that are critical for landscape management (McKenzie et al. 2011[in press]). The trend and magnitude of these interactions are mostly unknown for many ecosystems and they are difficult to study outside of a simulation approach. One important interaction is the role that humans play in past (e.g., Native American burning), present (fire exclusion era), and future (enlightened fire management) on landscape dynamics (Kay 2007). Interactions can dictate important thresholds and phase transitions of landscapes in changing climates so that management can anticipate these changes and respond (Allen 2007). Also important are how multiple factor interactions, such as multiple disturbances, create novel landscape conditions that may accelerate landscapes toward important thresholds and phase transitions. Ecosystem science has only scratched the surface in determining the sign and amplitude of most ecological interactions largely because of the complexity in the nested scales of time and space involved (Allen and Starr 1982, King and Pimm 1983).

One last challenge is balancing complexity with utility in model design. In general, simulations are more difficult to conduct as model complexity increases because with complexity come additional parameterization, detailed initializations, higher computing demands, and complicated model behavior. Developing a parsimonious list of important variables to model is critical to efficiently simulating ecological processes, otherwise a model can become overly complex and difficult to parameterize because of lack of information. It is also easy to oversimplify model design such that simulation results are meaningless. Conversely, if too much detail is included, the intrinsic uncertainties associated with each modeled process may compound to produce equally meaningless results (Rastetter et al. 1991, McKenzie et al. 1996). Moreover, managers may not have the time, expertise, and resources to operate and interpret highly complex fire models. Therefore, it is critical that simulation design is properly matched to the level of information required by managers and this is effectively accomplished by plainly stating the objectives of the simulation effort.

4.3.2 Model use

One of the greatest challenges in modeling is to clearly articulate simulation objectives to inform the simulation project. While seemingly obvious, this is easily the least understood concept in the design and implementation of fire models. Without an explicit statement of simulation objectives, it is problematic, and perhaps impossible, to build a comprehensive model that provides easily understandable results to address research and management concerns. A clear modeling objective allows the modeler to easily identify the (i) variables to include in the model structure, (ii) sequence of simulation for selected variables, (iii) input and output file structures, (iv) critical ecosystem processes to simulate, (v) important interactions to include, (vi) time steps to implement, and (vii) appropriate spatial and temporal resolutions and extent. It is important to state this objective so that the most appropriate models are selected or built, the right parameters are selected or quantified, the simulations are successfully completed in an acceptable time, and the results are easily understood. While, in general, additional objectives can be explored as the complexity of mechanistic models increases, it is unlikely that there will be a über-model that addresses all objectives because there will never be sufficient science to support its development or computing resources to conduct the simulation. Therefore, it will always be imperative to focus model development with a clear simulation objective.

It seems logical that fire simulation models will be much more complex in the future, and this will demand increased computer resources, higher expertise in model use, and more extensive parameterization. Complex models are critically needed because it is nearly intractable to design wildland fire experiments that explore the dynamic relationships of fire and ecosystem behavior over multiple time and space scales. Crown fires, for example, are extremely difficult to study using empirical approaches because it is difficult and costly to measure heat flux across the large spatial scales involved using contemporary experimental equipment (Albini 1999). The long temporal scales involved in exploring dynamic fire regimes may preclude short-term answers from intensive field surveys, which are undoubtedly invaluable in the longer term. However, it is unlikely that the fire management would adopt these new complicated models or necessarily afford the computers needed to run these models in an operational application. Therefore, the challenge will be to develop complex spatially explicit fire models for research purposes and then synthesize them and their results to create management-oriented models that may not be as robust as expected but will perform well in operational applications because they are easy to parameterize, execute, and understand (Keane and Finney 2003).

The input of climate into simulation models is an increasing challenge facing modelers and model users in the future (Keane et al. 2010). It is important that models have an explicit representation of climate across multiple scales to ensure a realistic response of ecosystem dynamics to climate. Phenology, for example, may need a daily climate time step at 100-m resolution, whereas decomposition may need monthly time steps at 1-km resolution (Edmonds 1991, White et al. 1997). Identification of the appropriate hierarchical scales will be difficult, but the task of matching the ecosystem processes of disturbance and plant dynamics to the appropriate climate scales may be even more challenging. Identifying the most parsimonious climate data stream to input into models is also crucial given that too much weather data could potentially complicate and slow fire model simulations. This is inherently difficult because each weather variable has an intrinsic scale (e.g., microsite, regional) and resolution (e.g., vertical layers above ground, grid size). Multiple weather streams representing past, present, and future projections of climate are needed to determine potential climate effects on disturbance and vegetation. And, multiple climate scenarios are needed to bracket the range of potential effects and to identify important thresholds of ecosystem change. An explicit simulation of atmospheric transport is also desirable for fire models. Wind speed and direction at various heights can feed disturbance processes (e.g., windthrow, fire spotting, insect epidemics) and plant dynamics (e.g., seed dispersal) (Greene and Johnson 1995). Atmospheric transport can also be used to simulate important feedbacks such as smoke dispersal, atmospheric deposition, and radiation budgets.

A last challenge is quantifying the uncertainty involved in fire simulations so that fire managers and researchers fully understand the impact and significance of the predictions and results (Bunnell 1989, Araujo et al. 2005). This includes developing methods to present simulation results that contain an estimate of error or degree of uncertainty (Bart 1995). This assessment of uncertainty should account for the error in parameterization, initialization, and model algorithms, as well as the error and variability in model predictions (Brown and Kulasiri 1996). The IPCC (2007) report contains protocol and classification that they propose that all modelers use to describe the uncertainty assessment for modelers. Results must also be synthesized into variables and formats that are commonly employed by fire management.

4.4 A fire management simulation example

An example of how diverse simulation modeling approaches can be integrated together to create a viable management tool is presented to illustrate the use of landscape modeling in fire management.

4.4.1 The LANDFIRE mapping project

The US Healthy Forest Restoration Act and the National Fire Plan's Cohesive Strategy established a national commitment to reduce fire hazard and restore fire to those ecosystems where it had been excluded for decades (Laverty and Williams 2000). This commitment required detailed multi-scale spatial data for prioritizing, planning, and designing fuel reduction and ecosystem restoration treatments across the entire nation (GAO 2007). These spatial data layers must provide essential fuel, fire regime, and vegetation information critical for designing treatments and activities at spatial scales compatible with effective land management (Hann and Bunnell 2001). The Fire Regime Condition Class (FRCC, an ordinal index with three categories that describe how far the current landscape has departed from historical conditions) has been identified as one of the primary metrics to be used for distributing resources and prioritizing treatment areas to protect homes, save lives, and restore declining fire-adapted ecosystems (Hann 2004). The LANDFIRE project was initiated in 2005 to create a scientifically credible and ecologically meaningful national map of FRCC, along with developing a number of supporting maps of vegetation, fuels, and biophysical settings, at 30 m pixel resolution that could be used across multiple organizational scales. This project integrated mechanistic statistical modeling with landscape fire succession simulation to create the desired products to serve as an example of how fire management might solve the current challenges mentioned above.

All LANDFIRE methods and protocols were based on a data-driven, empirical approach where the majority of mapped and simulated entities were created from complex spatially explicit mechanistically based statistical modeling (Keane et al. 2007) because managers required the LANDFIRE products to be scientifically credible, repeatable, and accurate with a minimum of subjectivity. To meet this challenge, the LANDFIRE reference database was created by collecting georeferenced data from thousands of plots obtained from a variety of sources, most importantly, the USDA Forest Service Forest Inventory and Analysis program (Caratti 2006) (Table 4.1, Table 4.2). These data were used for (i) developing training sites for imagery classification, (ii) parameterizing, validating, and testing simulation models, (iii) developing vegetation classifications, (iv) creating statistical models, (v) determining data layer attributes, (vi) describing mapped categories, and (vii) assessing the accuracy of maps, models, and classifications (Rollins and Frame 2006). The concept of historical range and variability (HRV) was used as the premise of all FRCC calculations (Landres et al. 1999; Swetnam et al. 1999). HRV was defined as the quantification of temporal and spatial fluctuations of landscape composition (portion of area by each vegetation map unit) prior to western European-American settlement (Hann and Bunnell 2001). Historical landscapes were then compared to current landscape composition to compute FRCC (Hann 2004).

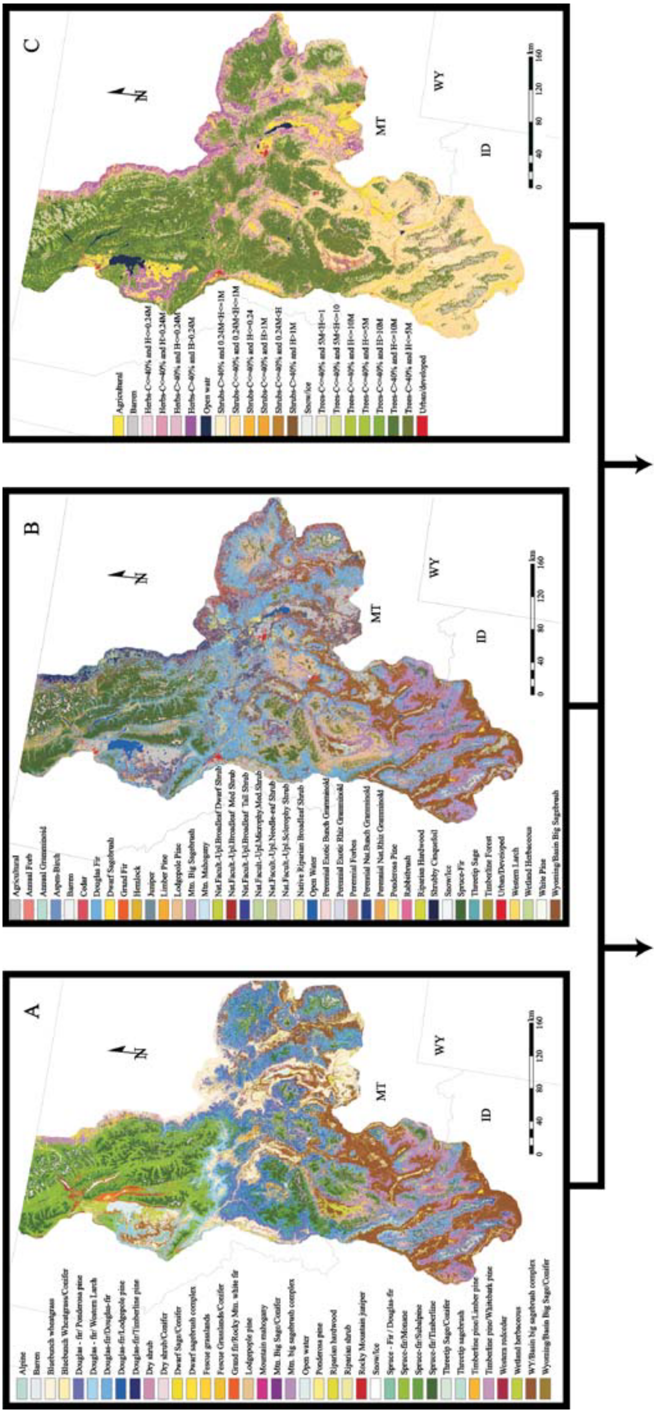
Table 4.1 The linked models used in the LANDFIRE project.

Model	Description	Purpose	Data	Sources
LANDSUM	A landscape succession model	Generate HRV time series and map fire regimes	Vegetation studies, fire history studies	Keane et al. (2002), Keane et al. (2006b)
WXFIRE	A biophysical weather extrapolation model	Extrapolate coarse scale gridded weather to 30 meters, compute climate variables for vegetation mapping	CLIMET gridded database, Digital elevation models	Keane and Holsinger (2006)
BGC	A biogeochemical ecosystem model	Compute ecosystem process variables for vegetation mapping	WXFIRE, CLIMET gridded database	Thornton (1998), Thornton et al. (2002)
HRVSTAT	A statistical analysis program	Computing FRCC from HRV-current comparison	LANDSUM, LANDFIRE maps	Steele et al. (2006)
FIREMON	A database and analysis system for data management	Create the LANDFIRE database	Legacy data from university, government and private agencies	Lutes et al. (2006)
DAYMET	A model to create gridded daily weather across the US	Create the DAYMET database for use in simulation modeling	Weather station data collected throughout the US	Thornton et al. (1997)
See5	Statistical algorithms for regression tree empirical modeling	Create vegetation and fuels maps	Simulated data from all models	Quinlan (2000)

The LANDFIRE prototype project developed the methods and protocols used to map FRCC across the United States using a complex integration of several ecological models (Table 4.2) (Rollins et al. 2006). The historical spatial time series that represented HRV were created from landscape simulation modeling since historical maps and data are absent for much of the US; the LANDSUM model was used to simulate landscape dynamics and output landscape composition over 5,000-year HRV simulations (Keane et al. 2006b; Pratt et al. 2006). Historical fire regime and vegetation succession field data collected from numerous studies were used to parameterize LANDSUM (Long et al. 2006). LANDSUM stratifies these parameters by three vegetation-based classifications: (i) Potential Vegetation Type (PVT) defined by biophysical settings, (ii) cover types described by dominant vegetation, and (iii) structural stage described by vertical stand structure. The PVT approximates biophysical setting by assuming that the unique "climax" vegetation community that would eventually develop in the absence of disturbance can be used to identify unique environmental conditions (Daubenmire 1966). The LANDFIRE PVT classification is a biophysically based site classification that uses plant species names as indicators of unique environmental conditions (Holsinger et al. 2006). Cover types were named for the species with plurality of canopy cover or basal area, while structural stage was based on canopy cover and height (Zhu et al. 2006).

Table 4.2 Flow of LANDFIRE tasks to create the fire regime condition class (FRCC) and various fire management projects that use LANDFIRE data (Models are defined in Table 4.1).

Data	Task	Models
	Flow of LANDFIRE tasks	
FIA data, research data, legacy data	Compile LANDFIRE database (Caratti 2006)	FIREMON
LANDFIRE database, Simulated outputs,	Build PVT map (Holsinger et al. 2006)	WXFIRE, BGC, See5
LANDFIRE database, simulated outputs, satellite imagery	Create current cover type and structural stage maps (Zhu et al. 2006)	WXFIRE, BGC, See5
LANDFIRE database, literature, PVT map, cover type map, structural stage map	Develop ancillary fuels data layers (Keane et al. 2006a)	WXFIRE, BGC, See5
LANDFIRE database, literature, PVT map, cover type map, structural stage map, NIFMID database	Simulate HRV historical time series (Pratt et al. 2006)	LANDSUM
LANDSUM output, PVT map, cover type map, structural stage map	Compute departure (Pratt et al. 2006)	HRVSTAT
Departure estimates	Compute FRCC (Steele et al. 2006)	HRVSTAT
	Use of LANDFIRE data	
LANDFIRE fuels maps	Compute fire hazard and risk (Keane et al. 2010, keane and karau 2010)	FIREHARM
LANDFIRE fuels and vegetation maps	Compute potential fire severity (Karau and Keane 2010[in prep])	FLEAT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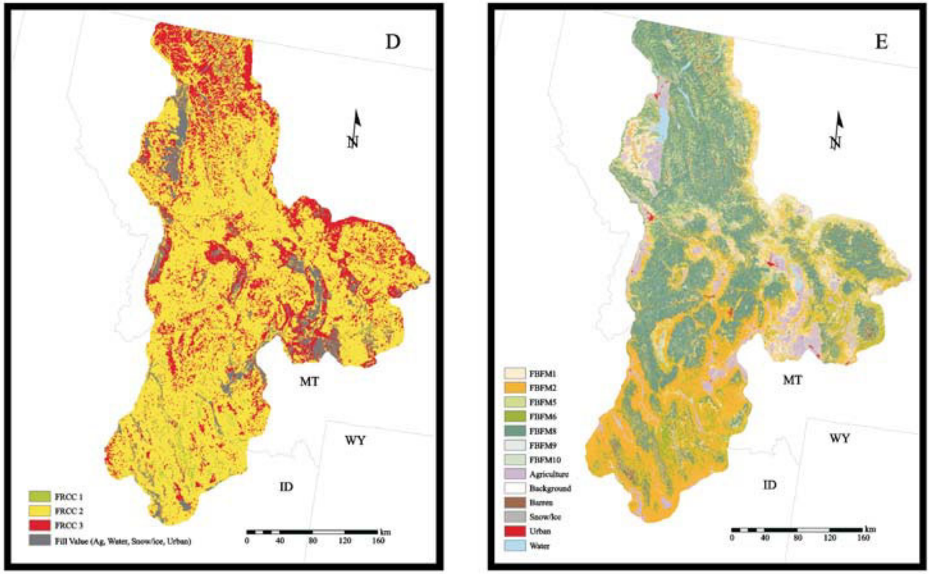

Fig. 4.1 Critical maps created by the LANDFIRE project to compute FRCC for the northern Rockies mapping zone 19. A, Potential vegetation type map; B, cover type map; C, structural stage map; D, FRCC map; E, fire behavior fuel model map.

Vegetation maps portraying current conditions were then needed to compare with the simulated HRV time series to compute FRCC (Table 4.2). The three classifications used in the LANDSUM modeling (PVT, cover type, structural stage) were mapped to describe current conditions so that simulated HRV data matched existing conditions. The PVT map (Fig. 4.1A) was created using a gradient modeling approach where plot-based assessments of PVT were modeled from a plethora of biophysical variables that were summarized from field data using regression tree (CART) statistical techniques (See5 model (Quinlan 2000)). The biophysical variables were computed from the biophysical WXFIRE model (Keane and Holsinger 2006), simulated using the BGC model (Holsinger et al. 2006), or taken from the DAYMET gridded weather database (Thornton et al. 1997) (Table 4.2). Resultant empirical CART models were then used to map PVT across the region using independent variables created from the same models (Holsinger et al. 2006). The biophysical PVT map was subsequently used with other biophysical variables and Landsat 7 Thematic Mapper satellite imagery to create the cover type and structure stage maps using a gradient modeling approach (Zhu et al. 2006) (Fig. 4.1B,C).

Two methods were used to calculate the departure statistic that quantitatively compares the existing condition to the many historical landscape compositions. Steele et al. (2006) developed the HRVSTAT model that computed departure and statistical significance using advanced regression techniques and Pratt et al. (2006) used a variation of the Sorenson's index based on management-oriented FRCC methods (Hann 2004, Barrett et al. 2006) (Fig. 4.1D). Both departure indexes ranged from zero to 100 with 100 being the most departed. FRCC was finally created by making classes of the departure statistic (Pratt et al. 2006).

Some products created from the LANDFIRE process (e.g., biophysical variables) were then used to create additional fuels and fire regime layers that are critical in the eventual planning and implementation of fuel and restoration treatments at local scales (Fig. 4.1E). Fire regimes were taken from the LANDSUM model (Pratt et al. 2006) and described fire return interval and probability of three fire severity types. Canopy fuels maps were created using the gradient modeling approach where the same independent variables used for vegetation mapping were correlated to plot level fuels characteristics (Keane et al. 2006a). Surface fuel models were assigned to combinations of categories in the three vegetation classifications (Fig. 4.1E). LANDFIRE products have been extensively used by fire management for a variety of applications. Fire hazard and risk maps have been created for large regions using the FIREHARM program (Hessburg et al. 2007; Keane et al. 2008). The FLEAT program is being used to estimate fire severity and quantify ecological benefits from wildfire using HRV simulations using LANDFIRE input data (Karau and Keane 2010 [in press]). The LANDFIRE fuels data layers are being used as inputs to FARSITE to simulate fire behavior on wildfires throughout the US.

4.5 Research and management needs and solutions

Irrespective of design, the performance of any simulation model is governed by the quality of the underlying science, the technical and creative ability of the modeler to quantitatively implement this understanding, and the data on which the model is based. Therefore, simulation research needs (i) high quality, relevant science on which to base future models, (ii) highly trained modelers to build, apply, test, and teach these models, and (iii) extensive data to develop, test, initialize, and parameterize models.

The quality of the underlying science is determined by the cumulative advancement of physical, ecological, and climatic knowledge gained by experimentation, observation, and publication in peer-reviewed journals. This information provides a mechanistic understanding of key processes that govern ecosystem dynamics as accepted by a broader scientific community. Because the future is so uncertain, it is vitally important that there are comprehensive research programs aiming at understanding novel ecosystems, fuel conditions, and social issues that will evolve as climate are modified, human populations increase, and attitudes change. Since mechanistic approaches are suggested, it is important that field research projects should endeavor to quantify the causal relationships that govern fire and ecosystem dynamics so that these mechanistic equations can be developed for implementation in future simulation models. There is a critical need for fundamental research into the basic physical processes that control fire behavior and subsequent effects. Fine scale fire behavior outputs should feed detailed ecosystem models to mechanistically predict ecological response. This includes a new theory of wildland combustion physics and a more physiological approach to simulating vegetation response to fire and the subsequent development.

Good modelers, possessing comprehensive knowledge across diverse disciplines, are rare. However, even good modelers are limited by many reasons. First, the background, knowledge, and experience of a modeler can limit the scope, quality and complexity of model design. Second, there is an extent to which evolving ecological understanding can be feasibly incorporated into existing model structures based on the modeler's skills. There are also limitations of data parsimony and availability, and engineering restrictions, which hinder the incorporation of new knowledge into existing model structures by even the most accomplished modeler. More theoretical issues, such as temporal and spatial scale reconciliation and representation (Urban 2005) and socio-economic factors, also pose additional challenges to the modeling community. Therefore, education programs, particularly at the graduate level, must emphasize a diversity of modeling approaches and multi-disciplinary understandings if future fire models are going to possess the attributes needed in the future. In addition, modeling projects must involve collaboration across multiple disciplines to ensure that current science is appropriately integrated into model algorithms.

At the center of future simulation research is a need for comprehensive data to run and validate future models. The balance of data needs versus model advancement reflects a critical imperative for cross-fertilization between field ecologists, who provide data and equations to modelers, and modelers, who must then integrate that knowledge to provide descriptions of phenomena at different spatial and temporal scales. It is critical that extensive field programs should be intimately integrated with simulation efforts to ensure that sufficient parameter and validation data are measured for model applications. Temporally deep, spatially explicit databases created from extensive field measurements are needed to quantify input parameters, describe initial conditions, and provide a reference for model testing and validation, especially as landscape fire models are ported across large geographic areas and to new ecosystems (Jenkins and Birdsey 1998). For example, Hessl et al. (2004) compiled a number of ecophysiological parameters for use in mechanistic ecosystem models, which has increased parameter standardization and decreased the time modelers spend on parameterization. New sampling methods and techniques for collecting the data are needed to ensure that the right variables are being compared at the right scales. Field data useful in simulation modeling should be stored in standardized databases, such as FIREMON (Lutes et al. 2006), and stored on web sites so that they are easily accessible for complex modeling tasks. Last, new instruments are needed to quantify important simulation variables such as canopy bulk density, to initialize and parameterize fire behavior models (Keane et al. 2005).

Model validation research is also critically needed to ensure that future models are behaving realistically and accurately (Rykiel 1996; Gardner and Urban 2003). There are many ways to validate models. The most preferable one is direct comparison of model results with field measurements in the proper spatial and temporal context. Next, intermediate results from model algorithms or modules can be compared against appropriate field data (Oderwald and Hans 1993). Results from complex sensitivity analyses can also be used to evaluate model behavior and to compare behavior against measured data, expert opinion, or modeler experience (Cariboni et al. 2007). Comparative modeling exercises or ensemble modeling is also another potential tool for validation where several models are applied to the same area (e.g., stand or landscape) under the same initial conditions with comparable parameterizations (Cary et al. 2006). Results from ensemble modeling can be used to evaluate the sensitivity, accuracy, and validity of model results and to explore new ecosystem responses (Cary et al. 2009). Model outputs can also be evaluated by a panel of experts to estimate the degree of accuracy and realism (Keane et al. 1996).

New quantitative methods are also needed to evaluate the uncertainty around model predictions (Gardner and Urban 2003). Statistical tests and analysis methods are needed to support validation comparisons and sensitivity tests that account for spatial and temporal autocorrelation and test for significance (Mayer and Butler 1993). Critical to testing and validating models is an assessment of whether the internal complexity of the model design is actually manifested in results (Cary et al. 2006) and to bound complexity in an estimate of uncertainty (Kleijnen et al. 1992). Modelers need statistical tests that compare the variance and trend of simulation results to the expected outcomes from model algorithms (O'Neill 1973). They also need both statistical algorithms and software to test the model over its entire range of applicability and create response surfaces for various initial conditions, parameterizations, and scenarios. There are also needs for a modeling science research agenda where new modeling approaches, methods, and protocols are developed to ensure that models are used correctly by fire management. Optimum simulation landscape size and shapes are needed to define the spatial context for future simulation projects (Karau and Keane 2007) and proper equilibration periods must be determined to ensure that managers incorporate meaningful results (Pratt et al. 2006). The appropriate number of simulation replicates for stochastic models must be defined along with the appropriate simulation time spans for creating fire regime maps (Keane et al. 2002). Methods and guides for selecting the most appropriate model for a management application are also needed.

There is a need for future modeling endeavors to create programming code that is efficient, fast, and useful to management and other modelers for many purposes and applications. There are many programming concerns that provide challenges for optimal model design that include:

- *Cross-platform design.* Ability to compile the model on many machines for many operating systems.
- *Modular design.* Model functions should be built in modular form with open code so that modelers can modify coded algorithms for integration in another model.
- *Graphical User Interface input/output.* Easy way to enter input and understand output.
- *Open source and integrated code.* Source code is written in modular style and is posted or published for others to use.
- *Multi-threaded executions.* Ability to run on many processors across many computers.
- *Extensive documentation.* All written code is fully documented including clearly defined variables and associated units, descriptions of modules and their input and output structure, and descriptions of all functions. User manuals, model descriptions, and design descriptions should be published and meta-data recorded for all input/output parameters.
- *Simulation history retention.* Ability to remember past simulations to inform future simulations.

New software technologies must be developed to ensure efficient modeling building, rapid execution, data access and storage, modular sharing, and diverse debugging abilities. New computer technology might be needed to support this new software. This may require fire management to evolve a new capacity to implement, run, and interpret the fire models of the future, suggesting specialized training and forming teams of modeling experts within and across agencies.

4.6 Summary

Fire management and research will continue to depend on computer simulation for many projects and applications. However, many aspects must be incorporated in new models to be useful in the future (Table 4.3). Models will need spatially explicit, mechanistic designs that simulate physical processes and their interactions over multiple scales. Management-oriented models must be synthesized from the complex fire models created for research explorations. Research and management field efforts must collect data that is useful to parameterization (research studies), initialization (inventory), and validation of

Table 4.3 Challenges and research needs for simulation in fire management and research applications.

Challenge	Research need	Possible directions	Major user
Data collections	Need sampling methods and protocols	Standardized databases, map creation procedures, database management technologies	Research and management
Mechanistic design, balancing complexity with utility	Field research in basic physical process, inventory systems that quantify mechanistic parameters	Ecophysiological research,	Research
More efficient models	Innovative software design, better computer resources	New programming software, open source code development, modular code design	Research
Accurate and realistic models	Model analyses and validation procedures and technology	Ensemble modeling, meta-modeling, novel field sampling techniques	Research and management
Model use by managers	Develop models that are parsimonious but explanatory, develop effective training and application vehicles	Expert cadres, centers of excellence, extensive training courses and workshops	Management

fire models. Software and hardware technologies need to be developed that facilitate efficient and rapid simulation. New test and validation statistical designs will be needed to evaluate the reliability and uncertainty in simulation results. And, modeling science research will need to develop suitable guidelines for using and interpreting models. To effectively use these advances in modeling technology, management will need to train modeling specialists to effectively utilize these models and interpret their results. Simulation holds an important role in the future of fire management but it is up to research to develop comprehensive models that predict and explain important ecological phenomena, and it is up to fire management to understand these complex models so they can be used effectively in common analysis tasks.

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Chapter 5 Using Landscape Disturbance and Succession Models to Support Forest Management

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Abstract

Managers of forested landscapes must account for multiple, interacting ecological processes operating at broad spatial and temporal scales. These interactions can be of such complexity that predictions of future forest ecosystem states are beyond the analytical capability of the human mind. Landscape disturbance and succession models (LDSM) are predictive and analytical tools that can integrate these processes and provide critical decision support information. We briefly review the state of the art of LDSMs and provide two case studies to illustrate the application and utility of one LDSM, LANDIS. We conclude that LDSMs are able to provide useful information to support management decisions for a number of reasons: (i) they operate at scale that is relevant to many forest management problems, (ii) they account for interactions among ecological and anthropogenic processes, (iii) they can produce objective and comparable projections of alternative management options or various global change scenarios, (iv) LDSMs are based on current ecological knowledge and theory, (v) LDSMs provide a vehicle for collaboration among decision-makers, resource experts and scientists, (vi) LDSMs are the only feasible research tool that can be used to investigate long-term, large area dynamics.

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Keywords
Landscape models, disturbance, decision support, scale.

5.1 Introduction

Forest managers must balance increasing demand for wood products and bioenergy feedstocks with the long-term maintenance of the integrity of the ecosystems that provide multiple valuable ecosystem services. Because forest ecosystems are characterized by many processes operating at multiple scales, interacting with each other and with the biotic and abiotic environment, a landscape perspective must be integrated into the thinking of land managers. Managing forests while considering only the stand scale will not achieve ecosystem sustainability objectives. Forests develop through the interplay of dynamic processes such as plant establishment, growth, competition and reproduction, and these are mediated by the abiotic environment (e.g., substrate, climate) and disturbances such as fire, herbivory and harvest. Interactions among these many processes can be complex; so much so that predictions of future forest ecosystem states are beyond the analytical capability of the human mind. Consequently, forest managers need computer-based tools to provide the predictive and analytical decision support information they require.

Decision support tools used by forest management agencies are typically non-spatial, non-ecological, non-process based models. For example, forest optimization models combine growth and yield with harvest scheduling to support timber-oriented forestry (e.g., Cogswell and Feunekes 1997). Such models are well suited for their intended purpose, but they lack integration with key ecological processes such as succession and natural disturbance, which limits their use when ecological sustainability is also a management goal (Fall et al. 2004). Habitat Suitability Index models rely on empirical relationships, but they rarely have a spatial component or a mechanistic basis. Forest management decision support tools may include an *ad-hoc* collection of non-spatial models, spreadsheet models, GIS analysis and expert opinion (Baskent and Keles 2005). Consequently, there is an urgent need for comprehensive spatial models that can (i) accommodate multiple management goals and actions, (ii) include multiple ecological processes and their interactions, (iii) include spatial interactions, (iv) evaluate large areas and (v) make holistic predictions about ecosystem properties. Because multiple global changes are affecting forest ecosystems, it is also desirable that the models can predict responses to novel conditions that have not been empirically observed before. In these situations, the only reliable way to project landscape change and estimate ecological sustainability is through modeling based on ecological processes rather than statistical relationships estimated under past conditions.

Comprehensive spatial models that can integrate multiple ecosystem processes are invaluable to decision makers because they provide information that cannot be derived from other tools. This includes projections of the spatial distribution of forest composition (species and age classes), carbon and nutrient cycling and disturbance regimes. Perhaps the most useful characteristic of modeling is the ability to objectively compare the response of the ecosystem to alternative management strategies or global change scenarios.

Dynamic landscape models combine the scientific knowledge accumulated in hundreds of disparate forestry and ecological studies to project how a forested ecosystem might be expected to respond to specific internal and external driving forces. These models are simply computational formulations of our human understanding of the components of complex ecological systems, and they are able to integrate these complex components in ways that the human mind simply cannot do.

This paper focuses on dynamic forest models that make projections of forest conditions over large areas (landscape scale) and long time periods by simulating forest succession and one or more forest disturbances. Such Landscape Disturbance and Succession Models (LDSM) have been constructed to achieve at least one of the following objectives. (i) Understand implications and interactions of scientific assumptions and hypotheses (i.e., if assumptions are correct, then model output represents how the system will behave.). (ii) Identify important processes for further study (sensitivity). (iii) Enhance understanding of complex ecological systems (heuristic). (iv) Integrate ecological and forestry issues for research and planning purposes. (v) Support an ecosystem approach to management. (vi) Account for spatial processes and spatial dynamics. (vii) Consider long temporal and large spatial scales. (viii) Account for interactions among ecological and management processes. (ix) Make projections about future forest ecosystem states — composition and pattern. (x) Conduct virtual landscape experiments and scenario analysis — to answer the "what if" questions.

5.2 Overview of landscape disturbance and succession models

One major component of LDSMs is the ability to simulate disturbance. Most disturbance simulators are process-based, simulating disturbance events (e.g., triggers, probabilities, location, size, intensity, spatial characteristics) and effects on species or community type (e.g., cohort mortality, biomass reduction, change to another type). Alternatively, a pattern-based approach places disturbance patches spatially and temporally on the landscape using mean disturbance regime properties. Disturbance effects depend on pre-disturbance site conditions and disturbance intensity. Disturbance is modeled as explicitly spatial processes, and these processes interact with the spatial pattern of vegetation and the environment.

Succession is the other major component of LDSMs, and it is simulated in one of two ways in most LDSMs — pathway-based or process-based. In a pathway-based system, there is a well-defined successional trajectory, and communities transition from one successional stage to the next at a predefined temporal rate unless disturbance resets them to another stage. The number of pathways is often limited. In a process-based system, succession may have many possible endpoints, and it is simulated based on the life-history attributes of the species and conditions found at each site on the landscape. The suitability of each approach is quite dependent on the ecosystem. For systems that have fairly predictable successional trajectories, such as in the American West, the pathway approach can save considerable computing time. For ecosystems where multiple successional trajectories may occur somewhat stochastically, such as in temperate mixedwood forests, then a process-based approach may produce more realistic results. Pathway-or transition-based LDSMs include VDDT/TELSA (Merzenich et al. 1999), LANDSUM (Keane et al. 1997), SIMPPLLE (Chew 1997), BFOLDS (Perera et al. 2008) and RM-LANDS (http://www.umass.edu/landeco/research/rmlands/rmlands.html), while the major process-based LDSMs are LANDIS (Mladenoff 2004; Scheller et al. 2007) and LANDSIM (Roberts and Betz 1999) (Table 5.1). See Scheller and Mladenoff (2007) and Messier et al. (2003) for excellent reviews of LDSMs.

Table 5.1 How succession is modeled by the major LDSMs.

Model	Succession trajectory	Succession process
VDDT/TELSA	Pathway	Deterministic
SIMPPLE	Pathway	Stochastic
LANDSUM	Pathway	Deterministic
BFOLDS	Pathway	Stochastic
RMLANDS	Pathway	Stochastic
LANDIS	Process	Stochastic
LANDSIM	Process	Stochastic
SELES	User-defined	User-defined

The primary distinctive of LDSMs is spatial interactions. A model is spatial if it represents system components in geographic space, and considers the spatial relationships between objects. A model is spatially dynamic if these spatially-referenced components can change, therefore changing the spatial pattern of the modeled system. Most LDSMs simulate: (i) establishment and growth of tree species or communities, (ii) modification of species or communities by disturbance, and (iii) a fairly large spatial domain (100 to >10,000 km^2). Many LDSMs model ecological communities, which are assemblages of species, and in some models, these communities are composed of specific species and there are no compositional dynamics within them. In others, individual species or guilds are modeled, and communities are therefore dynamic and become an emergent property of the simulations. One LDSM (SELES, Fall and Fall 2001) is actually a declarative modeling language with a library of routines and functions that allows users to model spatial, landscape-level processes on raster map layers. This approach allows users to customize the way succession and disturbance is simulated.

The strengths and limitations of the various LDSMs are directly determined by the objective for their use and the ecosystem to which they are applied. A good rule of thumb is to use the simplest model that allows the question at hand to be answered. Complexity can increase uncertainty by adding parameterization and specification error. On the other hand, if a process has an important effect on landscape conditions and dynamics, its omission also increases prediction error. LDSMs have the capability to model most major disturbance processes, and in some cases, specific disturbances can be turned on or off, depending on the question (Table 5.2). LDSMs vary considerably in the amount of spatial dynamism that can result from the simulated processes. Spatial dynamism refers to the ability of the spatial pattern of the landscape (e.g., forest type, age classes, fuels) to change in response to the simulated processes. The algorithms used to simulate processes must be consistent with the way each process works in the ecosystem being studied. It is advisable to match the question and the level of detail for the ecosystem processes that are to be modeled to the modeling approach of a specific LDSM. The reason many LDSMs are in use is because each fills an important modeling niche. In this chapter we describe two applications of the LANDIS LDSM that provide examples of matching the model to the question, and illustrate ways that LDSM simulation results can be useful for decision support at landscape scales.

Table 5.2 Disturbance processes modeled by the major LDSMs.

Model	Fire	Insects	Disease	Wind	Harvest	Climate Change
VDDT/TELSA	X	X	X		X	
SIMPPLLE	X	X	X		X	
LANDSUM	X		X		X	
BFOLDS	X					X
RMLANDS	X	X	X		X	X
LANDIS	X	X	X	X	X	X
SELES	X	X	X	X	X	

5.3 Case studies

In this chapter we describe two applications of the LANDIS LDSM that provide examples of matching the model to the question, and illustrate ways that LDSM simulation results can be useful for decision support at landscape scales.

5.3.1 Reducing landscape-level wildfire risk on the Chequamegon-Nicolet National Forest

The first case study used the LANDIS LDSM (Mladenoff 2004) to address a strategic question at a landscape scale, being aimed at determining which of several strategies is most effective to reduce the landscape-scale risk of wildfire. Fire mitigation is especially problematic for managers of large public forests in the United States because public lands are surrounded by land over which agency managers have no control. Wildfire in fire-prone ecosystems is a landscape-scale phenomenon, so management strategies to mitigate landscape-level fire risk are exceptionally difficult to develop when much of the land base is outside of the manager's control. LANDIS is well suited for evaluating alternative potential solutions to such a complex management problem.

In this case study (Sturtevant et al. in press), LANDIS 4.0 (He et al. 2005) was applied to evaluating the relative effectiveness of four alternative fire mitigation strategies on the Chequamegon-Nicolet National Forest (CNNF) in Wisconsin (USA), where fire-dependent pine and oak systems overlap with a rapidly developing wildland urban interface (WUI). Much of northern Wisconsin is dominated by fire-resistant hardwood forests, but in places there are significant areas of pine and oak forests associated with sandy glacial landforms that are prone to high intensity fires and dependent on frequent fire for long-term persistence (Radeloff et al. 2000). Fire-dependent ecosystems are currently in decline in Wisconsin because of aggressive fire suppression policies (Radeloff et al. 2000). Human populations are rapidly encroaching on forested areas in the region primarily for quality-of-life reasons. Consequently, human-caused fire ignitions are increasing (Cardille and Ventura 2001) and there are more homes to be destroyed by wildfires.

LANDIS represents landscapes as a grid of interacting cells. Each cell may contain multiple species and each species can be represented by one or many age cohorts. Each cohort will establish and respond to disturbance as a function of its life history attributes (e.g., shade tolerance) and, in the case of disturbance, its age. The succession and disturbance processes act on the cohorts found on cells, and their interactions emerge as a consequence of the changes each imposes on landscape cells. Spatial inputs for LANDIS take the form of raster maps and include the land types (ecoregions), tree species cohorts initially found on each cell, and timber harvest management areas. Model output primarily consists of maps.

The 780 km^2 study area is defined by the outer boundary of the Lakewood subdistrict of the CNNF, located in northeastern Wisconsin (Fig. 5.1). Seventy-four percent of the land area is owned by the CNNF, and the remainder is privately owned. The majority of private land in the study area contains low density housing, but there are several locations where housing density exceeds 6.17 houses per km^2. Land cover is dominated by forest (81%), with some agricultural and hay fields (4.5%) and open wetlands (12.5%). Forested ecosystems in the study area are strongly influenced by glacial landforms that create a sharp soil moisture gradient from west (mesic and nutrient-rich, northern hardwoods) to east (xeric and nutrient-poor, pine and oak). An extensive unimproved road network is maintained to provide access for both harvest and fire suppression activities, linked by improved county and state roads (Fig. 5.1). The research team (Sturtevant et al. in press) assisted the CNNF in developing and evaluating alternative fire and fuel mitigation strategies for the study area.

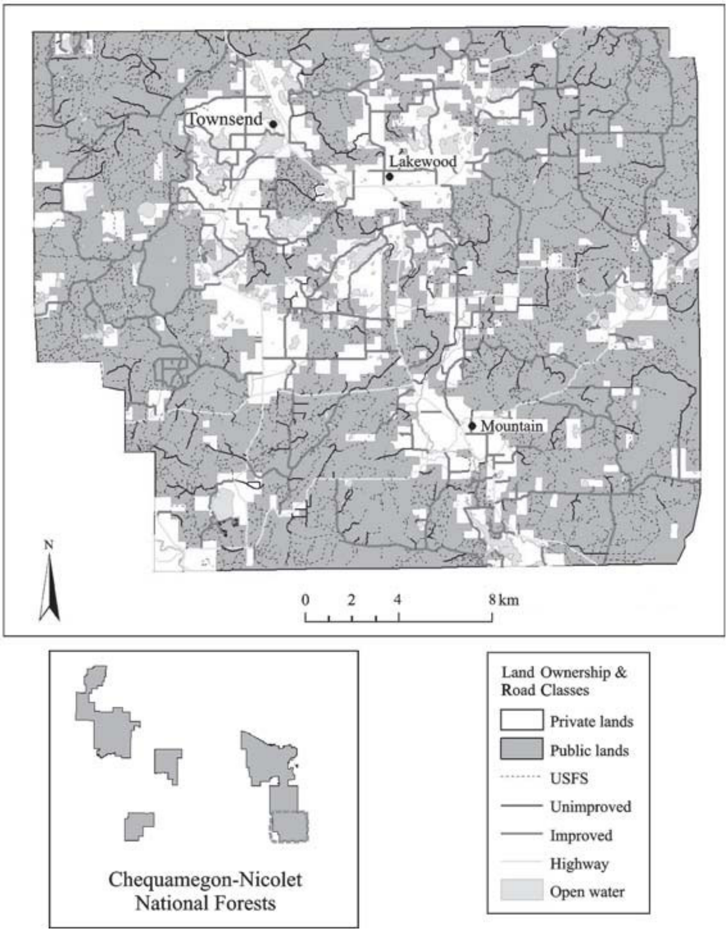


Fig. 5.1 Study area on the Chequamegon-Nicolet National Forest in Wisconsin, USA.

The alternative strategies were (i) placement of permanent firebreaks within fire-prone land types (FIREBREAK), (ii) redistribution of "risky" management treatments (i.e., those establishing pine or oak) to zones with low housing density (ZONE), (iii) reducing fire ignitions by 25% by banning local debris-burning practices (DEBRIS) and (iv) reducing fire ignition rates along roads by roadside vegetation management on federal lands (ROAD). The alternatives were evaluated by comparing a simulation of each alternative strategy to a base scenario representing current natural and anthropogenic disturbance processes of fire (including human ignitions and suppression), wind and timber harvest. The details of model parameterization can be found in Sturtevant et al. (in press) A 4×2 factorial experiment was designed with three replicates of each combination. Simulations were run for 250 years. Response variables were the cumulative area burned both inside and outside WUI areas during the 250-year time period. MANOVA was used to evaluate the null global hypothesis that neither treatments nor their interactions had significant effects on the response variables. The treatments were evaluated to determine if they had unintended consequences on ecological goals by comparing ecological indicators with targets outlined in the CNNF forest plan. Spatial maps of fire risk were estimated as the cell-scale probability of burning during 100 replicate simulations.

Results indicated that eliminating debris fires as an ignition source had the greatest influence on the area burned, decreasing the cumulative area burned relative to the base scenario by 35% (Fig. 5.2). This response was consistent both within and outside WUI areas. The ZONE treatment had the next largest influence on area burned, though the magnitude of change was small relative to the DEBRIS treatment. The ZONE treatment decreased the area burned inside the WUI by about 15%, but slightly increased the area burned outside the WUI, though the latter was not significant ($p > 0.05$) (Fig. 5.2). The ROAD treatment had marginal influence on area burned, and the FIREBREAK treatment had virtually no effect. No interaction terms were significant and were therefore removed from the analysis. Simulated mitigation treatments had little influence on either landscape-scale forest composition or the ecological goals of the CNNF.

Fire mitigation strategies may hold promise for coexistence of human and fire-dependent forest types. The simulated ban on debris-burning practices substantially decreased fire risk, suggesting that fire prevention and education is an important strategy for reducing fire risk within the Lakewood area. The simulations also showed that landscape-scale forest management strategies, such as the redistribution of fire-dependent forest types away from human ignition sources, can offer viable solutions for mitigating long-term fire risk and reducing land-use conflict in multi-ownership landscapes. However, because the legacy of previous forest composition is typically a prerequisite for the reestablishment and long-term maintenance of fire-dependent forest types, strategic planning will be essential for identifying opportunities for ecosystem restoration while minimizing fire risk. Landscape simulations, such as those presented here, can help guide the planning process by exploring the conse-

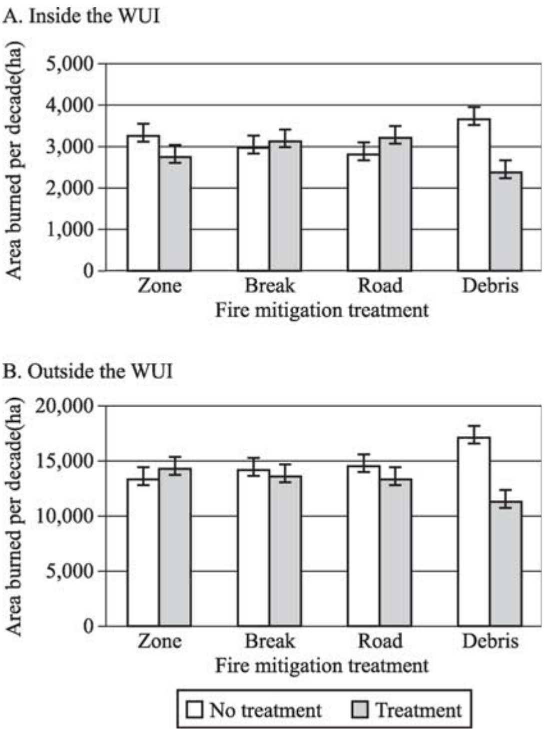


Fig. 5.2 Mean area burned per decade. A, inside the WUI, and B, outside the WUI, in response to the four main fire mitigation treatments. Error bars correspond to standard errors of the mean (Sturtevant et al. in press).

quences of different management options in a spatial context. Such exploration is critical before long-term management investments are made. For example, fire breaks can have unintended influence on fire risk due to the spatial interaction between human activities and human-caused ignition patterns. Given the declining trend of fire-dependent communities and the increasing trend of rural development, public land managers are poised to play an essential role in long-term conservation and maintenance of these key communities — but only if the conflict between fire disturbance and human safety can be resolved.

To that end, the results of the study were presented by Sturtevant to forest and fire management personnel in the region. Reactions to the results ranged from affirmation of their own perception of key relationships to surprise. Many were relieved to see patterns they intuitively understood but had difficulty expressing to decision-makers at higher levels. For example, much of fire research and resulting policies in the United States come from the western states. The idea that fire suppression can lead to reduced fire risk can be foreign to those with a “western” perspective. The reality of decline in fire-dependent ecosystems was another issue they were well acquainted with, but this fact is not well appreciated at higher organizational levels. The simulations clearly showed a long-term loss of fire-dependent tree species. By contrast, some (but not all) participants were surprised that fire breaks did not significantly reduce fire risk. Construction of fire breaks is another fire mitigation strategy common to the western US that may not transfer well to more settled areas of the upper Midwest, where fires are generally smaller and existing fire breaks, including a dense road network, are already in place (Malamud et al. 2005). The simulation results showed that fire breaks can have a strong local effect, but the effect is not significant at the landscape scale. Finally, the results affirm the current policy of the State of Wisconsin to control where and when debris burning is allowed through a simple, no-cost permit system (http://dnr.wi.gov/forestry/fire/burning-rp.htm). In each case, the simulation results provided objective evidence to help land managers communicate the rationale for their management priorities and to allocate limited resources for fire risk reduction.

5.3.2 Global change effects in Siberia

The second case study used LANDIS-II to address a policy-relevant question at the national scale, and focused on how multiple, overlapping global changes will affect the forests of south-central Siberia (Russia). Some of the authors are part of the team that re-engineered the LANDIS model using modern software development techniques (Scheller et al. in press) to create LANDIS-II (Scheller et al. 2007). LANDIS-II consists of a core collection of libraries (Scheller and Domingo 2006) and a collection of optional extensions that represent the ecological processes of interest (described below). LANDIS-II was specifically designed to address climate change effects on forested ecosystems (Xu et al. 2007; Scheller and Mladenoff 2008; Xu et al. 2009), by linking to the outputs of global circulation models (GCMs) to allow climate change to interact with landscape processes in the simulation environment.

The forested regions of Siberian Russia are vast and contain about a quarter of the unexploited forests worldwide (Dirk et al. 1997). However, many Siberian forests are facing twin pressures of rapidly changing climate and increasing timber harvest activity. Mean temperatures have risen significantly over the past 40 years, and this trend is expected to continue, while precipitation trends are variable (IPCC 2007). The combination of altered climate and altered species interactions will eventually produce altered disturbance regimes. The incidence and severity of fires is likely to increase (Litkina 2003; Goldammer et al. 2004; Efremov and Shvidenko 2004). A moderation of the harsh Siberian winters may allow insect pests to become more widespread. The frontier of timber harvest activity is pushing into previously inaccessible areas. New forest openings will increase fragmentation, and the building of roads may increase human access and fire ignition rates. Forest policy and management systems must take into account changing conditions and multiple interacting processes in order to achieve sustainable forest use in the future and to avoid unintended consequences.

The 3,165 km^2 study area is situated in the north-eastern part of the *Severny leshoz* (i.e. Northern forest enterprise) near the city of Ust-Ilimsk (Fig. 5.3). The forests of the study area are comprised of seven dominant species (*Picea obovata, Abies sibirica, Larix sibirica, Pinus sylvestris, Pinus sibirica, Betula pendula* and *Populus tremula*). The major natural disturbances are wildfire and windthrow. The study area is remote, but was recently opened to timber production and a warming climate may allow outbreaks of a major insect defoliator (Siberian silk moth, *Dendrolimus sibiricus superanse*) to become more common (Kondakov 1974).

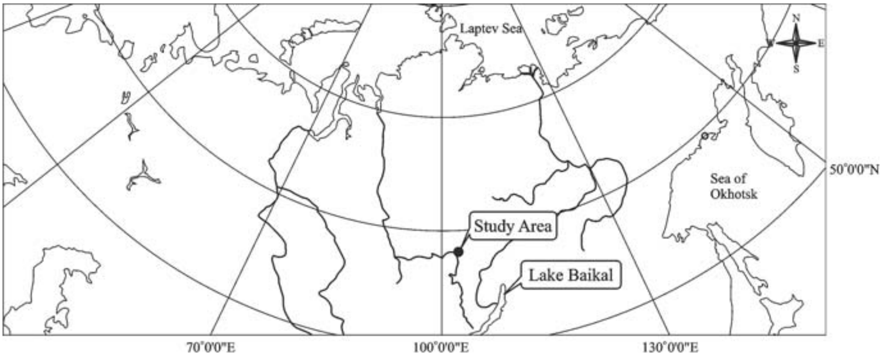


Fig. 5.3 Location of the study area in Siberia, centered at 58.9° N, 103.0° E.

To explore the effects of these impending global changes, we used LANDIS-II to simulate five scenarios: (i) the range of natural variability (recent climate and disturbance regime), (ii) increased timber harvest, (iii) changing climate through 2099 as predicted by Hadley A2 scenario (+5.1°C, +20% precipitation), which resulted in an altered fire regime (longer fire season, altered weather), (iv) Siberian silk moth outbreaks (with warmer climate) and (v) all changes combined (climate, harvest and insects). We used the simulation parameters described in Gustafson et al. (in review). Response variables were measures of forest composition, forest biomass and the landscape pattern of the forest.

Forest composition was influenced most strongly by timber harvest and insects (Fig. 5.4 and Fig. 5.5). The effect of the expected future climate treatment was significant, but its effect was minor compared to harvest and insects, excepting the abundance of Scot's pine. Climate did have a modest effect on the fire regime (Fig. 5.6). The total area burned per decade and mean severity of fires was projected to be slightly increased, with higher variability under the future scenario. However, both the area burned and fire severity were lower by year 300 under the future climate scenario because of changes in the species composition of the forest (Fig. 5.5). The amount of live aboveground biomass and the level of forest fragmentation were related to the amount of disturbance associated with each scenario (Fig. 5.7). Biomass increased during the last 100 years of the simulations under the insect scenario because insects favor tree species with higher growth rates. Harvest scenarios show a similar trend for similar reasons.

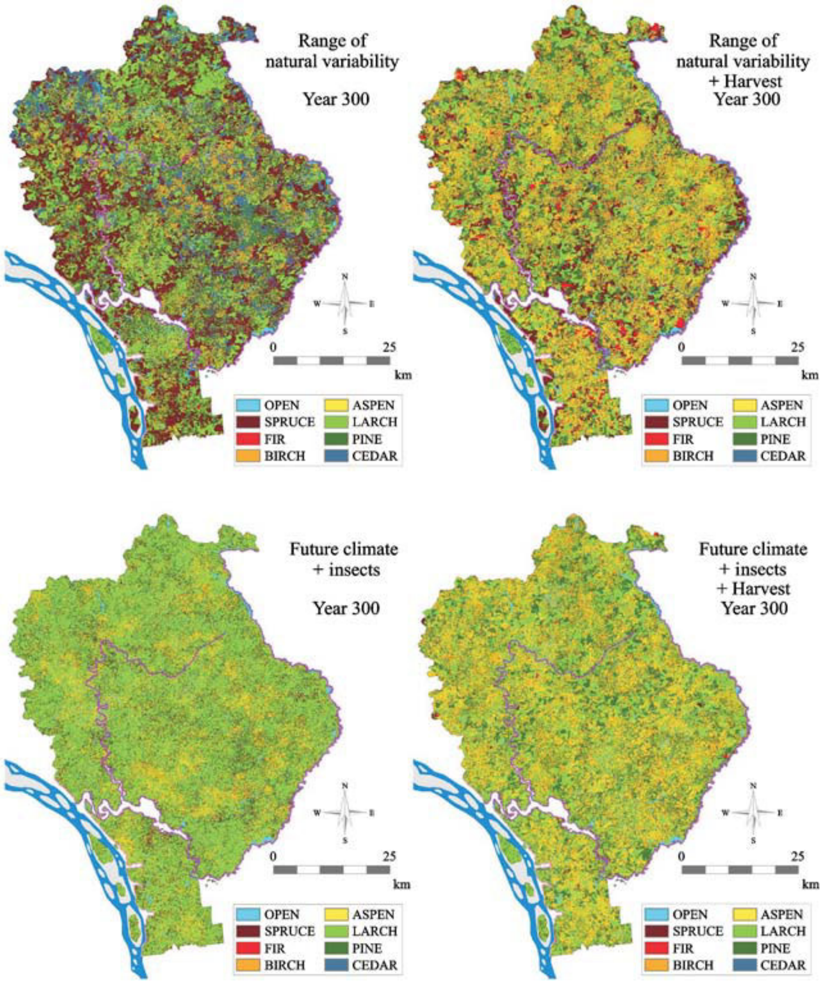


Fig. 5.4 Maps of forest composition at year 300 under four different scenarios.

Based on a comparison of these scenarios and on the results of simulation experiments by Gustafson et al. (in review), the following conclusions relevant to forest policy in the study area can be drawn. (i) The direct effects of climate change in the study area are not as significant as the exploitation of virgin forest by timber harvest and the potential increase in outbreaks of the Siberian silk moth. (ii) Global change is likely to significantly change

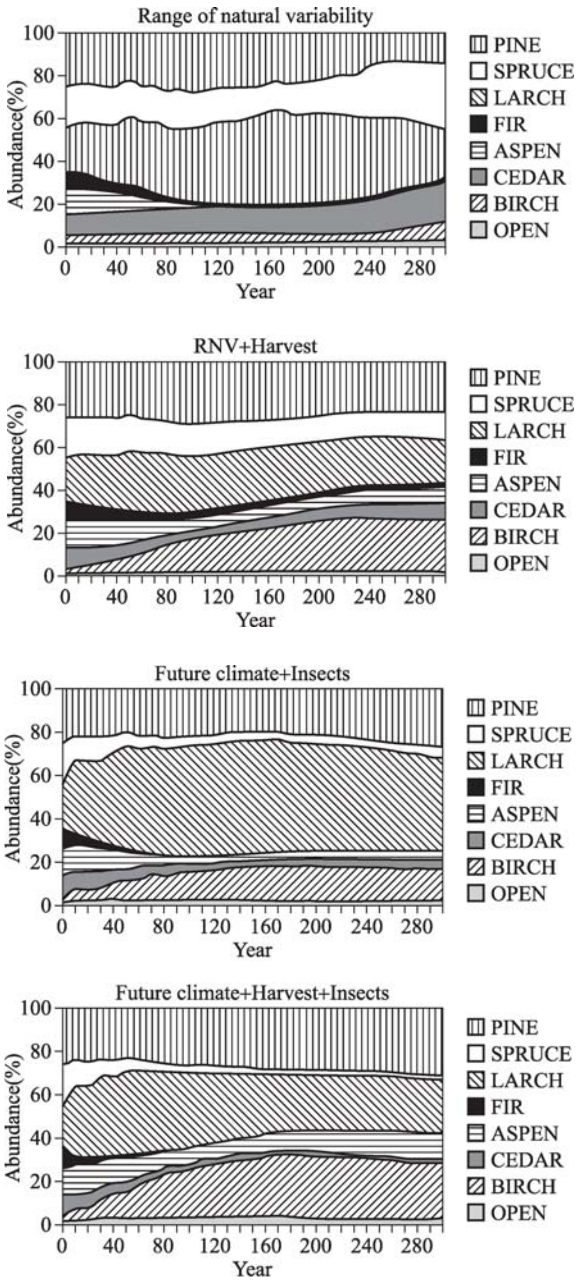


Fig. 5.5 Abundance of forest types defined by dominant species through time for four scenarios.

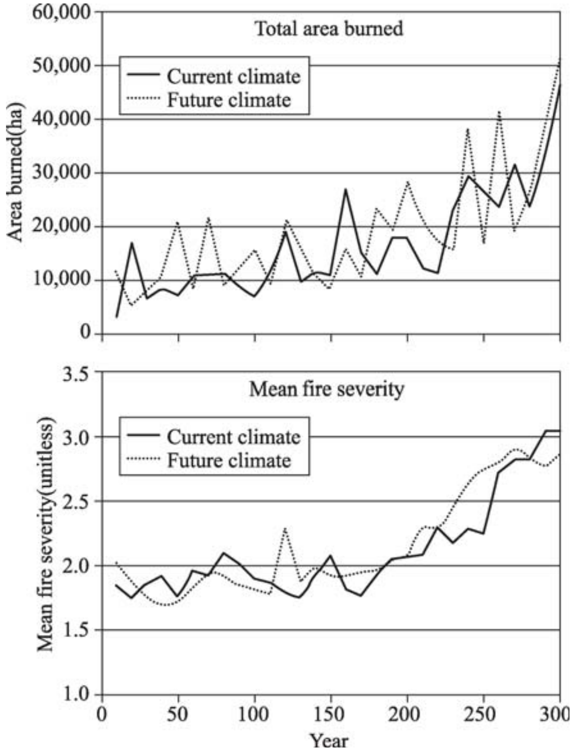
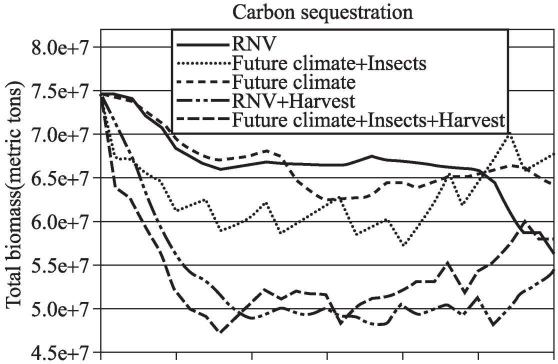


Fig. 5.6 Comparison of total area burned per decade and mean fire severity between current and future climate scenarios in the absence of harvest and insect disturbance.

forest composition of central Siberian landscapes, with some changes taking ecosystems outside the historic range of variability. (iii) Novel disturbance by timber harvest and insect outbreaks may greatly reduce the ability of Siberian forests to sequester carbon, and may significantly alter ecosystem dynamics and wildlife populations by increasing forest fragmentation.



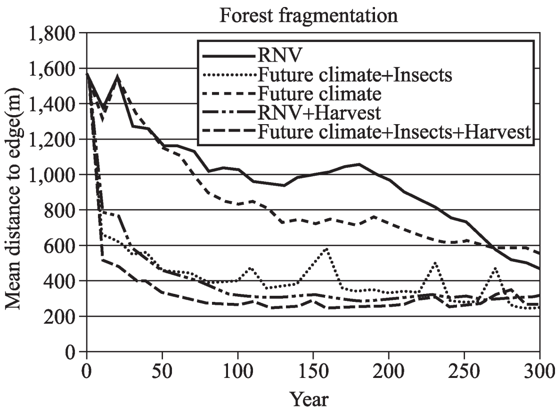


Fig. 5.7 Comparison of carbon sequestration and forest fragmentation through time among scenarios.

The results also suggest some forest management strategies that may help the forests in the region adapt to global change. (i) Encourage the regeneration of species that will be more productive under future climate (e.g., pine and birch) or able to tolerate increased fire (e.g., larch). (ii) Silk moth will have a negative impact on all conifers except larch. A potential strategy to mitigate insect losses is to begin to reduce landscape concentrations of spruce and fir, since these are major hosts for the silk moth.

5.4 General conclusions

LDSMs are able to provide useful information to support management decisions for a number of reasons. (i) They operate at a scale that is relevant to many forest management problems. A landscape perspective and long-time horizons are critical to understand most forest ecosystem dynamics and to make predictions about biodiversity and sustainability. Furthermore, many ecological processes have an important spatial component that cannot be ignored. (ii) LDSMs account for interactions among ecological and anthropogenic processes. These interactions are often complex and nonlinear, and are therefore difficult to predict without modeling tools. (iii) LDSMs produce objective and comparable projections of alternative management options or various global change scenarios. Results are reproducible, in a scientific sense, and can be peer-reviewed. This provides a level of objectivity, transparency and defensibility that managers need. (iv) LDSMs are based on current ecological knowledge and theory. This is both a blessing and a curse. The models are reliable when they have robustly encapsulated the conceptual models derived from ecological theory (Scheller et al. in press), and therefore their use carries significant stature. However, current theory and knowledge is subject to falsification as the scientific enterprise pushes back the frontiers of ignorance. Models (or model building platforms) that easily allow new knowledge to be inserted and the flexibility to eliminate processes irrelevant to the question at hand are often the most useful and enduring (Fall and Fall 2001; Scheller et al. in press). Equally important is recognizing the appropriate domain of each LDSM to ensure that an appropriate model is selected for the system being modeled and question being asked. (v) LDSMs provide a vehicle for collaboration among decision-makers, resource experts and scientists. When LDSMs are applied for decision support purposes, the most positive results accrue when modelers and managers collaborate in an iterative process focused on outcomes rather than the tools (Gustafson et al. 2006). Collaboration ensures that both modeling expertise and local ecological knowledge are brought to bear equally on the problem to be solved. One approach that our research group has found effective is the collaborative, iterative approach (Gustafson et al. 2006). Rather than expecting managers to learn to independently use the complex LANDIS model, we collaborate on decision support projects. We scientists provide the technical modeling expertise, the decision maker frames the question and defines the information needed, and local resources experts provide the ecological knowledge needed to ensure that the model behavior conforms to reality. Because the model and its results are described and discussed at some length as an integral part of the iterative process, the managers become educated about the technology, and the model is much less likely to be perceived as a mysterious "black box." (vi) LDSMs are the only research tool that can be used to investigate long-term, large area dynamics. Replicated, manipulative experiments are not feasible at landscape scales and temporal scales of decades or centuries. Yet LDSMs can provide useful insights into our understanding of ecological processes and dynamics at these scales.

We have described the application of the LANDIS LDSM to forest management questions, which illustrates several specific strengths of the LANDIS model. (i) LANDIS uses a process-based approach (spatial and non–spatial) to account for interactions among disturbances and succession to predict future forest ecosystem states. It is among the few LDSMs in which tree species respond individually to different disturbance processes (Keane et al. 2004; Scheller et al. 2007). This design allows vegetation patterns to emerge from the interplay between multiple disturbances, environmental drivers and species life history traits so that succession is not deterministic. (ii) LANDIS is flexible enough to allow application to varied problems, ecosystems and decision support needs. Our case studies illustrate this flexibility, being applied in temperate North America and boreal Russia. (iii) LANDIS can be updated to reflect new knowledge. Most ecological knowledge is input to the model in parameter files, although some is implicit in model design. Parameters can easily be changed as new knowledge is gained. (iv) If assumptions and relationships that are coded into extensions need to be changed, the LANDIS-II open-source extensions are readily modified and plugged into the model core.

LANDIS-II is one of only a few open source LDSMs, and this increases the transparency and verifiability of the model, which should increase confidence of model users.

5.5 Future of LDSMs in decision-making

Most LDSMs require detailed information about various ecosystem properties (e.g., species and age classes, fuel loading) at relatively high spatial resolution (at least 30 m) and extent (across an entire landscape). Data about age classes and specific species present are almost never available at high resolution, and so are estimated using various techniques. While these initial condition estimation techniques may be statistically accurate at landscape scales, they do introduce uncertainty and error at the cell scale, which persists for some time until the model itself produces new landscape patterns, a process that usually takes 50-100 simulation years. However, many management questions are focused on specific locations with a time horizon of less than 50 years, which is where uncertainty is highest. Methods to create input maps directly from remotely sensed data or broad-scale inventory data would reduce the uncertainty and error in the initial conditions. Methods to reduce the uncertainty of input parameter values are also needed. LDSM simulations can be used to identify the parameters to which the results are most sensitive, prompting new research to reduce the uncertainty of key parameters.

Most LDSMs were initially developed as scientific research tools, and their application for decision support is often secondary, in reality (King and Perera 2006). Explicit design and development of a user interface and application protocols are rarely done because of the expense, and therefore LDSMs are difficult for non-modelers to use. This situation presents a significant barrier to the adoption of LDSM technology. To bring the power of LDSMs to bear on the forest management questions of our time, these barriers must be removed. Investments must be made to design and implement systems that give non-modelers reasonable access to the technology. Alternatively a well designed user interface can automate input data generation, help the user specify parameters and model runs, conduct automated error checking of inputs, and provide some analytical tools to evaluate model output. For example, an exceptional interface has greatly expanded the use of FVS, a stand-scale model (http://www.fs.fed.us/fmsc/fvs/index.shtml). Such a system would make LDSMs much more attractive for adoption as a routine decision support tool by lowering the investment in training specialists to use the technology.

Reliability, scientific credibility and longevity are important for decision-makers. Most managers of public forests expect their decisions to be legally challenged, and they must be confident that decision support from models would be defensible in a court of law. Furthermore, if they commit to a modeling tool for decision support, there is often the expectation that the tool will also be used for future decisions. Therefore, they must have some confidence that the tool will be maintained, and that it will always reflect the most current scientific knowledge. Unfortunately, most ecological models are developed and maintained in an *ad hoc* manner, and their reliability often decreases with time. The application of modern software engineering techniques may greatly improve the reliability of LDSMs and make it easier to keep them current with scientific advances (Scheller et al. in press).

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Chapter 6 Research Methods for Assessing the Impacts of Forest Disturbance on Hydrology at Large-scale Watersheds

Xiaohua Wei* and Mingfang Zhang

Abstract

The impact of forest disturbance on hydrology has long been an important research topic, but the majority of this research has been conducted on small-scale watersheds. Large-scale watershed studies are hampered by the difficulty of conducting paired watershed experiments, and by insufficient data, significant landscape complexity and a lack of commonly-accepted research methodologies. However, the ever-increasing demand on information at large scales to support forest and watershed planning and management highlights the need for large-scale watershed research. This paper provides a review of research methods for assessing impacts of forest disturbance on hydrology in large-scale watersheds. It focuses on definition of large-scale watersheds, quantification of cumulative forest disturbance, and research methods used to detect its impact on hydrology.

There is no a commonly-agreed definition of large scales for watersheds. Researchers have called various sizes as large scales, leading to confusion and inconsistence of comparisons. We suggest usage of a common size ($\geqslant$1,000 km^2) to define large-scale watersheds, which is consistent with the majority of published research reports reviewed. Forest disturbances including human (e.g. timber harvesting) and natural ones (e.g. wildfire) are typically cumulative over time and space in large-scale watersheds. How to use a single indicator to represent this cumulative disturbance is challenging. We suggest a concept named "equivalent disturbed area" (EDA) to replace traditionally applied forest cover mea-

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sures as the former considers hydrological recovery process during forest re-establishment following disturbance. Several recent studies of this kind have confirmed that EDA is a better indicator or concept for studying watershed processes and functions particularly at large scales. Because of the difficulty of applying paired watershed techniques to large-scale watershed study areas, researchers have generally used modeling and statistical approaches. In this review, we grouped statistical techniques according to the number of selected watersheds. For a single watershed, methods like time series analysis, Bayesian methods, non-parametric analyses and flow duration curves could be used whereas to multiple watersheds double mass curve and regression techniques could be applied. We conclude that the selection of research methods largely depends on data availability and the number of available watersheds experiencing a gradient of forest disturbances. As there is no commonly-accepted method, a combination of several techniques can provide a more robust assessment than just one alone. Future research directions are also discussed.

Keywords

Large-scale watersheds, forest disturbance, ECA, hydrology, single watershed study, multiple watersheds study, time series analysis, DMCs.

6.1 Introduction

It is well recognized that forest disturbance such as timber harvesting, wildfire, hurricane and insect infestation can influence both quantity and quality of water resource. Numerous researches have focused on the impacts of forest disturbance on hydrology in small-scale or meso-scale watersheds of less than 1,000 km^2 (e.g. Cheng 1989; Wright et al. 1990; Keppeler and Ziemer 1990; Lavabre and Torres 1993; Burton 1997; Sun et al. 2001; Caissie et al. 2002; Whitaker et al. 2002; Woodsmith et al. 2004; Gomi et al. 2006; Brath et al. 2006; Amatya et al. 2006). However, research targeting large-scale watersheds (>1,000 km^2) is rare. When we searched in three major hydrological journals (*Water Resource Research, Journal of Hydrology* and *Hydrological Processes*), we were able to locate less than 30 papers published in the last decade, among more than 150 papers on small-scale watersheds (Fig. 6.1).

There are several key reasons contributing to fewer large-scale watershed studies. Perhaps the most important one is that the traditional, classic paired watershed approach designed for small-scale watersheds is not suitable for large-scale watersheds because of the difficulty of locating reference watersheds. Second, large-scale watersheds have diverse land uses, and complex components together with their interactions, and cumulative behavior. These characteristics make it a challenging task to isolate the forest disturbance

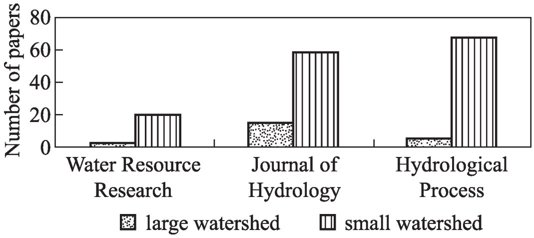

Fig. 6.1 Comparison of the number of published papers in the last decade on forest disturbance and hydrology between small- (<1,000 km^2) and large-scale ($\geqslant$1,000 km^2) watershed studies.

impacts on hydrology among many other confounding variables. Third, insufficient data coverage both temporally and spatially constrains our ability to assess cumulative hydrological response to forest disturbance at large scales. For the majority of world watersheds, data are simply sparse and of no sufficient long records to assess long-term forest disturbance history and associated hydrological changes. Finally, lack of commonly-accepted scientific methodologies for large-scale research is another critical factor. In spite of those constraints, research to address large-scale forest disturbance and hydrology has been in increasing demand, and also become more feasible recently, mainly because more advanced statistical techniques and computing power are available to be applied to more extensive data now obtained from the wide application of GIS, and remote sensing technologies.

Many issues on the environment are cumulative in effect and operative on a large scale. For example, climate change and its impacts on the environment are typically operating at regional and even global scales. With climate change and human impacts (e.g. implementation of forest fire suppression policy), large-scale forest disturbances such as wildfire and insect infestation (e.g. mountain pine beetle infestation in British Columbia, Canada) are more frequent and catastrophic. As a result, resource managers and planners require appropriate information to support management decisions to deal with those large-scale watershed environmental issues and to conduct sustainable forest and watershed planning. In the forest sector, management and planning strategies have gone through a significant shift from traditional stand-level to landscape or regional levels. Such a shift is necessary for achieving the balance between the need of economic development and the need for environmental protection. In short, demand for large-scale research has grown significantly in the past few decades.

Unfortunately, our scientific understanding of large-scale ecosystem processes is rather limited despite growing research. The lack of data or knowledge at large scales has forced resource managers to rely on data generated at small scales. However, it is widely recognized that extrapolation of information and data from small scales to large scales is risky and in some cases invalid. This suggests a critical need to conduct large-scale specific scientific research. The objective of this paper is to address one of the large-scale research constraints: research methods. Our specific objectives are (i) to provide a definition of large-scale watershed; (ii) to suggest a suitable indicator of watershed-scale forest disturbance; and (iii) to review and identify suitable research methods for detecting large-scale forest disturbance impacts on hydrology.

6.2 Definition of large-scale watersheds

How large is a large-scale watershed? To date, there is no consensus regarding the size for constituting or defining large-scale watersheds. Various sizes have been used in the past. Among the hydrological literature we examined, only 14 papers clearly specified the sizes of their large-scale watersheds (Table 6.1). Of these, 2 papers considered a watershed with an area of more than 100 km^2 as a large one, more than half used 1,000 km^2 and the remainder used 10,000 or 100,000 km^2. This wide range of sizes recommended in available literature makes it difficult for cross-comparison studies.

Table 6.1 Sizes used to define large-scale watersheds.

Scale(km^2)	Number of Reports	Literatures
$\geqslant 100$	1	Major and Mark 2006
$\geqslant 1,000$	8	Soulsby et al. 2004; Buttle and Metcalfe 2000; Sanford et al. 2007; Wilk 2001; Bronstert et al. 2002; Shaman and Burns 2004; He and Croley 2007
$\geqslant 10,000$	3	Dooge 1988; Lian et al. 2007; Donohue et al. 2007
$\geqslant 100,000$	2	D'Almeida et al. 2006; Abdulla et al. 1996

Historically, when there were no fast computers and spatial technology tools (e.g. GIS, remote sensing), spatially delineating and computing watershed characteristics and parameters were not easy tasks. Thus, relatively small to medium sizes of watersheds (e.g. +100 km^2) were considered large scale. With advancement in modern computation technologies, the definition of "large scale" has tended to increase. We suggest a size of around 1,000 km^2 would be suitable for this definition for the following reasons. First, large-scale watersheds in the size range of one to several thousand square kilometers are suitable for landscape or community level planning so that any research results at this scale can have practical applications. Second, watershed sizes ranging from one to several thousand square kilometers are suitable for studying forest disturbances and their impacts on watershed processes at a large scale. Sizes of less than 1,000 km^2 may not be large enough to include some important components (e.g. lakes, ponds or wetlands) and complexities, which are typical characteristics of large-scale watersheds. Conversely, sizes >10,000 km^2 may contain other significant land uses (e.g. cities, agriculture) which renders it difficult to differentiate forest disturbance effects from other confounding variables. Third, the majority of previous researches have used this size (Table 6.1). Nevertheless, it is a subjective divide, but a united definition can greatly help communication and comparison in a consistent way.

In studying climate change issues, large-scale watersheds sometimes were referred to as macro-scale watersheds (Kite et al. 1994; Hattermann 2005; Ma 2000). This is a term frequently used in meteorology, particularly in climate circulation modeling, which stands for basins larger than 100,000 km^2 or even continental scale. Clearly, this term or magnitude of scale is not suitable for studying forest disturbances and associated watershed processes.

6.3 Quantification of forest disturbance

Forest disturbances including wildfire, insect infestation, logging, mining and so on operate cumulatively over space and time in large-scale watersheds. In order to represent this cumulative nature, an integrated index is needed to quantify cumulative forest disturbances for a given watershed. This section introduces an integrated forest disturbance indicator, equivalent disturbed area (EDA).

6.3.1 Forest disturbance

Forest disturbances can be natural or anthropogenic or both, and they are normally characterized by regimes comprising frequency, intensity, size, pattern and agent(s). Natural disturbances (e.g. wildfire, flood, drought, hurricane, insect, etc.) are part of natural processes and they can play an important ecological role in forest and watershed ecosystems (Dale et al. 2000). They contribute to landscape diversity, nutrient circulation, species evolution, forest succession, and thus more resilient ecosystems. Because of their ecological significance, natural disturbance regimes are, therefore, often viewed as the best model for forest management guides (Roberts 2007; Bouchard et al. 2008). Natural disturbances generally have large variations in frequency, intensity, landscape patterns and consequently effects. By contrast, anthropogenic disturbances (e.g. forest harvesting, road construction, agricultural activities, urbanization, mining and recreation) generally have low variations, are relatively uniform, and can be permanent and catastrophic. Understanding of the differences between the two is needed for quantitatively describing forest disturbances.

In a large-scale watershed, forest disturbances operate at both broad and fine spatiotemporal scales from forest stand to landscape levels and from time to time. Disturbances of the various types have different effects in terms of patterns, sizes and severities. This clearly indicates that watershed disturbances are cumulative over time and space, involving various types. To assess impacts of forest disturbance on hydrology, an indicator of disturbance magnitude over a whole watershed is needed. To select an indicator to represent disturbance magnitude in a large-scale watershed, we must recognize the cumulative and complex nature of forest disturbances. In the past, some researchers have used a forest cover rate or percentage of a disturbed watershed to indicate watershed-scale disturbance (Buttle and Metcalfe 2000; Jones et al. 1996; Edwards and Troendle 2009). Although this concept is simple and relatively easy to generate, it has serious shortcomings. It cannot differentiate forest species, forest growth or recovery after disturbance, land use types, etc. For example, it treats urban paved area (i.e. roads) and open lands the same even though they are different in terms of infiltration capacity, and for that matter quantity and timing of runoff generation.

6.3.2 Quantification of forest disturbance

The most direct way to quantify forest disturbance in a large-scale watershed is to compute disturbed area (e.g. cumulative harvested area), mainly because these data are normally available. However, it serves merely as a basal forest disturbance indicator, and it cannot capture disturbance spatial patterns and subsequent forest recovery processes. A suitable forest disturbance indicator for a large-scale watershed should not only express all types of disturbances, their intensities and severities, but also reveal their cumulative forest disturbance history and subsequent recovery processes over space and time.

Equivalent roaded acre or area (ERA) and equivalent clear-cut area (ECA) are believed better indicators than disturbed area or forest cover rate because they account for dynamic vegetation condition or change following disturbance. A brief comparison of pros. and cons. between these indicators is presented in Table 6.2.

Table 6.2 Methods for quantifying forest disturbances.

Name	Advantage	Disadvantage
Disturbed area or forest cover rate	Simple calculation	Only available for single disturbance; No consideration of hydrological recovery
ERA (Equivalent roaded area/acre)	Accounts for various types of disturbance; assesses erosion risk and sediment yield	Complex calculation; No consideration of hydrological recovery; Lacks spatial representation(such as position of harvest)
ECA (Equivalent clear-cut area)	Accounts for various types of disturbance; Considers disturbance severity and hydrological recovery	Complex calculation; Lacks spatial representation (such as position of harvest) (Pike et al. 2007)

ERA was originally developed in the early 1980s by Region 5 of the USDA Forest Service to evaluate channel destabilization (McGurk and Fong 1995). Although it has been broadened to include some other cumulative impact sources, it mainly works for assessing sediment and erosion yield. Since it quantifies the total disturbance through the use of empirical coefficients and recovery curves for each forest activity (Cobourn 1989), the accuracy of ERA relies greatly on foresters' professional judgment for each activity. To some extent, it just indicates a level of risk but does not reflect the actual effect of forest practices. Moreover, ERA is not spatially explicit and the impacts of an activity cannot be tested against its location in a watershed (Menning et al. 1996; Cobourn 1989).

A similar index developed by the USDA Forest Service is ECA, which is widely used to assess the cumulative effects of forest harvesting on annual water yield. The ECA concept has also been widely used in Canada, particularly in British Columbia and Alberta. Roads, clearcuts, burned areas and partial cuts can all be expressed as "equivalent clear-cut area". There are various revised versions of ECA procedures, but the core concepts are similar (USDA Forest Service 1974; British Columbia Ministry of Forests 1996; King 1989; Silins 2000). Also, for fast ECA computation, Ager and Clifton (2005) developed a software program called ETAC (Equivalent Treatment Area Calculator), which has been successfully applied in some US forest management projects.

In a revised version developed by BC Ministry of Forests, ECA is defined as the area that has been clear-cut, with a reduction factor to account for the hydrological recovery due to forest regeneration (British Columbia Ministry of Forests 1996). Although originally designed for clear-cut areas, it can be applied to wildfire-killed areas, roads, and other open spaces. Research has established the relationships between vegetation growth (ages or tree heights) following disturbance and hydrological recovery rates so that ECA can be derived spatially and temporally (Hudson 2000; Talbot and Plamondon 2002). A simple and generalized relationship is shown in Fig. 6.2. ECA has been used in BC to test watershed-scale forest disturbance and its effects on various watershed processes including aquatic habitat (Chen and Wei 2008), hydrology (Lin and Wei 2008) and aquatic biology (Whitaker et al. 2002; Jost et al. 2008).

Generation of an ECA indicator for a large-scale watershed is not a trivial task. It involves data collection and calculation over millions of harvested or burned blocks. In some cases, such data may be stored in several areas of administration. In addition, the ECA concept may not differentiate disturbance severities in great details, and it may not explicitly consider other land uses (e.g. agricultural land) due to the lack of empirical relationships between these other land uses and watershed processes. Despite its weaknesses, ECA is so far the best indicator for assessing forest disturbance effects on hydrology in a large-scale forest-dominated watershed. Because of various types of

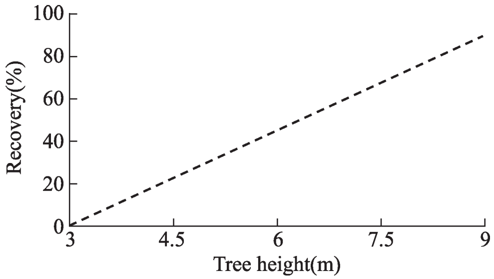

Fig. 6.2 Percent hydrological recovery with average forest stand height increases.

forest disturbances, we suggest a more inclusive concept, equivalent disturbed area (EDA) should replace ECA for future research. Our researches have further demonstrated that a more appropriate indicator than EDA or ECA is percent EDA or ECA because they show no confounding correlations with watershed properties (watershed area, elevation and gradient) (Chen and Wei 2008; Macdonald 2000). A further refinement on EDA or ECA would be spatially explicit assessments of the closeness of EDA or ECA to rivers or lakes and their effects on watershed processes.

6.4 Research methods on assessing impacts of forest disturbance on hydrology at large-scale watersheds

There are no commonly accepted methods for studying large-scale forest hydrology. In the past, researchers applied various methods according to objectives, data availability and watershed characteristics. This section provides a brief description of a general research approach, with more emphases placed on the statistical and graphical methods.

6.4.1 General approach

Current studies on hydrological changes in association with forest disturbance in large watersheds generally fall into two categories—analysis of change tendency and estimation of change magnitude. To detect whether forest disturbances have caused significant change in hydrology, statistical methods (e.g. non-parametric analysis, time series analysis) (Haan 2002) or graphical methods such as FDC (flow duration curve) and DMC (double mass curve) are commonly used (Maidment 1993). For forest management purposes, change magnitude is also of most interest, and it can be assessed by either a modeling approach or statistical regression. Selection of a suitable research method is largely dependent upon the purpose of the research, data availability and the number of selected watersheds. Fig. 6.3 shows a flow chart for research method selection.

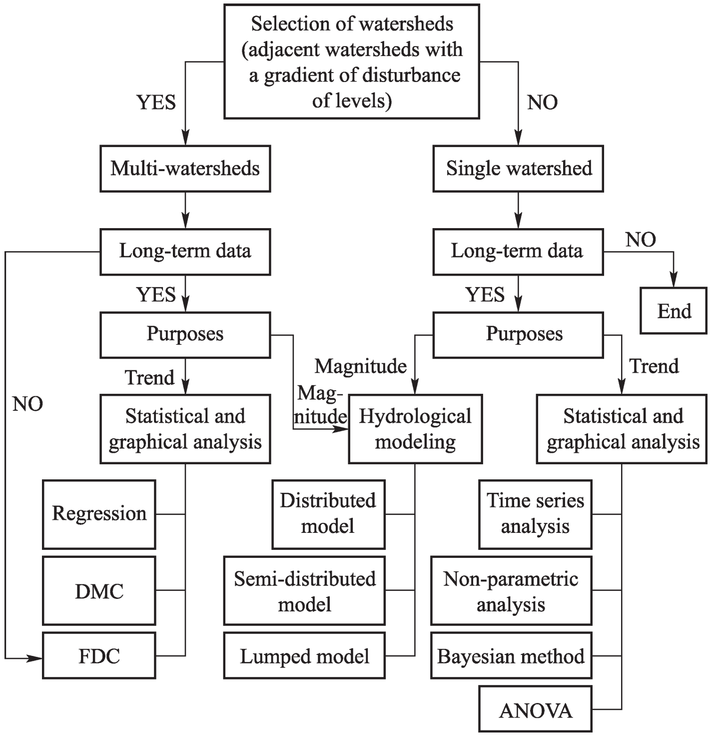

Fig. 6.3 A flow chart for selection of methodology for large-scale watershed research (FDC, flow duration curve; DMC, double mass curve; ANOVA, analysis of variance).

Although a modeling approach is not the focus of this review, a brief summary of key hydrological models used for large-scale watershed research provides a more complete context for this review. Hydrological models can be classified as lumped, semi-distributed and distributed in light of their spatial representation. Lumped models are not spatially explicit, but view the watershed as a whole, using the average values of the watershed characteristics and inputs, which consequently lead to averaging of hydrological processes. The model calibration and computation processes are simple, mainly suitable for a very large-scale watershed where there are not enough detailed spatial data for a more detailed analysis. In semi-distributed models, a watershed is divided into several sub-basins or landscapes, whose hydrological processes are modeled separately as independent response units. Spatial heterogeneity can be expressed to some extent, but not in great details. Unlike lumped or semi-distributed models, distributed models can well represent a watershed by assigning input data and physical characteristics to grids or elements (Putz et al. 2003). Physically based distributed models are able to provide distributed approximations or predictions of hydrological variables across watersheds, and thus have a better representation of reality. But a full distributed model requires a large dataset with data on various processes, components and their interactions. For a large-scale watershed research, a semi-distributed model is commonly used because of a general absence of detailed data at large scales.

Table 6.3 presents some examples of models applied to large-scale watershed studies to assess forest disturbance effects on hydrology. Clearly, different models serve different purposes (subject to data availability), and each one has its own strength and limitation. For example, DHSVM, a widely used distributed hydrological model, was developed particularly for assessing the effects of forest management on streamflow (Alila 2001). Like many other distributed models, its ability to simulate groundwater flow in unsaturated and saturated zones is limited. By contrast, MIKE-SHE is able to simulate the entire land phase of the hydrologic cycle including both surface and subsurface flows and their integrations(Danish Hydraulic Institute 2004), but its representation on evapotranspiration process is not as complete as DHSVM (Sun et al. 2006; Thyer et al. 2004).

In spite of increased applications, hydrological models are still based on our current theories that are deeply rooted in the physics of small-scale processes. This gives rise to difficulties in representing nonlinear hydrological processes and their interactions at all scales across heterogeneous landscapes. In addition, calibrating and testing a model may not always assure its validity, since there are some inherent drawbacks in the approaches of parameter calibration and validation. We often over-parameterize our models to meet high accuracy levels, ignoring the equifinality problem (Beven 1992) that different parameter sets for a model might yield the same result during calibration, but distinctly different predictions when conditions are altered (Kirchner 2006). Ideally, a hydrological model for a watershed describes the hydrological system well enough even when condition changes occur. A model like a distributed one is characterized by hundreds of free parameters. The tuning process can be very tedious and time-consuming, and potentially lead to the equifinality problem because of an excessive number of free parameters. This issue can be a major problem for a large-scale watershed modeling as the dynamics of vegetation development over space and time make hydrologic processes more complicated and thus result in more difficulties in selecting good parameters.

Despite some limitations, a modeling approach is a good choice for the watersheds that have been well observed and monitored, for smaller ones in particular, while its applications are largely constrained for large-scale watersheds mainly due to lack of detailed data and empirical relationships between various processes for model calibration and validation.

In the following two sections, we present some promising alternative statistical and graphical methods for studying forest hydrology questions in large-scale watersheds. As shown in Fig. 6.2, selection of a suitable method or a set of methods depends on data availability and research objectives. We grouped methods according to the number of selected watersheds. One of the most notable advantages of these methods is their ability to assess the relationship between forest disturbance and hydrology with less data (only hydrological and forest data), compared with hydrological modeling.

Table 6.3 Major hydrologic models available for application in large-scale forested watersheds.

Name	Spatial Representation	Hydrological Process	Key Features
HSPF (Hydrological simulation program-Fortran)	Lumped	Surface water	Water quantity and quality (Choi and Deal 2008)
VIC	Semi-distributed	Surface, Subsurface	Available for large watershed or continent scale; Water and energy modeling; Simple unsaturated zone flow (Thanapakpawin et al. 2007)
SWAT (Soil water assessment tool)	Semi-distributed	Surface, Subsurface	Water quality and quantity; Better for agricultural watersheds; Simple unsaturated zone flow (Arnold et al. 1998; Franczyk 2008; Ma 2009)
LHEM (Library of hydro-ecological modules)	Semi-distributed	Surface, Subsurface	Hydrologic process and Ecological process(nutrient cycling, vegetation growth, decomposition); Simple unsaturated zone flow (Voinov et al. 2004)
DHSVM (Distributed hydrological soil vegetation)	Distributed	Surface, Subsurface	Water and energy balance; Forest watershed only; Simple unsaturated zone flow (VanShaar et al. 2002; Bowling et al. 2000; La Marche and Lettenmaier 2001; Stonesifer 2007)
MIKE-SHE	Distributed	Surface, water groundwater	Entire land phase of hydrological cycle; Available for wetland, forest, agriculture; Completely modeled unsaturated zone flow(3-dimention)(Sun et al. 2005)
RHESSys	Distributed	Surface, Subsurface	Hydrologic process; Ecological process such as nutrient and carbon cycling; Simple unsaturated zone flow (Tague and Band 2004)

6.4.2 Methods for a single watershed

A prerequisite for forest hydrology studies using a single large watershed is the availability of long-term hydrology and forest disturbance data. Also, given that large watersheds have a strong ability to buffer changes resulting from disturbances, it is critical to select a watershed that has experienced significant large-scale forest disturbance (e.g., ECA or EDA $>$ 20 or 30%). Climate data are also important for supporting data analysis and helping interpretation of findings from statistical tests.

Several methods are available for detecting changes on hydrology as a result of forest disturbance. These methods include statistical (e.g., non-parametric tests, time series analysis and Bayesian approaches) and graphical (e.g., flow duration curve or FDC) ones. While non-parametric tests are generally used for testing trends of statistical significance, Bayesian and time series analyses are made for both testing trends and forecasting. The followings are brief descriptions of these techniques, summarized in Table 6.4.

Table 6.4 Statistical methods for change detection of hydrological data series.

Methods		Specific requirements or important features	Application
ARIMA model		• Sample size⩾50 • Stationary data • Constant parameters	Monotonic trend detection; forecasting
ANOVA		• Output is assumed normally distributed • No assumption is required for the function type • Available for both continuous and discrete data	Step change detection
Bayesian methods		• Distribution-free • Change point unknown • Both available for small and large sample size • Parameters are viewed as random variables	Step change detection; forecasting
Non-parametric tests	**Spearman's rho**	• Distribution-free • Using the Pearson product moment as a parameter for measuring a correlation	Monotonic trend detection
	Kendall's tau/Mann-Kendal test	• Distribution-free • Serially independent • Using a correlation without parameter analogue	Monotonic trend detection

Continued

Methods		Specific requirements or important features	Application
Non-parametric tests	**Seasonal Kendal test**	• Distribution-free • Allowing for seasonality and autocorrelation in data	Monotonic trend detection
	Wilconxon-Mann-Whiteney test	• Distribution-free • Data with exact time of change known	Step change detection
	Kruskal-Wallis test	• Distribution-free	Step change detection
Parametric tests	**Student's *t*-test**	• Normal distribution of data • Equality of variance	Step change detection

*A step change, also called step trend, is a distinct change between two periods.

6.4.2.1 Time series analysis

Time series analysis is a powerful tool to analyze serially correlated hydrological data (Chatfield 1989). It can identify the underlying mechanism and structure represented by the data, and forecast future trends (Box and Jenkins 1976; Box et al. 1994). There are many methods of modeling time series, including Box-Jenkins ARIMA (Auto-regressive integrated moving average) models, Box-Jenkins multivariate models and Holt-Winters exponential smoothing (single, double, triple). The most extensively used one is ARIMA, either univariate or multivariate, generally referred to as an ARIMA (p, d, q) model where p, d, and q stand for the order of the autoregressive, integrated, and moving average parts of the model, respectively (Statsoft 1995).

However, before starting, it is essential to know that an ARIMA model is appropriate only for a stationary series or a series that can be made stationary by transformations such as differencing and logging (Box and Jenkins 1976). The minimum recommended size for input data is at least 50 observations (Hartmann et al. 1980; Tryon 1982). This implies that at least a 50-year set of records of hydrological data is needed for analysis on an annual time scale variable (Statsoft 1995).

Though time series analysis is widely utilized to detect patterns in hydrological data and to predict future trends, most of these studies are conducted using a single time series. To address forest disturbance-induced hydrological change, two time series, one on forest disturbance data series and the other on hydrological data series, are required (Lin and Wei 2008). Jassby and Powel (1990) recommended that cross-correlation be used to test statistical causal relationship between two data series. However, a whitening process to remove serial correlation within a data series must be conducted prior to cross-correlation analysis (Wei and Davidson 1998; Sun and Wang 1996; Tsai and Chai 1992; Law et al. 2005; Jordan et al. 1991).

6.4.2.2 Non-parametric methods

Non-parametric tests are useful for eliminating the influence of extreme or outlier values in data series. Given that hydrological data are typically neither independent nor normally distributed and often containing extreme values, distribution-free tests procedures such as non-parametric tests are more suitable for identifying changes in hydrological time series. Various non-parametric tests are available. Most are rank-based, such as Spearman's rho, Kendall's tau/Mann-Kendal test, and Seasonal Kendal test, Wilconxon-Mann-Whiteney test, Kruskal-Wallis test, Median change point test. Among these tests, Spearman's rho, Kendall's tau/Mann-Kendal test (Siegel and Castellan 1988) and Seasonal Kendal test are used for gradual change (trend) tests, whereas Wilconxon-Mann-Whiteney test, Median change point test and Kruskal-Wallis test detect step changes (Kundzewicz and Robson 2004). The Mann-Kendal test is the most widely used one for identifying trends in hydrological variables (Abdul and Burn 2006).

To assess forest disturbance effects on hydrology, the selection of a specific non-parametric test depends mainly on the availability of long-term forest data both before and after disturbance in the target watershed. When both long-time series of hydrological data and forest disturbance data are available, trend analysis methods like Spearman's rho and Kendall's tau can be adopted to detect the correlation between hydrological (e.g. annual mean runoff) and forest disturbance data series (e.g. ECA or EDA). The correlation coefficient reflects the potential impact of forest disturbance on the hydrology.

For example, we can use the Spearman's rho test to investigate the hydrological impact of forest disturbance. Suppose there are n years annual runoff data $(X1, X2, \cdots, Xn)$ and ECA data $(Y1, Y2, \cdots, Yn)$. First, we make an assumption that annual runoff is independent of EDA (forest disturbance). Second, we rank the two-time series of data separately, and compare the rank difference for each pair (referred to as d_i), and then calculate the Spearman correlation coefficient (r) using the following equation (Wackerly et al. 2001):

$$r = 1 - \frac{6\sum_{i=1}^{n} d_i^2}{n^3 - n} \tag{6.1}$$

The final step is to compare the calculated spearman correlation coefficient with critical values of a certain confidence level (e.g. 95% or 99%) to decide if there is a significant effect of forest disturbance on annual runoff.

For watersheds with relative short lengths of data records, we can split annual runoff data into two series: before and after disturbance, and then apply non-parametric tests (e.g. Wilconxon-Mann-Whiteney test, Kruskal-Wallis test) to testing if forest disturbance significantly affects the hydrology. Because precipitation variation over the two periods can confound the tests, the conclusions are valid only if the precipitation amounts are similar or if precipitation influences have been removed.

6.4.2.3 Bayesian methods

Bayesian methods are a group of approaches incorporating an old probability theory: Bayes' theorem or Bayes' law. They are often used to compute posterior probabilities from a prior distribution based on a subjective understanding of the time series being examined, or simply with an unknown prior distribution (Bernardo and Smith 2001). Traditionally, probabilities are assigned to random events according to their frequencies of occurrence or to subsets of populations as proportions of the whole. Methods such as Markov chain Monte Carlo and Laplace approximation are extensively applied to calculating the Bayesian probability for post distribution (Smith and Roberts 1993) and can be implemented by numerous free software tools including BUGS, CABeN and IDEAL (Lunn et al. 2000).

Bayesian methods have been applied to hydrological time series since the 1970s. They are widely used to detect statistically significant changes in runoff and precipitation data series, especially in addressing issues with great uncertainty (Rao and Tirtotjondro 1996; Perreault et al. 1999). At present, Bayesian methods are frequently used to evaluate hydrological models, and help determine the optimal value for hydrological parameters (Marshall et al. 2005; Engeland and Gottschalk 2002; Kavetski et al. 2006)

There are various types of methods named Bayesian. For examples, it can be a mathematical model such as a neural network (Khan and Coulibaly 2006; Zhang and Govindaraju 2000; Ha and Stenstrom 2003) or a regression model developed under a Bayesian concept (Agarwal et al. 2005; Raftery et al. 1997), or a revised ARIMA model with parameters predicted by a Bayesian procedure (Monahan 1983; Ray and Tsay 2002).

Artificial neural network techniques (ANN) are able to simulate nonlinear and complex systems with fewer requirements for input data than other procedures, and so are widely used in hydrological modeling (Gautam 2000). A Bayesian neural network is a common supervised neural network multilayer perceptron (MLP) incorporating Bayesian probability theory to assess the relationship between forest disturbance and hydrology. Unlike traditional neuron networks where only weights between neurons are adjusted during model training, a Bayesian neural network allows for the alteration of weights between neurons and the estimation of the distribution of parameters or bias, which can reduce the prediction errors.

A Bayesian network for modeling hydrological response to forest disturbance can potentially be constructed by use of, for example, annual runoff, yearly ECA and yearly precipitation data, from 1960 to 2000. In a similar way to hydrological models, the data are subdivided into two sets, one (from 1960 to 1990) for model calibration or training and the other (from 1991 to 2000) for validation. Through BN modeling, we can get simulated runoff data under the historical scenario. Then by changing the disturbance scenarios, for example, with ECA increased or decreased by 10% but with the same precipitation, we can get a new set of simulated runoff data. And the difference in simulated runoffs between the two scenarios can be attributed to forest disturbance.

6.4.2.4 Flow duration curve (FDC)

FDC shows the percentage of time that streamflow was equal to or exceeded a given amount. As a frequently used hydrograph, FDC describes a watershed's flow regime by magnitude, frequency, duration, timing and rate of change of runoff at a given point (Brown et al. 2005). Its shape provides substantial information on the influence of watershed characteristics (watershed size, vegetation cover, topography and land use type), precipitation pattern, and human activities (e.g. dam constructions, water abstraction). For example, when drought or large-scale land cover conversion occurs, the shape of FDC is altered. Thereby, through comparison of FDCs before and after changes, we can investigate the likely impact of disturbance on runoff with direct, visual evidence. This can be very useful in the initial design phase of a watershed study. It also assists us with better interpretation and presentation of changes as detected by statistical methods (Kundzewicz and Robson 2004).

It is important to note that when making comparisons between daily, monthly, and annual FDCs before and after disturbance, it is a sound practice to select periods with comparable amounts of precipitation. This reduces climate-induced variation (Burt and Swank 1992). The main deficiency of FDCs is their limited ability to demonstrate the magnitude of change. But still, FDC can serve as a good exploratory, complementary tool for studying disturbance effects on hydrology, particularly on hydrological variables over short-term durations (e.g. daily, monthly flows, timing of flow changes).

With large-scale forest hydrology research based on a single watershed, it is always necessary to integrate several approaches to gain a higher level of confidence in one's conclusions. This strategy will not only increase reliability of the results, but also enable us to better understand the possible causes. For example, Lin and Wei (2008) combined time series analysis, Neyman-Pearson and non-parametric tests (Spearman's rho, Kendall's tau) to study the impacts of forest harvesting on annual and seasonal peak, mean and low flows in the Willow watershed in the interior of British Columbia, Canada. They selected ECA as a forest disturbance index to examine cumulative impacts on hydrology. Their decision criteria were that if all three types of tests showed the same results, the conclusions would be clear, otherwise inconclusive. Another example is the study on the impacts of deforestation on annual and seasonal runoff (50-year data) in the Tocantins River, Southeastern Amazonia (Costa et al. 2003). They applied a combination of t-test, ANOVA and hydrographs. A two-way ANOVA was used to account for climate variability, while the hydrographs were used to interpret runoff variations caused by land cover changes.

The main disadvantage of a single watershed study lies in the difficulty in separating hydrological variations caused by variables such as climate variability and watershed topographies in addition to forest disturbance. A second disadvantage is the long-term data requirement. In most large forested watersheds, long-term data on hydrology and forest disturbance history are unavailable.

6.4.3 Methods for multiple watersheds

The paired-watershed experiment is commonly accepted as a traditional, classic approach worldwide to study relationships between forest changes and water yield through comparisons between two adjacent watersheds (one as a controlled and the other treated) in a way that the influence on runoff from other factors is removed. But this approach is suitable only for relatively small watersheds (<100 km^2). For large-scale watersheds, it is impossible to locate a reference one to make a pair.

In the absence of strictly controlled large watersheds, Buttle et al. (1996) proposed a concept named "quasi-paired watersheds" to study the effects of forest harvesting on streamflow using several large-scale watersheds in Ontario, Canada. In this concept, the watersheds with greater disturbance served as the treated watersheds while the nearby least disturbed watersheds were considered as the controlled or partially controlled ones. In terms of the concept of quasi-paired watersheds, statistical and graphical methods such as DMC (double mass curve), FDC, linear regression and ANOVA can be applied. In short, if we can find a series of comparable large watersheds that experienced different levels of disturbance ranging from least to most disturbed, the hydrological responses to forest disturbances can be analyzed.

6.4.3.1 Double Mass Curve (DMC)

DMC is of great use in detecting hydrological impacts of land cover changes. It was first designed to check and adjust inconsistencies in hydrological data induced by changes in the methods of observation or data processing (Wigbout 1973). A break in this curve indicates a change in the relationship between the two records.

When applying DMCs to quasi-paired watersheds, the cumulative runoff from the more disturbed watershed is plotted against the cumulative runoff from the less disturbed one. Prior to disturbance, the plot should produce an almost straight and predictable baseline while a deviation from this baseline is expected to occur after disturbance. This deviation is attributed to the hydrological change caused by disturbances. Fig. 6.4 further illustrates the application of DMC for detecting the effects of forest disturbances on hydrology using a hypothesized example of two quasi-paired watersheds with 30-year (e.g. 1971-2000) cumulative annual streamflow data. Assuming that a large-scale harvesting occurred in the early 1980s in the more disturbed watershed, an inflection was found in 1985, about 5 years after the disturbance. This post disturbance deviation between the expected line and observed one was caused by the large-scale disturbance. This effect can be calculated by the following formula:

$$E_{\mathrm{d}} = 100\% \times (Q\mathrm{e} - Q\mathrm{o})/Q\mathrm{e} \tag{6.2}$$

where E_{d}, Qe, and Qo stand for disturbance effect, expected streamflow and observed streamflow, respectively.

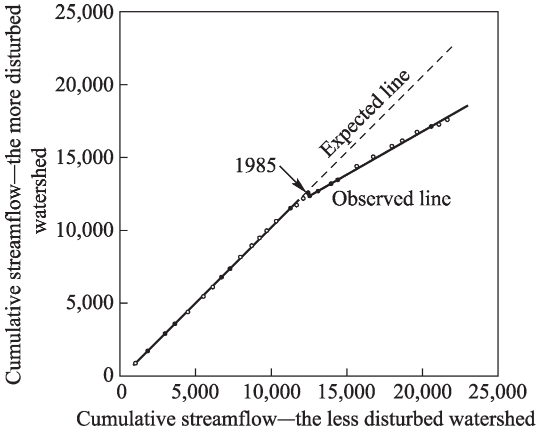

Fig. 6.4 A hypothesized example of DMC (double mass curve) application of cumulative streamflow to two quasi-paired watersheds (more disturbed vs less disturbed watersheds).

Application of DMCs to assessing hydrological impacts of forest disturbance in large-scale watersheds requires long-term data (e.g. 20 years or longer) on both hydrological records and forest disturbance history. In addition, it is also important that significant disturbance events occurred near the middle of a long-term record for a more disturbed watershed so that there is sufficient length of data available before and after the disturbance events.

6.4.3.2 Linear regression

Linear regression may be the most commonly used method in quasi-paired watersheds, simply because of its ease use as well as its ability to provide direct prediction of any magnitude change. For example, Jones et al. (1996) detected peak flow responses to clearing-cutting in 3 pairs of adjacent large watersheds in the western Cascades of Oregon (U.S.A) by the use of linear regression. They found that peak discharge increased by 100% over the past 50 years. Using the similar approaches, Major and Mark (2006) examined the hydrological response to a volcano eruption in Washington and discovered a short-lived increase in autumn and winter peak flows after the eruption. More examples are presented in Table 6.5.

Table 6.5 Examples of the quasi-paired watershed studies.

Titles	Disturbance type	Methods	Key Results
Boreal forest disturbance and streamflow response, northeastern Ontario (Buttle and Metcalfe 2000)	Harvesting dominated	DMCs FDCs	Limited streamflow response to land cover change was found; No definitive changes in water year runoff or peak flow magnitude and timing
Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon (Jones et al. 1996)*	Clear-cutting Road construction	Linear regression	Forest harvesting has increased peak discharge by 100% over the past 50 years
Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon: A second opinion (Thomas and Megahan 1998)*	Clear-cutting Road construction	Linear regression (Logarithm transformed)	No effect of cutting on peak flows in one of the basin pairs; Inconclusive results for the other 2 basin pairs
Peakflow responses to forest practices in the western cascades of Oregon, USA (Beschta et al. 2000)*	Clear-cutting Road construction	Multiple linear regression	No strong evidence for peak flow increases in large basins
Peak flow responses to landscape disturbances caused by the cataclysmic 1980 eruption of Mount St. Helens, Washington (Major and Mark 2006)	volcano Eruption	Linear regression	A short-lived (5-10 yr) increase in the magnitudes of autumn and winter peak flows

*The large basin sizes range from 60 to 600 km^2. Though these studies use smaller sizes than our recommended size for large-scale watersheds, they are included here for showing how to use regression methods to study larger watersheds.

Table 6.5 clearly shows that an application of linear regression to quasi-paired watersheds can provide an approximate evaluation of potential magnitude change on hydrological variables as a result of forest disturbances. For a more robust estimation, long-term data (e.g. 20 years or longer) are crucial and there should be contrasting disturbance levels between quasi-paired watersheds so that hydrological impacts can be detected if any. Moreover, quasi-paired watersheds should have similar climate conditions, watershed characteristics, land use types and topographies, all of which help to reduce confounding variables. Compared with the single watershed study, the multiple watersheds approach may provide better and more straightforward assessment capacity to detect a change in magnitude in hydrology. However, selecting quasi-paired watersheds containing long-term hydrological data records and contrasting forest disturbance levels is a challenging task.

6.5 Future directions

Due to complicated interactions among climate, forest disturbance and water, large-scale forest hydrology study is limited and challenging. With more data available and application of more advanced technologies and statistical tools, opportunity in this research direction will be improved.

6.5.1 Long-term data collection

Long-term data are critical for large-scale watershed researches. Without long-term data, it is impossible to conduct large-scale watershed studies through statistical approaches (e.g. time series analysis) using one large single watershed. Lack of long-term data can also impose great difficulty in assessing the effects of forest disturbance on hydrology using multiple large-scale watersheds. No matter whether we use hydrological models or statistical methods when selecting research methods, it is certain that longer term data can produce more persuadable results or detectable changes caused by forest disturbance. However, there are two issues to consider in long-term hydrological and forest disturbance history data. First, hydrological data are collected in many large-scale watersheds in the world, but hydrometric stations are generally limited in forest-dominated watersheds and they are short-term. More importantly, many gauged watersheds are regulated for water storage (reservoirs, etc.) and power generation. The hydrological records on regulated systems must be naturalized prior to data analysis, and those records also make it impossible to analyze peak or/and low flows. Second, data on forest disturbance history in large-scale watersheds are generally not long enough. Remote sensing technology is a powerful tool for capturing vegetation changes over a large scale, but satellite images are only available after the 1980s. In addition, very few jurisdictions have the capability to manage GIS-based forest disturbance history data accumulated from millions of disturbed blocks or polygons over a long time. Because of these issues or challenges, it is not surprising that large-scale watershed studies are rather limited, and most of them have to rely on modeling.

To allow for more large-scale watershed studies in the future, more long-term data (forest disturbance, hydrology and climate) must be collected. Given that most large rivers or watersheds have been dammed for water storage and power generation, future focuses should be directed to those rivers in national parks or conservation areas or in regions with lesser influence by human development. Moreover, more advanced technologies such as remote sensing, and GIS should be used as much as possible to increase efficiency in data collection and management.

6.5.2 Integrated surface water and groundwater research

Almost all the studies on this topic dealt with river streamflow, ignoring interactions between surface water and groundwater. A watershed with a size greater than 1,000 km^2 likely contains regional aquifers, and thus surface water and groundwater interactions. In some large watersheds where groundwater may contribute to a greater portion of streamflow, hydrological responses to forest disturbance are expected to be less in amount and slower to appear because groundwater may contribute a buffering role in streamflow change. Compared with small-scale watersheds, hydrological processes in large-scale watersheds are more complex and influenced by greater heterogeneity of both land surface and geology. Inclusion of a groundwater component in a large-scale watershed study not only helps us understand hydrological behavior and characteristics, but also provides a powerful mechanism for a more complete evaluation of the relationships between forest disturbances and hydrology.

To include groundwater in a large-scale watershed hydrology study, long-term groundwater data from sufficient observation wells are needed. Such data, together with geochemistry and isotopic data can greatly assist in the integration of surface water and groundwater flows, and their responses to forest disturbances.

6.5.3 Integrated research approach

As there is no a single satisfactory, commonly-accepted method for studying large-scale forest hydrology, it is logical to take an integrated or combined approach of two or even more methods. Each method has unique strengths and weaknesses, and serves different purposes. For example, hydrological models can estimate the magnitude of hydrological change caused by forest disturbances, but for model calibration, they require huge data sets on hydrology, climate, vegetation, soil and land use, and their simulation results may be difficult to validate due to inadequate knowledge of empirical relationships between processes and between components in large-scale watersheds. Statistical methods are able to address this issue because they require less detailed data, but the majority of them can merely provide us with significance tests of trends and correlations, and some of them (e.g. time series analysis) may even need extraordinarily long record sets. Graphical methods such as DMC are likely to provide us with an overall approximation of hydrology change caused by forest disturbance for the whole watershed, but the spatial heterogeneity cannot be fully accounted for. Furthermore, all these methods fail to include an important component of water—groundwater. Given the ever increasing demand for research in large watersheds and limited data availability for most large-scale watersheds in the world, we expect there will be a growing interest in a combined research strategy to study the relationships between hydrology and forest disturbance for large-scale watersheds.

6.5.4 Inclusion of climate change

Global climate change is dramatically influencing our forests, forest disturbance and hydrological cycles, which makes forest ecosystems more sensitive and fragile. Climate change, as an important disturbance driver, can alter forest disturbance regimes such as frequency, intensity and timing. The recent outbreak of mountain pine beetle infestation in BC, Canada is a good example. In large-scale forested watersheds, climate change impacts can be more pronounced and direct. Several recent studies assessed the relative contribution of climate variability and land use changes on streamflow (e.g. Tuteja et al. 2007; Zhang et al. 2008; Zheng et al. 2009). As for groundwater, the inclusion of climate change into large-scale forest hydrology studies will provide a more complete context to evaluate the relationship between forest disturbance and hydrology. This is particularly relevant as climate change is both a direct and an indirect (by affecting forest disturbance regimes) factor on hydrology (Fig. 6.5.)

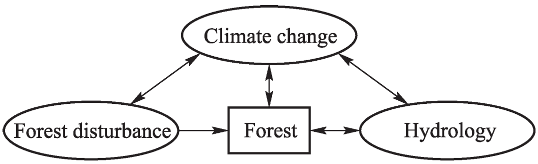

Fig. 6.5 A direct and indirect role of climate change on hydrology.

The direct effects of climate change on forest growth (e.g. primary net production), forest disturbance (e.g. wildfire, drought and insect infestation), carbon budgets and the hydrological cycle have been intensively examined particularly in the past decade. However, the indirect effect of climate-change-induced disturbance on hydrology has been almost untouched. This issue is important for designing our adaptation strategies to address the impacts of climate change. We expect research strategies that include climate change in a large-scale forest hydrology study will likely combine hydrological models with statistical techniques.

6.6 Conclusions

There is significantly less research on forest disturbances and their effects on hydrology in large-scale watersheds than in small-scale ones. This is mainly because of the difficulty in locating large reference watersheds for the paired watershed approach commonly applied in to small-scale watersheds. Other factors include insufficient long-term hydrological and disturbance data, more complexities and landscape heterogeneities, and a lack of commonly accepted research methods. However, the need to conduct large-scale watershed research to support landscape or watershed-scale planning and management is growing, particularly in the context of climate change typically operating on a large scale.

We propose that a large-scale watershed be defined as 1,000 km^2 or greater in area. In addition, we also suggest a concept of equivalent disturbance area or EDA as a means to quantify cumulative forest disturbances in large-scale watersheds. Those two concepts or terms can provide a consistent frame for future research of this kind and to support comparisons between studies.

Although there is a lack of a commonly accepted method to study large-scale forest hydrology, various methods are available, including hydrological modeling, statistical techniques and hydrological graphics. Selection of a suitable method is largely dependent on research purposes, data availability and the number of available watersheds. We conclude that a suitable research strategy is to use combinations of two or more techniques so that more robust conclusions can be reached than from the use of a single technique alone. Future directions should focus on a continuation of long-term data collection, and inclusion of groundwater and climate change into large-scale forest hydrology studies.

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Chapter 7 Software Applications to Three-Dimensional Visualization of Forest Landscapes — A Case Study Demonstrating the Use of Visual Nature Studio (VNS) in Visualizing Fire Spread in Forest Landscapes

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Abstract
Three-dimensional (3D) visualization is a useful tool that depicts virtual forest landscapes on computer. Previous studies in visualization have required high end computer hardware and specialized technical skills. A virtual forest landscape can be used to show different effects of disturbances and management scenarios on a computer, which allows observation of forest landscapes without time limitations. This chapter lists updated methods and software used for 3D visualization of forest landscapes, and demonstrates a fire visualization using Visual Nature Studio and standard computer hardware. Elevation and vegetation data were used to create a representation of the New Jersey pine barrens environment. Photographic images were edited to use as image object models for forest vegetation. The FARSITE fire behavioral model was used to model a fire typical of that area. Output from FARSITE was used to visualize the fire with tree models edited to simulate burning and flame models. Both static and animated views of the fire spread and effects were visualized. The visualization method was evaluated for advantages and disadvantages. VNS visualization is realistic, including many effects such as ground textures, lighting, user-made models, and atmospheric effects. However VNS has additional hardware requirements in terms or

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memory or hard drive space and sometimes renders images slowly.

Keywords
Visualization software, FARSITE model, ArcGIS, VNS.

7.1 Introduction

Visualization is defined as any technique for creating images, diagrams, or animation in order to convey a message. Advances in computer hardware and software over the past 30 years have allowed researchers to simulate and visualize complex forms, phenomena, and dynamics, such as plant growth or changes in atmospheric conditions, in natural systems (Ervin and Hasbrouck, 1999). Visualization that previously required specialized computer systems and hardware can now be done on more affordable desktop/laptop systems. With this increased accessibility, visualization has become a tool available to even more researchers as a means of analyzing data.

In the field of landscape management and forestry, visualization can be used to demonstrate and analyze data from geographic information systems (GIS) or remotely sensed imagery such as aerial photographs or satellite images. Through the commercialization of hybrid 2D/3D visualization software such as Bryce 3D (DAZ Productions, 2009), World Construction Set and Visual Nature Studio (3D Nature Inc., 2009), and VistaPro (Virtual Reality Laboratories Inc., 1993), 3D modeling of landscapes possessing a high degree of realism, from different viewpoints, and having animation paths is now possible (McGraughey, 1998; Muhar, 2001). Due to the release of these programs, the use of visualization has become an important tool for analyzing existing forest landscape resources and for assessing the impact of proposed management practices (Lange, 1994; Orland, 1994b; McCarter, 1997; McGaughey, 1998). Furthermore, it can aid the understanding of successional dynamics and spatial patterns within a forest ecosystem. It may also be of aid to forest managers when selecting management practices to more efficiently utilize forest resources.

With the ability to produce realistic representation of data, visualization can play an important role in the land management decision making process. An important question that can be raised is what level of realism is necessary to draw meaningful conclusions from visualized images. Understanding how people observe and process visualization can assist in making them better present data and natural phenomena.

Oh (1994) studied the effects of representational image quality on perception. In scoring of representative images ranging from a simple wire frame view to fully digitized, color photographs, the wire frame and simple views were rated much lower. Likewise, these simpler views were also rated lower among observers having less knowledge of visualization and of the site.

Further studies by Bergen et al. (1995) compared scenic beauty ratings based on photographs to those obtained from computer-generated visualization of the same scene. Overall, the correlation between the rendered scenes and photographs were not significantly different, but the correlation with a smaller subset of five views had a greater significance where the rendered scenes played a more important role in the beauty rating. Computer-rendered visualization may have played a role in the preliminary assessment of the scene, but final quality decisions were found to be best done using photographs.

Daniel and Meitner (2001) compared different levels of realism/abstraction among visualized scenes. The most realistic scenes had full 16 bit color images with each successive scene having lower realistic qualities ranging from 4 bit color to black and white sketches. Visualized scenes with the highest realism had the highest correlation with perceived beauty. Likewise, it was also found that each reduction in realistic quality resulted in a corresponding reduction in correlation with perceived beauty.

The visualization methodologies identified in these studies play a crucial role in making useful visualized scenes. Moreover, software applications to visualization must take these processes into account in order to make them effective.

7.2 Forest landscape visualization

Landscape visualization requires a combination of several aspects scuh as software, geospatial data, and methods of representing data in order to be successful. A good visualization application will provide features that can accurately render landscape scenes in a realistic manner. Furthermore it should also include realistic models or the ability to import such models in order to represent features of the landscape being modeled. Geospatial data is necessary for modeling landscape features accurately. It can allow placement of features such as forests, lakes, roads, and buildings that can allow the user to identify and draw crucial information from the scene being presented. Visualization techniques can be simple things such as rendering features such as lakes or forest with accepted colors such as blue and green. More advanced methods can include using cloud models or the use of light and shadows. Techniques such as these acts to enhance the realism of the scene in order to allow the view to make a more personal connection to it, which in turn may help them draw more meaningful information.

7.2.1 Review of forest landscape visualization software

There are various different applications to landscape visualization (Table 7.1). However, each of these software packages is written with different goals in mind and each has different advantages and disadvantages. Table 7.1 lists the critical features for 3D visualization comparisons of various 3D visualization software packages.

In a comparative study, Karjalaimen and Tyrväimen (2002) evaluated the three applications of MONSU (Pukkala, 1998), Smart Forest (Orland, 1994a), and FORSI (Plustech Ltd.) for their ease of use and suitability for producing landscape visualization.

MONSU (Fig. 7.1) produces automated computer line drawings based on site and tree parameters, rendering trees accurately in both size and shape as 2D or 3D images. However, understory elements are not present and ground elements are drawn with different colors. Landscape elements such as bushes, uniquely shaped trees, and buildings are not present either. The lack of these

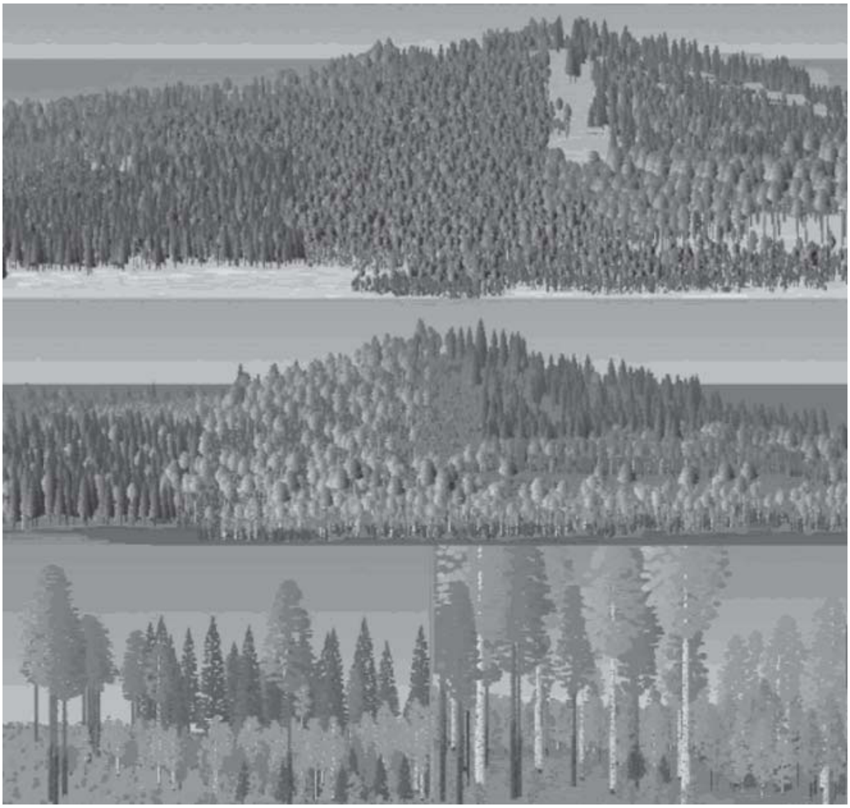


Fig. 7.1 Long and near-distance views produced using MONSU (Karjalaimen and Tyrväimen, 2002).

Table 7.1 Comparison of commonly used 3D forest visualization software packages (updated from Song et al., 2006)

Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
ArcGIS 3D Analyst/ ArcGlobe/ ArcScene	Image draping	All scales	PC-Windows, Macintosh	V 8.2 $2,500	http://www.esri.com/software/arcgis/extensions/3danalyst/ Source: ESRI	Real time interaction, Easy link between visualization and data information
Blueberry3D	Image draping and geometric modeling	All scales	PC-Windows, Macintosh	V 2.0 $2,490	http://www.blueberry3d.com/ Source: Sjoland & Thyselius Virtual Reality Systems AB	A procedural geometry engine, Allows rapid development and visualization of realtime 3D databases in high resolution for the Military Simulation Industry, Bridging the gap between high resolution 3D content and realtime display performance
Digital Landscapes	Geometric modeling	landscape	Windows NT; Silicon Graphics IRIX	500,000 Japanese yen	http://www.jfp.co.jp/landscape_e/dland_top.htm JFP Morioka, Japan	Uses terrain data from Japan's National Geographical Institute
EnVision	Geometric modeling	Stand or landscape	PC-Windows	Free	http://forsys.cfr.washington.edu/envision.html Source: Pacific Northwest Research Station, USDA Forest Service	Easy to use, Simple individual tree forms so low photorealism
Enviro (3D runtime environment)	Image draping and geometric modeling	Landscape	PC-Windows, Macintosh	Free	http://www.vterrain.org/ Source: Virtual Terrain Project (VTP)	Allows rapid construction of an interactive 3D visualization

Continued

Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
FORSI	Image draping and geometric modeling	Landscape	PC-Windows	No longer available	Source: Finland company Plustech Oy, a subsidiary of the John Deere Company's *Timberjack* Division.	Achieves a high degree of fidelity in rendering forest scenery with texture-mapping techniques, Does not allow the viewer to move around in a forest setting or to interact with underlying data
GenesisIV	Image draping and geometric modeling	Landscape	PC-Windows	Educational: $50 Enterprise: $300	http://www.geomantics.com/genesis4.htm Source: Geomantics	Fast rendering, Low price, Good teaching tool; Simple individual tree forms, Low photorealism
Google Earth	Image draping	Landscape	PC-Windows	Standard: Free Pro: $ 400	http://earth.google.com/ Sorce: Google™	Displays satellite images of varying resolution of the Earth's surface, Allows users to visually see things like houses and cars from a bird's eye view
Landscape Explorer	Image draping	Landscape	PC-Windows	$50; Educational: Free	http://www.geomantics.com/le2000.htm Source: Geomantics	Realtime 3D map/landscape renderer
Landscape management system (LMS)2	Geometric modeling	All scales	PC-Windows	Free	http://lms.cfr.washington.edu/ Source: University of Washington	Designed to aid in landscape level management, and linked to forest growth model, Simple individual tree forms, Low photorealism

Continued

Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
Light Detection and Ranging (LIDAR) based forest visualization (LI	Geometric modeling	Stand or landscape	Specific equipment, such as CAVE	Expensive, Flight costs: \$100,000 + Software: \$1,500 to \$3,000	http://www.cfr.msstate.edu/forestry/sitl/sitl_ikuko/vis_ikuko.htm Source: Visualization, Analysis, and Imaging Laboratory (VAIL) at Mississippi State University	Expensive, Real time interaction
LViz (LIDAR/ALSM 3D Visualization tool)	Image draping	All scales	PC-Windows	Free	http://lidar.asu.edu/LViz.html Source: Active Tectonics Research Group at Arizona State University	Designed for 3D visualization of LiDAR / ALSM point and interpolated data, Offers import of LiDAR point cloud data (delimited text file) or interpolated surfaces (in ASCII or Arc ASCII grid formats)
MapInfo	Image draping and geometric modeling	All scales	PC-Windows	Pro V8.0: \$1450	http://www.mapinfo.com/products/applications/mapping-and-analytical-applications/mapinfo-professional Source: MapInfo	Windows®-based mapping and geographic analysis application, Designed to easily visualize the relationships between data and geography
MONSU	Geometric modeling	Landscape	PC-Windows	Free	http://www.monsu.net Source:: University of Joensuu, Fenny	Software product for multiple-use forest planning by producing a set of management schedules on the basis of user-defined instructions

Continued

Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
Natural Scene Designer	Geometric modeling	Stand or landscape	PC-Windows, Macintosh	V 4.0 $139 Pro 4.0 $329	http://www.naturalgfx.com/nsdwin.htm Source: Natural Graphics	Produces 360 degree panoramas from any location
PAVAN (Performing and Virtual Arts Northwest)	Image draping	Stand or landscape	PC-Windows		www.pavanw.com Source: DataView Solution	A Virtual Reality Modeling Language (VRML) compiler and project management toolkit for use in conjunction with MapInfo software
Persistence of vision raytracer (POV-Ray)	Geometric modeling	All scales	Many platforms	Free	www.povray.org Source: povray.org	A general purpose ray-tracing system, Capable of producing detailed, photorealistic images of geometric models
Rapid Terrain Visualization (RTV) LIDAR Toolkit	Geometric modeling	Stand or landscape	PC-Windows	Free	http://rockyweb.cr.usgs.gov/html/extract/ Source: SAIC	Extension of ESRI ArcView 3.2 requiring the Spatial Analyst extension, ArcView's 3D Analyst extension is required for exporting 3D Shape files
SmartForest II	Geometric modeling	Stand or landscape	UNIX (SGI or IBM-RS6000 with OpenGL) and PC-Windows	Free	http://www.imlab.psu.edu/smartforest/index.html Source: University of Illinois at Urbana-Champaign	3D ecosystem simulator, Requires detailed data, Visualization capacity is around 5,000 stems

Continued

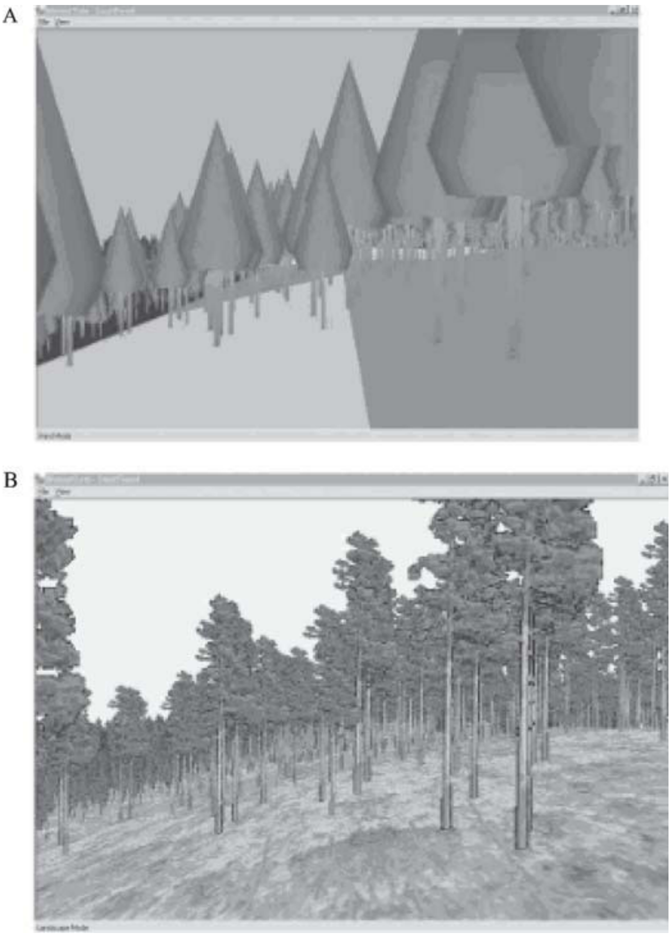
Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
UTOOLS and UVIEW	Geometric modeling	Stand or landscape	PC-DOS	Free	http://faculty.washington.edu/ mcgoy/utools.html Source: Pacific Northwest Research Station, USDA Forest Service	Similar to Stand Visualization Simulator http://forsys.cfr.washington.edu/ svs.html, Functions mainly at landscape scale
VirtualGIS	Image draping and geometric modeling	Stand or landscape	PC-Windows	£ 2,140	http://gi.leica-geosystems.com/LGISub1x39x0.aspx Source: ERDAS IMAGE	Visual analysis tool that offers GIS functions and capabilities in a 3D environments, Creates accurate 3D interpretations of your projects for interactive presentations
VirtualEarth	Image draping	Landscape	PC-Windows	Free	http://www.microsoft.com/virtualearth/platform/default.aspx Source: Microsoft®	Integrated set of services that provides quality geospatial data, Rich images, Dependable performance, Visualizes data to provide immersive end-user experiences
VistaPro	Geometric modeling; image draping	Landscape	PC-DOS, PC-Windows, and Macintosh	$49	http://www.vendornation.com /*ws4d-db-query-QuickShow?vp001 Source: VendorNation	3D landscape rendering program

Continued

Software Package Name	Visualization Technique	Scale	HW – OS	Cost	Additional Information Reference	Comments
WCS and VNS, with add-on software of "Scene Express", "NatureView Express, a real-time 3D viewer", and "Forestry Edition of VNS" for more efficient use of the powerful forestry features in VNS.	Image draping and geometric modeling	All scales	PC-Windows, Macintosh	WCS 6: \$500; VNS 2: \$2475; Scene Express: \$399 for WCS and \$699 for VNS; NatureView Express: Free; Discount for Education	http://www.3dnature.com Source: 3D Nature LLC	Photorealistic visualization, High compatibility with GIS data, Linkage to GIS data means that different vegetation or land use polygons will be rendered according to their attributes, Libraries of foliage objects and textures, pre-built components, and sample projects
World Builder	geometric modeling	All scales	PC-Windows	Standard: \$399 Pro: \$999	http://www.digi-element.com/wb/index.htm Source: Digital Element	Photorealistic visualization, Panorama rendering
World Perfect	Image draping and geometric modeling		PC-Windows	\$15,000	http://www.metavr.com/products/worldperfect/worldperfect.html Source: MetaVR	Terrain-and-content generation system for creating geographically large-scale detailed virtual worlds, Creates highly realistic 3D environments quickly and efficiently from imagery, elevation, and feature source data

features being rendered makes MONSU a poor choice for visualizing landscapes with special scenic beauty. The application's main strength comes from the ability to use forest inventory data very efficiently, being compatible with available forest inventory and satellite data. MONSU can simulate movement through a visualized forest by drawing scenes from points along a selected path; however, at the time of publication, real time movement was not possible due to hardware constraints.

Smart Forest (Fig. 7.2) is an interactive, 3D software package that has a management and landscape mode. Management mode provides a simplified view of the forest for quick queries of individual trees and forest stands in order to analyze the data. Landscape mode renders much more detailed and realistic scenes, with trees and water being presented as texture-mapped objects and ground details wrapped with 2D images generated from digitized photos. Smart Forest also allows user-defined heights to view scenes at dif-



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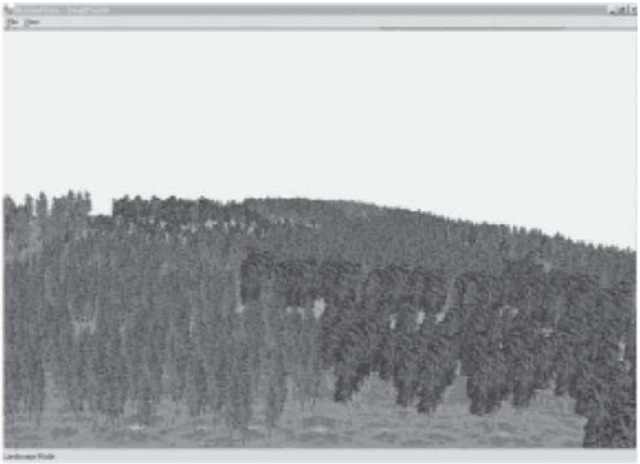

Fig. 7.2 Near-distance view in manager mode (A), long-distance view (B), and landscape mode view (C) produced using Smart Forest (Karjalaimen and Tyrväimen, 2002).

ferent levels and movement within the scene for a real time walk-through of a virtual forest. However, the application optimizes graphic quality in order to produce smooth moving walk-through animation. Due to these optimizations, the ability to accurately represent local forest scenes when using Smart Forest's real time movement is greatly reduced.

FORSI (Fig. 7.3) is a smaller landscape visualization application designed mainly for Finnish organizations; however, it was written for realism and flexibility. Forest elements are represented by 2D objects generated from digitized photos.

Because these objects are digitized from photographs, high quality images can be produced that are only limited to the color depth and resolution of the original photograph. Likewise, new objects can easily be added to the standard library of tree and forest elements by simply digitizing them from a photograph. FORSI also has the ability to illustrate seasonal and atmospheric effects. This allows for more realistic scenes due to differing light and sky conditions. Movement simulation in FORSI is very similar to that in MONSU. Points are manually selected and scenes are drawn along that path in order to simulate movement throughout the forest.

Of these three software packages, Smart Forest and FORSI are commercial packages, whereas MONSU is free for use (Karjalaimen and Tyrväimen, 2002). Currently only Smart Forest and MONSU have websites (http://www.imlab.psu.edu/smartforest/ and http://www.monsu.net/Englishmonsu.htm); however, neither is available for download at this time. Due to their age, these software packages are no longer in use.

Song et al. (2006) investigated general approaches to landscape visualization and presented some information on software applications, finding that

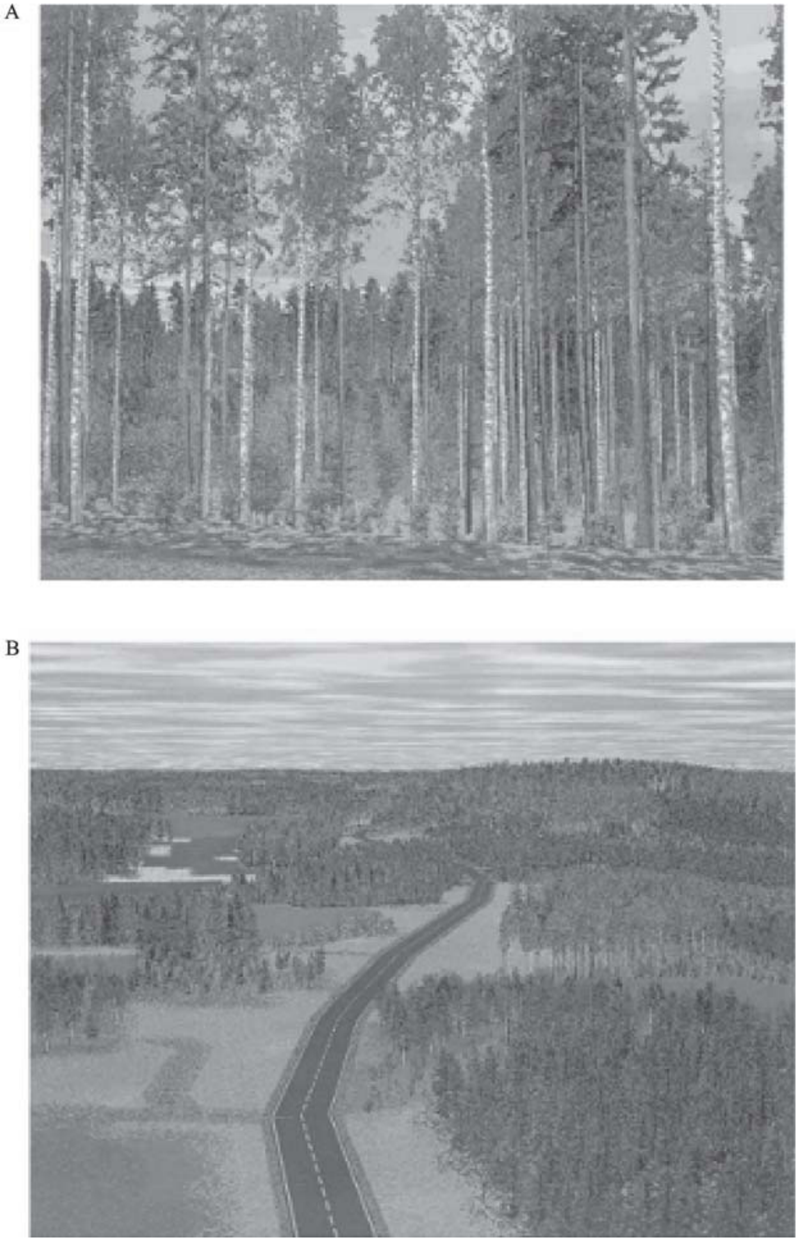

Fig. 7.3 Near-distance (A) and long-distance (B) views produced using FORSI (Karjalaimen and Tyrväimen, 2002).

one of the key challenges facing software applications for use in visualization is real time interaction. However, many software packages are improving on this. Commercial software packages such as Ecomodeler/Ecoviewer (Viewscape 3D Graphics Ltd) can display visualization with point of view and fly-over capabilities in real time, with options to include management activities and the ability to navigate freely (Song et al., 2006). Another problematic aspect of many visualization applications is the lack of photorealism. However, some freeware applications, such as Persistence of Vision Raytracer (POV-Ray, Povray.org), are capable of producing detailed, photorealistic models. Development of software with real time user interaction and photorealistic modeling helps to provide better tools for the use of visualization in landscape management.

7.2.2 Case study: Forest landscape visualization of fire spread

Wang et al. (2006) created a visualization of the landscape of the Chequamegon National Forest in northern Wisconsin using publicly available data sources such as forest inventory analysis (FIA, USDA Forest Service) and GIS data. The visualization was done using Visual Nature Studio (VNS, 3D Nature Inc.), a relatively new visualization software package. Mathematical models were then applied to the forest data to determine changes in future succession and growth. The resulting data were applied to the initial forest visualization to show changes in forest stand structure and composition that may occur due to harvesting or some other disturbance events. Resultings visualizations that came from these data were time lapsed images showing the changes in the forest over extended periods of time.

We adopted the Wang et al. (2006) approach to develop visualization for a wildfire event in the pine barrens of New Jersey. An initial forest environment was constructed using public topographic, vegetation, and GIS data for the study area. Data output from the FARSITE model (Miles et al., 2001) was then applied to the visualization in order to show the spread and effects of the fire. FARSITE is a fire behavior and growth simulator program used in the prediction of fires based on initial fuel, weather, and topographic conditions. It is widely accepted and used by fire behavior analysts from a wide variety of agencies, including the USDA Forest Service, USDA National Park Service, and the USDI Bureau of Land Management. The program accepts vegetation, fuel load, weather, and topological data in the form of text files and raster images. Output from the program consists of ArcGIS raster images with data pertaining to fire line intensity, flame length, and other fire characteristics. Using FARSITE, managers can predict the occurrence of both surface and canopy fires along with fire behavior based on factors such as firebreaks and the influence of weather.

7.2.3 Visualization using visual nature studio

The computer used for working with VNS was a custom built system with an AMD Athlon X2 4200+ dual core CPU, 2 gigabytes of RAM, and an nVidia GeForce 7600 GS graphics card. An updated version of VNS version 2.7 was used for the study. Add-on tools such as Scene Express or the Forestry Edition were not installed or used.

A 2,590 hectare (10 square miles) section of the Cedar Bridge area in the New Jersey pine barrens was selected as our study site. This area consisted primarily of a mature pitch pine canopy with a small number of hardwood species and high shrub loads present in the understory. Most of the area is forested; however, there are some small areas with buildings and several highways. There is also one large lake and several smaller water bodies within the area (Fig. 7.4).

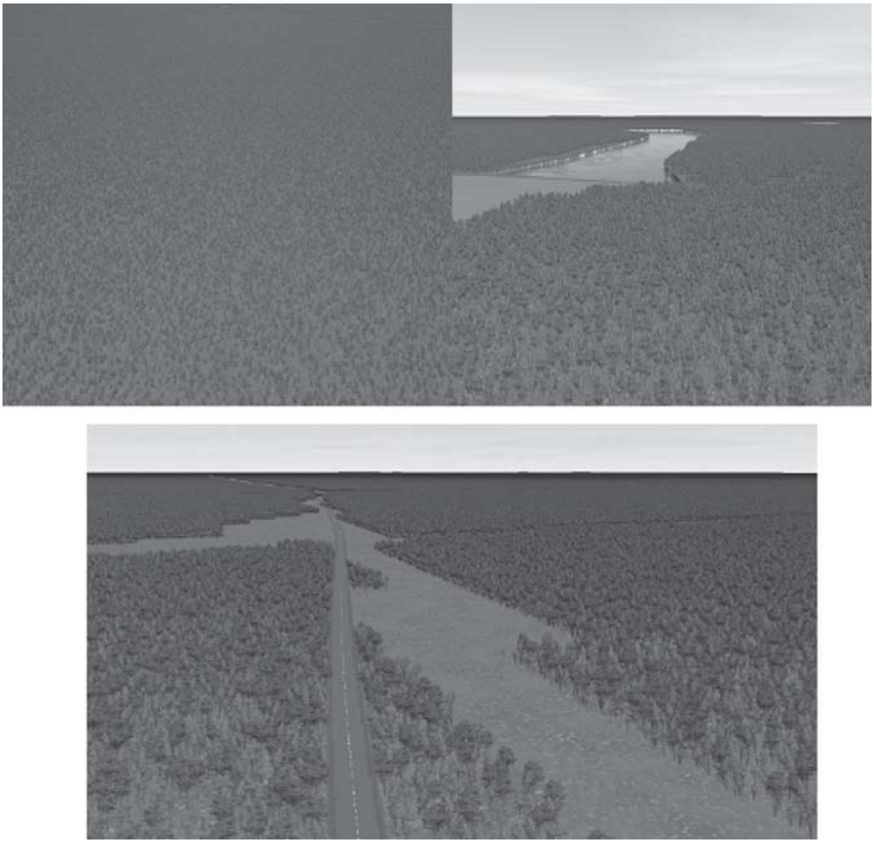

Fig. 7.4 Visualization of New Jersey pine barrens before fire showing surface features such as roads and lakes.

One of the first steps in this visualization was to recreate the forest environment in the pine barrens. Elevation and base heights for the landforms were obtained from a 10 meter digital elevation map (DEM) that was imported into VNS. A georeferenced aerial photograph was then used to digitize surface features such as roads, urban areas, and bodies of water. Surface features were digitized from the photograph into shapefiles using ArcMap. The shapefiles were imported into VNS in order to visualize the surface features. Road and water features were visualized using the included models in VNS, which looked similar to the features observed in the photograph. Urban areas were similarly visualized with a model comparable to asphalt, but with few or no buildings. This was done to reduce the total number of models in the visualization and to increase rendering speed.

To visualize the forest vegetation, user-made tree models were used instead of the models included with VNS. These tree models were made from photographs of local pine barrens tree species taken with a Nikon D70 digital camera during a visit in the summer of 2006. The photographs were loaded into Adobe Photoshop, where surrounding vegetation was removed to leave the tree of interest alone in the foreground, and the background was painted black. They were saved as JPG (JPEG, Joint Photographic Experts Group, compressed image) files and imported into the VNS graphics library to use as models.

To visualize the forest environment in a realistic manner, it was necessary to link the tree models with the actual forest structure in terms of characteristics such as tree height and density. This was accomplished using forest inventory data and a georeferenced Canopy Bulk Density (CBD) map. The forestry inventory data from test plots included tree height, species, diameter at breast height (DBH), basal area, and density. The data were analyzed to determine the averages and standard deviations for tree height and DBH in each plot. Test plot density was compared to values from the CBD map based on the equation for deriving Canopy Bulk Density in mixed conifer environments developed by Cruz et al. (2003):

$$\ln(\mathrm{CBD}) = (0.319 * \ln(\mathrm{basalarea})) + ((0.859 * \ln(\mathrm{treedensity})) - 8.445$$

When solved for density using the values from inventory data and the CBD map, the numbers obtained were highly correlated with measured density in the test plots. Therefore, the CBD map proved to be a suitable link between the visualization and physical inventory data. It was imported into ArcMap where each map symbology value was given a significantly different RGB (red, green, blue) color code and then exported as a GeoTIFF file (a Tagged Image File Format image retaining its spatial coordinates). When imported into VNS, this GeoTIFF acted as a color map to visualize various forest ecosystems of different structure based upon the CBD value.

Various forest “ecosystems” were constructed using the ecosystem function in VNS. An ecosystem is defined in VNS as an association of plant species all sharing common characteristics such as height, density, and relative frequency. Appropriate tree models were placed in the canopy and understory layers based on species from the inventory data. Average height was used for the main tree height while the standard deviation was used as an offset factor to vary tree heights. An included ground texture representing a forest floor with leaf litter was assigned to represent the ground. Due to the uniform nature of the site, most of these ecosystems were essentially the same in terms of species type, tree heights, and DBH classes. The most noticeable differences were in tree numbers and density, which was reflected in the CBD values used for color mapping.

Wildfire data were obtained from FARSITE simulations performed by Matthew Duveneck of Southern Maine Community College. The simulation was performed based on the conditions in the area on April 4, 2005 to recreate a fire that had occurred within the area previously. Temperature ranged from 4.8 to 18.9°C (40 to 66°F) with a humidity range of 16 to 61%. Wind speed ranged from 14.5 to 20.3 kilometers per hour (9 to 13 miles per hour) for the day. The wildfire was simulated over a series of 30 minute increments from 12:30 to 6:30 PM EST. Results from the simulation included a shapefile outlining the spread of the wildfire over each 30 minute increment and several raster files with data pertaining to flame length, fireline intensity, and crown activity (Fig. 7.5).

The custom tree models for tree visualization were recolored using Photoshop to represent burned trees. Models representing slightly burned trees remained mostly green, but had a small amount of brownish hue in the lower foliage and trunk. Additional yellow and red hues were added to the foliage to represent trees receiving more moderate damage. For more severe damage a larger amount of red hue was added to foliage along with a darker brown color or black marks to the trunk. Completely burned tree models had all foliage removed with very dark brown or black trunks and branches. A collection of flame models was also used to show fire occurrence (Fig. 7.6 and Fig. 7.7). A few models were drawn with Photoshop for the purpose of representing ground and understory fires. Another group of flame models came from a collection of Photoshop brushes created by Shimerlda (2007). These flame models were combined with the burned tree models to represent fire burning through the various layers of the canopy. Each tree model of different burn severity was merged with a corresponding flame model to represent fires occurring in the lower, middle, and upper canopies of the trees. To represent fire moving through the canopies of a group of trees, the larger flame models were merged with groups of three or four of the tree models.

Visualization of the wildfire was performed in a similar manner to ecosystem placement through the use of the CBD map. Examining the fireline intensity and flame length output in ArcMap showed a strong relationship between the two; areas of intense fire typically had flames of greater length. Therefore, the flame length output was chosen to serve as a color map to guide proper

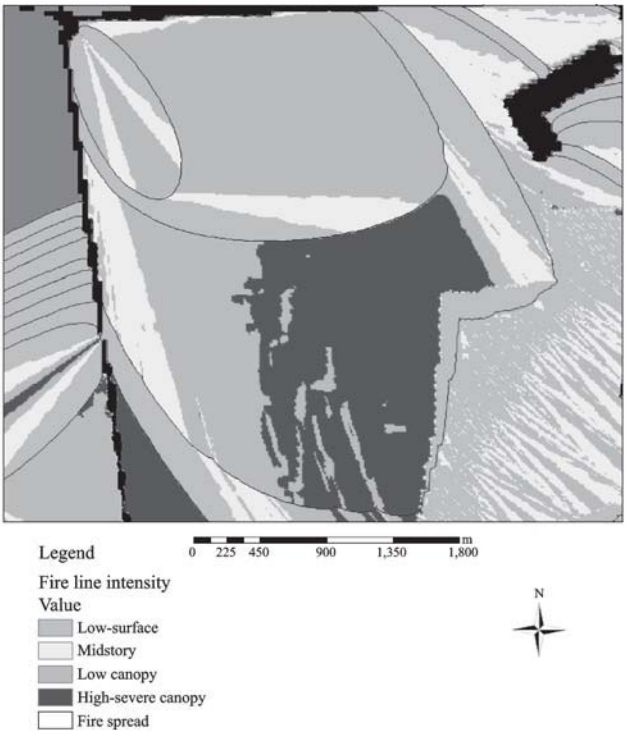

Fig. 7.5 An example of FARSITE output showing fire line intensity within the pine barrens study area.

fire visualization. Four different fire severity environments were constructed based on flame length: (i) flame lengths of 0 to 3 meters (0 to 9.84 feet) represented ground and understory fires; (ii) flame lengths of 3.01 to 6 meters (9.875 to 19.69 feet) were mid-canopy fires; (iii) flame lengths of 6.01 to 10 meters (19.72 to 32.81 feet) were overstory fires; and (iv) flame lengths of 10.01 to 16 meters (32.84 to 52.93 feet) represented fire extending over the top of the canopy. Three different burned environments were also constructed to show the effects of a fire passing through the forest: one for areas in which the flame front had just moved through it, one for areas with low intensity and flame length, and one for areas with high intensity fires.

These environments were constructed similarly to the regular forest environments in which the appropriate burned tree models were selected for each one. Ground textures were made using Visual Nature Studio's texture editor. The basic ground texture was edited to show increasing amounts of damage with increasing fire intensity. These different environments were then assigned to each of the different forest ecosystems as a material. Materials in VNS act much like an ecosystem with each possessing tree heights, density, ground models, etc. However, materials inherit all their characteristics

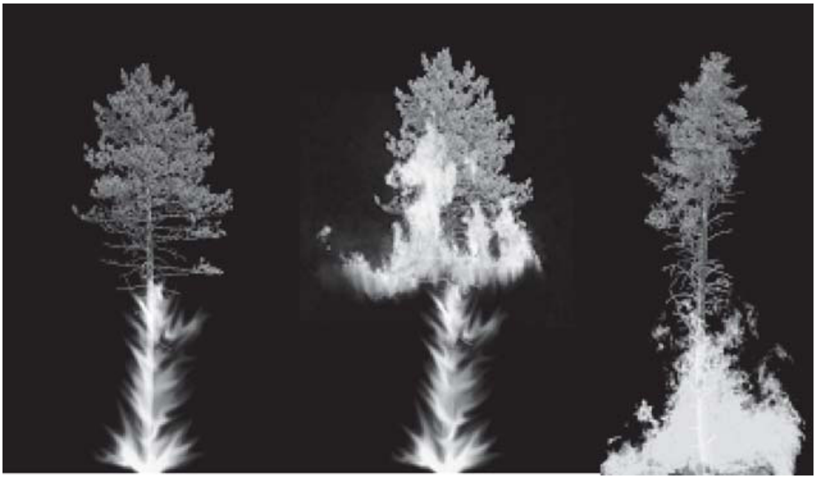

Fig. 7.6 Example of custom models used to illustrate fire damage. Individual tree models were edited to show varying degrees of damage based on severity. Different flame models were used to show movement of fire through the canopy.

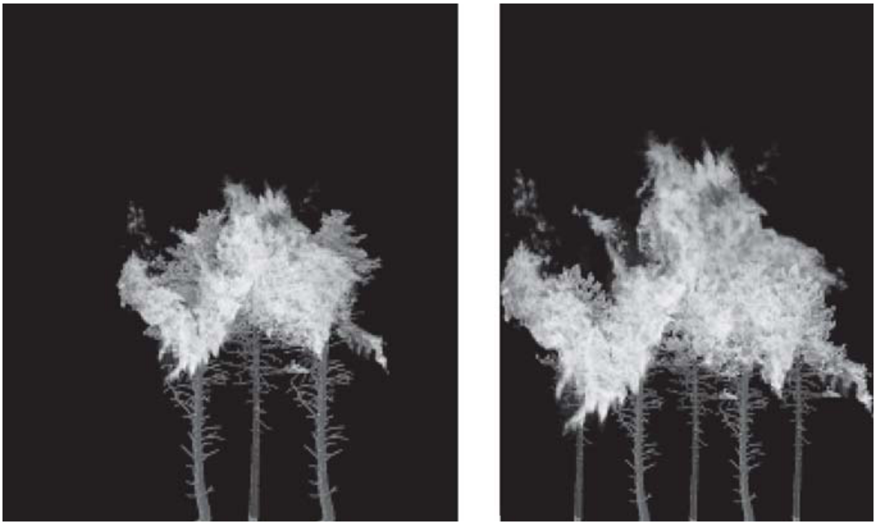

Fig. 7.7 Additional custom models for illustrating fire damage. Several tree models were used together with a larger flame in order to show movement of fire across the canopy.

from their parent ecosystems. As for the CBD map, the flame length raster and fire shapefiles were imported into ArcMap for editing to make a color map. Within each 30 minute increment of the fire as defined by the shape file, the map symbology of the flame length raster was changed to a varying grayscale RGB value for each type of fire ecosystem. As the fire progressed, it was necessary to change the type of ecosystem present in already burned areas to reflect the behavior and effects of the fire. Based on consultation with Matthew Duveneck (personal communication, 2007), it was estimated that a lowering of fire intensity would usually occur sometime during 30 minutes of the flame front moving, with a complete burn-out occurring after about an hour. Each of these edited images was saved as a GeoTiff and imported into VNS as a color map. The individual images were placed as a second color map overlying the original CBD color map. The use of both color maps was needed so that the proper forest ecosystem type was selected first and then the proper fire/burn materials within that ecosystem were visualized.

Animation was produced using the included animating editor. VNS' animation editor functions as a sequence moving through key frame images. Each key frame image is rendered in sequence as a still frame, with VNS adding a transition between frames. When combined, it produces a seamless animation through the entire sequence. By default, VNS saves animation in Windows .WMV format, but other formats such as QuickTime or .AVI can be made if the proper codec (a set of instructions for playing a specific computer media format) is present.

7.3 Results and discussion

As demonstrated by this study, Visual Nature Studio is capable of producing visualization for wildfires, with both still and animated images being produced (Fig. 7.8 and Fig. 7.9). Furthermore, it shows that such visualization is capable of being produced on a computer system running standard equipment. Despite being custom-built, the system used for this study possessed hardware easily purchased at most computer specialty stores. Likewise, the software applications used were standard for their given use: Photoshop for graphics and Arc Desktop for GIS analysis. Some specialized skills were needed for the graphics work to produce the custom models or for using VNS; however, no real programming knowledge was needed to construct the visualization. Unlike some past studies that needed custom programming to extend the capability of the visualization program, all the features needed for this study were included as part of VNS.

The still frame images and animation produced effectively showed the spread of the wildfire and its effects on the environment (Fig. 7.8 and Fig. 7.9). By combining the burned tree models with different flame models, the fire's movement through the canopy could be shown. The different models used illustrated the movement from the forest floor to the upper canopy of the forest and beyond. The different burn environments also showed the effects of the fire on the forest.

Tree models in each of the environments reflected fire severity and intensity, with high areas having almost total burned trees and low areas having

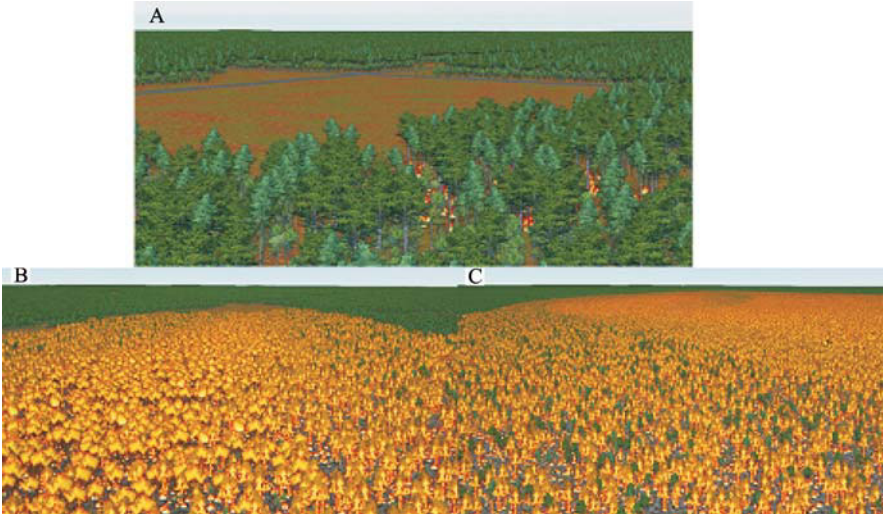

Fig. 7.8 Example of VNS visualization with fire in different canopies of the forest. A, surface fire; B, upper canopy; C, mid canopy. The position of the fire within the forest canopy relates back to the flame length output of FARSITE.

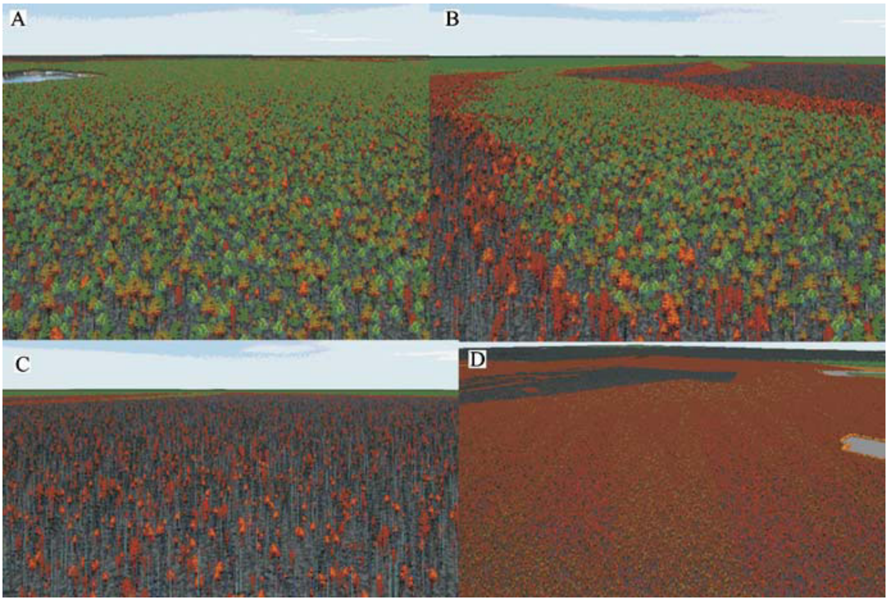

Fig. 7.9 Example of VNS visualization illustrating different levels of burn severity. A, surface fire; B, mid canopy; C, upper canopy; D, landscape level. The varying levels of severity can be linked back from the fire line intensity output of FARSITE.

trees with little or no damage. The edited ground textures served to show fire effects on ground and understory layers. With increasing severity, the textures grow darker with more and more understory material being burned out. Through the use of VNS' more advanced features, the scene realism is enhanced with effects such as clouds, lights and shadows, and reflections on water. As suggested by past visualization research, such realism allows observers to more readily identify with a scene. With such easy identification, observers are able to perceive it more easily and draw more meaningful conclusions.

There were several aspects that were detrimental to the realism of the visualized wildfire. One is the occurrence of a solid wall of flame for each fire type. Ordinarily, there should be an active fire front with varying levels of fire intensity behind it based on fuel types and amounts. Something similar was planned for the visualization but the mechanics of the shapefile prevented it from being implemented. The area for each 30 minute increment within the shapefile extended outward from the initial point of ignition. Using the outline of that area for a flame front would have yielded an unrealistic image of a flame front occurring all the way back to the original ignition point, which may have already burned out. Likewise, the raster images for flame length had to be reclassified into smaller groups more easily managed for visualization, so variation was limited. Therefore, areas of fire are best thought of as representations of the fire severity occurring over the area at the time, other than the fire actually occurring. Another problem for realism was the large extent of the study area. The large size made showing events over the whole area difficult to illustrate. Just to show the first three hours of the fire required the camera to be at height of nearly 150 meters. Such a height made showing areas further away difficult because many of the details were lost or they were almost indistinguishable. To show the entire wildfire in one scene would be extremely difficult and result in nearly all the details of faraway viewpoints being lost.

7.3.1 Data compatibility

Various data formats showed great compatibility with VNS. Shapefiles, raster images, and DEMs were easily imported and used with no conversion. Once imported, they could easily be used to visualize ground heights or other features based on the table attributes. Image files, such as JPEGs and bitmaps, could be imported easily to be used as custom models, but sometimes required some modification using Photoshop to paint the background black for a transparency mask. Georeferenced maps could be imported as color maps. However, some work was needed to change the RGB values in the map symbology due to VNS having difficulty to distinguish similar colors.

Data formats also showed a high degree of compatibility between VNS and ArcGIS. Features in shapefiles could be used in a similar manner in both programs. In fact, the shapefiles from ArcGIS could be easily imported into VNS and used with almost no modification. DEMs and raster files could also be used by both applications with no modifications.

7.3.2 Visual nature studio: Strengths and weaknesses

Visual Nature Studio has several advantages for producing realistic visualization of a scene. One is the ability to import user-made images as custom models. This feature is extremely useful for constructing more realistic tree models. Many of the other visualization applications such as VistaPro, Bryce 3D, and even ArcScene have tree models that are either very simplistic or composed of geometric shapes. While functional, these models may not look quite realistic or be recognizable by observers. By using photographs of the actual tree as models they look much more real and recognizable. In addition, there are options to vary the height and direction for the custom tree models. Custom models help recreate the variation among trees that is found in a natural forest. The custom model feature also allows the user to utilize models that are not normally part of VNS' graphics library, such as the flame models used in this study. It extends the function of VNS so that aspects not originally part of the application can be modeled and visualized.

Another advantage of VNS is its ability to use georeferenced images as color maps. Using each unique color as a guide, ecosystem matching can be done on the visualized landscape. Through this matching, forest ecosystems can be visualized more accurately as to how they appear in nature. They can be shown in the correct locations within the environment. Furthermore, two or more different color maps can be used for the same set of ecosystems. This allows for uses before and after visualization of a disturbance event or showing how a forest landscape changes over time. Another feature of the ecosystem mapping is the option to blend the edges of two different groups together. Raster images usually used for color maps have divisions with blocky shapes due to the information stored in each pixel. The blending of the edges allows ecosystems to appear more like they would in nature, merging in with the one next to it instead of instantly stopping after an imaginary line.

While the included tree models in VNS are rather poor, many of the other included models are of high quality and more realistic. Most of these models are part of the built-in tools used to simulate the general environment. These models include different types of clouds, water types, roads, ground textures, and sky models. The collection of high quality environmental models plays an important role in enhancing the realism of the scenes. Such high quality, realistic scenes have been found to be more effective for visualizations because the observer can identify with them better. This collection of models is an advantage for the user as well because it reduces the number of custom models needed. The user is able to quickly construct the base of the environments using the included models and then spend more time constructing custom models more crucial to the visualization. If every single model needed to be constructed by the user, creating visualization would be a long and cumbersome process.

While VNS has several advantages, there are also several limitations of the application. One disadvantage is that some computers need additional resources to optimally run VNS. Originally we had intended to do visualization on an older laptop system, but later had to use the AMD X2 instead. While many of the laptop's hardware components were near or above minimum requirements, more system resources would be needed for better usability. VNS should be run on a computer having at least 1 gigabyte of RAM, a 128 megabyte OpenGL video card (nVidia Geforce 5xxx family or above), and at least an 80 gigabyte hard drive. Many consumer computers sold currently can be purchased with this level of hardware or higher. However, users operating laptops or older computer systems may not have the necessary hardware to run VNS. Upgrading the system to high spec hardware in these cases may be expensive or not possible at all. These higher system requirements may be an important consideration for using VNS and may cost additional money to ensure that they are met.

Another feature of VNS that can be seen as a weakness is its rendering speed. Rendering in VNS occurs from the camera point out to edge of the horizon along the DEM of the landscape. In cases where a landscape covers a large area, the DEM is broken up into several smaller portions for rendering. VNS attempts to optimize rendering speeds by drawing only the portions of the scene that are immediately visible; however in many instances this is not what actually occurs. In many cases, VNS processes other partial DEMs and image objects that are not present in the immediate scene. The processing of these unnecessary items can slow rendering due to the application waiting to process the actual information needing to be rendered. Increased time for rendering can slow work if many previews of a visualization are needed while components are being edited. Increased time can complicate the production of animated images as well. With each additional individual key frame scene added to the animation, even more information is accumulated for processing. Time necessary for rendering these sequences can increase dramatically with the addition of more key frames. Even short sequences running up to a minute at 30 frames per second can take up to 72 hours to render.

VNS is a very powerful application capable of performing many different functions. However, many of these functions are quite complex, requiring many steps to carry out. There is a tutorial that explains the basic functions of the application, but many of the more advanced functions and settings are not explained. In addition, the explanations in the manual are not very detailed. This lack of sufficient explanation can be a potential problem for new users not familiar with VNS. Users may require additional time to learn the features and experiment with settings to determine their function. Extra time taken to obtain this familiarity can be added to the time needed to complete a visualization. An improvement in both the tutorials and documentation included with VNS could help to provide users with more information and better understanding of the application.

7.4 Conclusion

As demonstrated in this study, wildfire visualization is possible using Visual Nature Studio. Functions in the application allowed for high quality tree models and flame models to be used to create a realistic scene. Editing the tree models and ground textures allowed the effects of the fire to be illustrated both in the tree canopy and on the ground. The result was a realistic visualization of the New Jersey pine barrens showing the spread of fire and its effects within the environment. Still frame images were easily created during any point of the fire along with animated views by piecing together a series of key frame images. The large area in which the event took place presented challenges for visualizing the total extent of the wildfire. High aerial views often had to be used which reduced the graphical quality of the visualization to some extent.

This study also helped establish a protocol for visualizing fire using VNS. Such a protocol provides users with appropriate visualization methods without the need for all the original methodological research and investigation. Lab technicians with adequate computer operating skills can use the protocol for creating visualization of fires not only in New Jersey, but in any other area. These technicians would not necessarily need to understand the fundamental theories and mechanics of visualization as long as they possessed the necessary skills to use the various graphic and visualization applications. Having these methods established beforehand could potentially allow visualization to be constructed more quickly, saving both time and money.

Identifying the advantages and disadvantages of different applications can help users to choose the proper application for use based on available resources such as computer hardware, money, and operating ability. In our case study, Visual Nature Studio was found to be very good at creating realistic, high quality scenes. There is ample support for natural environments such as forests, mountains, and deserts. However, there is also support for more urban environments such as towns. Using the right application for a project can help to construct a visualization in a timely manner. Less time is spent on exploring features that are unnecessary, saving more time for aspects crucial to creating a meaningful visualization.

7.5 Future wildfire visualization research

3D visualization could be very useful for building and evaluating forest scenarios that do not currently exist, such as historic forests, alternative management scenarios, and possible future disturbances, allowing the public and scientists to better comprehend the dynamics of actual and potential forest and landscape patterns. 3D visualization provides managers and scientists with an opportunity to foresee forest landscape changes before a management plan is executed.

Fire visualization efforts for the immediate future should focus on improving the realism of rendered scenes based on the constraints of the application. For VNS, this could include features such as animated smoke and flames, ground textures from actual fire events, and tree models made from burnt trees in the field. VNS supports the NTSC video format so it may be possible to include video footage of a wildfire as part of the visualization. Likewise, observer rating studies would be valuable for determining how closely observers are able to identify with a visualization. While rating studies have been done for other types of visualization, there have been no studies to determine how observers relate to those done using VNS or other 3D visualization software. Working within the constraints of the current applications allows for effective visualization methods to be developed without modifying the application. As the applications are improved, these methods can then be applied using the newer technology.

With the recent introduction of multi core CPUs, faster RAM, and more powerful video cards, computing systems have grown even more powerful. With most software, visualization applications can be improved by taking advantage of this increased power. With more computing potential and faster graphic power, a goal of visualization programs should be the real time rendering of landscapes with animated fire and smoke. The program would be able to use high quality graphics to show real time movement of the fire through the environment.

While there is no software application that does everything described, an application that shows great promise is VRFire (Sherman et al., 2007). VRFire is a visualization application being developed at the University of Nevada that uses the FARSITE model to control fire visualization. Analysis is performed to determine the location of the flame front. When combined with landscape and vegetation data, the application is able to determine vegetation within the flame front zones and render appropriate tree models, flames, and smoke if necessary. When areas of flame front are combined for the intervals over the entire fire, the visualization is able to display burned areas in real time. VRFire has the advantage of running on an open source platform of Suse Linux and using open source graphical software. The application runs on a current Opteron CPU-based system with an nVidia Geforce 6800 GT graphics card. Output can be displayed to a wide variety of display outputs, from standard computer monitors to a display screen that interacts with a virtual reality head and wand tracking unit. Currently, VRFire suffers from problems such as limited graphical details and a simplified view of the flame front. However, these problems are being improved so that they accurately represent features that exist in nature.

Improving applications such as VRFire allows the software to take advantage of more powerful computer hardware. With increased technological capability, more features such as real time rendering and higher quality images can be made part of the application. As visualization incorporates more realistic and higher quality images in real time, they can begin to more effectively represent events that occur in nature. With a higher accuracy of representation, observers are more able to identify with the scene and draw conclusions from it, making visualized scenes more useful and important tools for exploring data or natural processes.

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Chapter 8 Predicting Tree Growth Dynamics of Boreal Forest in Response to Climate Change

Zewei Miao* and Chao Li

Abstract

Understanding how tree grow rate could change under various climatic conditions is vital for simulating forest dynamics and climate change impact and mitigation-related research. In this study, we evaluated 30 dendroclimatic equations with seven climate change scenarios to approximately predict growth responses of the boreal forest to climate change in west-central Canada. We used an index, relative climate response rate (RCR) of tree ring growth, to amplify the climate signal involved in tree growth rate and consistently compare tree-ring growth rate estimates of each equation. Our results suggested that for west-central Canada, the response of tree growth rate to climate was positively correlated with both precipitation and temperature. Using a 95% confidence interval for the mean of the estimates and validation with tree-ring network data, a dendroclimatic equation associated with current summer temperature and current May and previous summer precipitation can be identified as having average predictions of tree growth rate for the boreal forest that is appropriate for landscape scale.

Keywords

Dendroclimatic equations, relative climate response rate of tree growth, tree-ring network data, west-central Canada.

8.1 Introduction

The boreal forest is expected to be at the front range of climate change.

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Understanding how tree grow rate could change under various climatic conditions is vital for simulating forest dynamics in climate change impact and mitigation-related research. In boreal forests, impacts of climate changes on forest dynamics may result from the interaction between forest growth and disturbance regimes (e.g., insect, disease, fire occurrences, etc.) caused by climate change (Price and Apps 1996; Girardin et al. 2008; Yin et al. 2008). In recent years, increased attention has been paid to how climate change could affect disturbance regimes and the development of disturbance models that use projected climate change scenarios, primarily based on the GCM (Global Coupled Climate Model) and RCM (Regional Climate Models) predictions (Price and Apps 1996). Less emphasis, however, has been put on addressing how forest growth could be influenced by climate change and how to predict forest growth in response to climate change. This is mainly because of the lack of empirical observations suitable for traditional methods of generating forest growth and yield models or equations under changed climate variables. Consequently, many existing forest growth models or equations cannot reflect the forest response to climate change. This has become a challenge and barrier in climate change impact research.

Predicting boreal forest dynamics responding to climatic change needs efficient methods to facilitate the assessment of tree growth rate in response to climate change, particularly for mixed-wood boreal forest mosaics in west-central Canada. Dendroclimatic linear or nonlinear equations provide an appropriate approach for analyzing the past climate signal through tree growth to climatic responses (Fritts 1976; Devall et al. 1995, 1998; Szeicz and MacDonald 1995; St. George and Luckman 2001; Briffa et al. 2002; Moore et al. 2002; Watson and Luckman 2002, 2004), and for addressing the impacts of environmental variables on forest growth (Jozsa et al. 1984; Nöjd and Reams 1996; Tessier et al. 1997). However, fewer studies have stressed how to properly adopt existing chronology equations to predict tree growth rate under climate change conditions (Pan and Raynal 1995a, b; Laroque and Smith 2003; Girardin et al. 2008).

Variability of dendroclimatic equations makes their application to predict response of tree growth to climate change difficult at large spatial and temporal scales. Dendroclimatic equations usually have been developed by using chronology data from isolated study sites or with limited sampling, so the equations are often published inconsistently in equation form and regression values for a given species (or site) by different authors (Stockton and Fritts 1971; Fritts 1974, 1976, 1991; Jenkins et al. 2003). Most of these equations are site-, species-, and objective-dependent, and they may not be the best estimates for the chronology data sets collected from other regions. This presents a challenge on whether such dendroclimatic equations can be reasonably used in predicting tree growth at large spatial areas that include the areas outside the native region where the equation was originally developed. There are two approaches to meet this challenge. One is to collect chronology network data across the entire large spatial unit and re-estimate the parameter values, thus resulting in a new equation based on the new data set. This approach is straightforward in implementation, but it is expensive in terms of time and resources. The second possible approach is to evaluate the relative accuracy and bias of using an existing equation of tree-radial growth increments in predicting tree growth under various climatic conditions. This evaluation requires a standard test climatic data set that contains a wider range of climatic conditions. This approach may not be able to provide the best estimates for a specific empirical equation, but it can identify an existing dendroclimatic equation that could provide the lowest biased tree growth estimate under various climatic conditions. This approach is relatively inexpensive and time-saving to implement compared with the first approach.

The objective of this study is to produce a first approximation of growth response of boreal forest to climate change by using existing knowledge from the research field of dendrochronology. Through an extensive literature search, we selected 30 linear equations whose sampling sites were within 10 degrees north (or south) of mean latitude of our study sites. We constructed seven testing climate change scenarios for the three typical boreal forest sites on the basis of the thirty-year climate "Normals" and extant climate change projections of MAGICC(Model for the Assessment of Greenhouse-gas Induced Climate Change)/SCENGEN(A Regional climate SCENario GENerator) and CGCM(Coupled Global Climate Model) models for west-central Canada. We then calculated RCR (the relative climate response rate) surfaces using the test climatic change scenarios and validated the selected equations using tree-ring network data of west-central Canada. This paper will end with a discussion of the identification of the equation with the lowest risk through the 95% confidence interval of the estimates and validation with tree-ring network data. Our results can provide a way of addressing the challenge of applying the existing dendroclimatic equations to predict responses of tree growth to climate change at large spatial and temporal scales.

8.2 Materials and methods

This section mainly included selection of dendroclimatic equations, construction of test climate change scenarios, dimensionless analysis and identification of dendroclimatic equations for west-central Canada's boreal forest.

8.2.1 Site description

We chose three study sites which represented typical boreal forest areas in west-central Canada (Alberta, Saskatchewan, and Manitoba). The study sites consisted of Fort McMurray A Alberta (56°39′N 111°13′W, 369 m asl [above sea level]), La Ronge A Saskatchewan (55°09′N 105°16′W, 375 m asl), and Thompson A Manitoba stations (55°48′N 97°52′W, 215 m asl) (Fig. 8.1). 30-yr annual precipitation and mean temperature ("Normals") of the three sites were: 464.7 mm and 0.2°C for Fort McMurray A (1961 to 1990), 489.4 mm and −0.5°C for La Ronge A (1959 to 1990), 535.6 mm and −3.4°C for Thompson A (1967 to 1990), 496.6 mm and –1.2°C for three-site average, respectively (Environment Canada 1993). In these sites, boreal forest is dominated by softwood and mixedwoods species, e.g., white spruce (*Picea glauca* (Moench) Voss), black spruce (*Picea mariana* (Mill.) B.S.P.), trembling aspen (*Populus tremuloides* Michx.), jack pine (*Pinus banksiana* Lamb.), balsam fir (*Abies balsamea* (L.) Mill.), lodgepole pine (*Pinus contorta* Dougl. var. *latifolia* Engelm.), Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), balsam poplar (*Populus balsamifera* L.) and white birch (*Betula papyrifera* Marsh.) (Jozsa et al. 1984; Dang and Lieffers 1989; Hogg and Schwarz 1999).

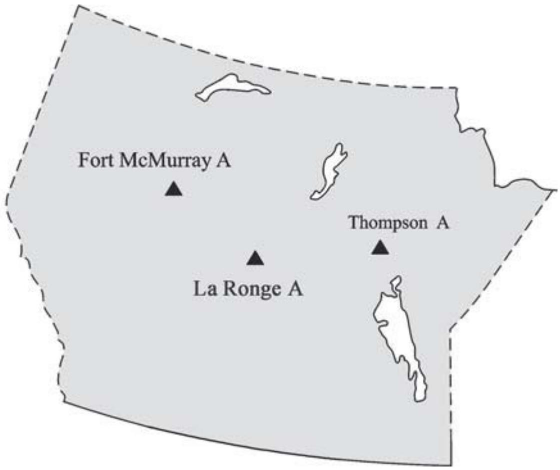


Fig. 8.1 Map of the study area and location of study sites.

8.2.2 Test climate change scenarios

In past decades, many climate models have projected that the greatest increases in temperatures might occur in the high latitudes, in winter, and over land, and that the warming could be greater in western Canada, with the most significant warming occurring during the winter and spring (McFarlane et al. 1992; Laprise et al. 1998). Hulme and Sheard (1999) reported that climate change depends largely on how sensitive the Earth's climate is to rising CO_2 concentrations. They chose three different values for this so-called climate sensitivity—low (1.5°C), medium (2.5°C) and high (4.5°C) and four different future greenhouse gas emissions scenarios defined in the preliminary Special Report on Emissions Scenarios (SRES) of the IPCC (Intergovernment Panel on Climate Change) to simulation climate change scenarios of Canada by using MAGICC/SCENGEN models.

By combining the climate change projections of CGCM and MAGICC/ SCENGEN models for west-central Canada (Canadian climate impacts scenarios 2003; Hulme and Sheard 1999), we developed seven scenarios for the three sites by considering possible increase, decrease, and/or no change in temperature and precipitation (Table 8.1). Scenarios I to V involved changes in temperature and precipitation in winter (November, December, and January) and summer (June, July, and August). Scenarios VI and VII involved either precipitation in winter or temperature for each month, respectively. Each scenario included six levels. Of them, level one was 30-yr mean climate data of three sites (1961-1990 for Fort McMurray A, 1959 to 1990 for La Ronge A, and 1967 to 1990 for Thompson A). Precipitation in winter or summer was set to change at rates of 10.25%, 15.50%, 15.50% , and 13.75% of level 1 from 1961 to 2080 for Fort McMurray A, La Ronge A, Thompson A, and the three-site average, respectively. Precipitation in spring and autumn was set to be unchanged. In scenario I, between two neighbor levels, winter and summer temperatures were changed by 1.32°C and 0.84°C for Fort McMurray A, 1.47°C and 0.86°C for La Ronge A, 1.66°C and 0.82°C for Thompson A, and 1.48°C and 0.84°C for the three-site average. For other scenarios, the temperature change rates of 1.08°C, 1.16°C, 1.24°C, and 1.16°C per level were designated to every month for Fort McMurray A, La Ronge A, Thompson A and the three-site average. It is worth noting that the scenarios and rate of precipitation and temperature changes conformed to climate change projections from 1960 to 2080 by Hulme and Sheard (1999) and Canadian climate impacts scenarios (2003). Hulme and Sheard (1999) and the Canadian climate impacts scenarios (2003) projected that climate change in the boreal forest area of Fort McMurray, La Ronge and Thompson could result in an increase of winter precipitation by about 10.25%, 15.5%, and 15.5% until 2080, no change of summer precipitation, an increase of winter temperature by about 4.40°C, 4.90°C, and 5.53°C, and an increase of summer temperature by about 2.8°C, 2.85°C, and 2.73°C above the 30-yr mean level (i.e., level one) for Fort McMurray A, La Ronge A and Thompson A, respectively.

In this study, evapotranspiration of the boreal forest of the Prairie provinces were calculated as follows:

$$PET = \begin{cases} 93 * D * \exp(A/9300),\ T_{\text{mean}} \geqslant 10°\text{C} \\ (6.2T + 31) * D * \exp(A/9300),\ -5°C \leqslant T_{\text{mean}} < 10°\text{C} \\ 0,\ T_{\text{mean}} < -5°\text{C} \end{cases} \tag{8.1}$$

where PET is potential evaportranspiration in mmmonth^{-1}, T_{mean} is monthly

Table 8.1 Climate change scenarios used in this study.

Scenario description[a]	Scenario level					
	Level 1[b]	Level 2	Level 3	Level 4	Level 5	Level 6
Scenario I						
Precipitation increase (mm): 14.00, 22.76, 24.91, 20.55 in summer	464.7, 489.4, 535.6, 496.6	478.7, 512.2, 560.5, 517.1	492.7, 534.9, 585.4, 537.7	506.7, 557.7, 610.3, 558.2	520.7, 580.4, 635.2, 578.8	534.7, 603.2, 660.1, 599.3
Mean temperature increase (°C): 1.32, 1.47, 1.66, 1.48 in winter, 0.84, 0.86, 0.82, 0.84 in summer, and 1.08, 1.16, 1.24, 1.16 in other seasons	0.2, -0.5, -3.4, -1.2	1.3, 0.7, -2.2, -0.1	2.4, 1.8, -0.9, 1.1	3.4, 3.0, 0.3, 2.2	4.5, 4.2, 1.6, 3.4	5.6, 5.3, 2.8, 4.6
Scenario II						
Precipitation increase (mm): 14.00, 22.76, 24.91, 20.55 in winter	464.7, 489.4, 535.6, 496.6	478.7, 512.2, 560.5, 517.1	492.7, 534.9, 585.4, 537.7	506.7, 557.7, 610.3, 558.2	520.7, 580.4, 635.2, 578.8	534.7, 603.2, 660.1, 599.3
Mean temperature increase (°C): 1.08, 1.16, 1.24, 1.16 in every month	0.2, -0.5, -3.4, -1.2	1.3, 0.7, -2.2, -0.1	2.4, 1.8, -0.9, 1.1	3.4, 3.0, 0.3, 2.2	4.5, 4.2, 1.6, 3.4	5.6, 5.3, 2.8, 4.6
Scenario III						
Precipitation decrease (mm): 14.00, 22.76, 24.91, 20.55 in summer	464.7, 489.4, 535.6, 496.6	450.7, 466.6, 510.7, 476.0	436.7, 443.9, 485.8, 455.5	422.7, 421.1, 460.9, 434.9	408.7, 398.4, 436.0, 414.4	394.7, 375.6, 411.1, 393.8
Mean temperature increase (°C): 1.08, 1.16, 1.24, 1.16 in every month	0.2, -0.5, -3.4, -1.2	1.3, 0.7, -2.2, -0.1	2.4, 1.8, -0.9, 1.1	3.4, 3.0, 0.3, 2.2	4.5, 4.2, 1.6, 3.4	5.6, 5.3, 2.8, 4.6
Scenario IV						
Precipitation increase (mm): 14.00, 22.76, 24.91, 20.55 in summer	464.7, 489.4, 535.6, 496.6	478.7, 512.2, 560.5, 517.1	492.7, 534.9, 585.4, 537.7	506.7, 557.7, 610.3, 558.2	520.7, 580.4, 635.2, 578.8	534.7, 603.2, 660.1, 599.3
Mean temperature decrease (°C): 1.08, 1.16, 1.24, 1.16 for every month	0.2, -0.5, -3.4, -1.2	-0.9, -1.7, -4.6, -2.4	-2.0, -2.8, -5.9, -3.6	-3.0, -4.0, -7.1, -4.7	-4.1, -5.2, -8.4, -5.9	-5.2, -6.3, -9.6, -7.0
Scenario V						
Precipitation decrease (mm): 14.00, 22.76, 24.91, 20.55 in summer	464.7, 489.4, 535.6, 496.6	450.7, 466.6, 510.7, 476.0	436.7, 443.9, 485.8, 455.5	422.7, 421.1, 460.9, 434.9	408.7, 398.4, 436.0, 414.4	394.7, 375.6, 411.1, 393.8
Mean temperature decrease (°C): 1.08, 1.16, 1.24, 1.16 for every month	0.2, -0.5, -3.4, -1.2	-0.9, -1.7, -4.6, -2.4	-2.0, -2.8, -5.9, -3.6	-3.0, -4.0, -7.1, -4.7	-4.1, -5.2, -8.4, -5.9	-5.2, -6.3, -9.6, -7.0

Continued

Scenario description[a]	Scenario level					
	Level 1[b]	Level 2	Level 3	Level 4	Level 5	Level 6
Scenario VI						
Precipitation increase (mm): 14.00, 22.76, 24.91, 20.55 in winter	464.7, 489.4, 535.6, 496.6	478.7, 512.2, 560.5, 517.1	492.7, 534.9, 585.4, 537.7	506.7, 557.7, 610.3, 558.2	520.7, 580.4, 635.2, 578.8	534.7, 603.2, 660.1, 599.3
Scenario VII						
Mean temperature increase (°C): 1.08, 1.16, 1.24, 1.16 in every month	0.2, -0.5, -3.4, -1.2	1.3, 0.7, -2.2, -0.1	2.4, 1.8, -0.9, 1.1	3.4, 3.0, 0.3, 2.2	4.5, 4.2, 1.6, 3.4	5.6, 5.3, 2.8, 4.6

[a] Precipitation (mm) in summer and winter was set to increase or decrease at a rate of 10.25%, 15.50%, 15.50%, and 13.75% of normal annual precipitation from 1961 to 2080 (i.e., by 14.00, 22.76, 24.91, and 20.55 mm for each level) for Fort McMurray A, La Ronge A, Thompson A, and the three site average, respectively. Precipitation in spring and autumn was fixed at the normal level. Monthly mean temperatures (°C) were set to change at the rate of 1.08°C, 1.16°C, 1.24°C, and 1.16°C for Fort McMurray A, La Ronge A, Thompson A, and the three site average, respectively. The climate change scenarios and change rates were designed on a basis of previous research (McFarlane et al. 1992; Laprise et al. 1998; Hulme and Sheard 1999; Canadian climate impacts scenarios 2003; Moore et al. 2002; Muzik 2002; Sauchyn 2002).

[b] 30-yr mean climate data (1961-1990) for Fort McMurray A, La Ronge A, and Thompson A meteorological stations. In the table, the four values per cell correspond to Fort McMurray A, La Ronge A, Thompson A, and the average of the three sites, respectively.

mean temperature (°C) in different scenarios, A is station altitude (m), and D is vapour pressure deficit (kPa) (Hogg and Schwarz 1999). Growing days (or degree-days) were fixed at level one, because of their greater uncertainty.

8.2.3 Selection of dendroclimatic equations

Criteria used to collect chronology equations of tree-radial growth increments included: (i) Site similarity. The sampling sites where the selected equations were developed should be similar to the study areas in terms of latitude, temperature and precipitation, i.e., the sampling sites should be within ± 10 degrees of the mean latitude of the three sites in the study. (ii) Equation simplicity. The selected equations should be linear or simply nonlinear (e.g., a quadratic equation), and climatic variables should be the major independent variables. For the aggregate chronology equations, we took into account only climatic signal residuals by excluding the influence of non-climatic factors on tree growth. (iii) Equation availability. Those selected should be published in peer-reviewed journals or serials (e.g., information reports), and the regression coefficients should be available in the literature. In this study, an emphasis was placed on factors of site rather than species, because site factors appeared most responsible for variations in tree growth response to climate change (Fritts 1974, 1976, 1991; Cook et al. 1987; Rolland 2002), and site differences were usually more significant than species differences for tree chronology across the northern United States and Canada (Bednarz 1982; Gray and Pilcher 1983; Fritts 1990; Watson and Luckman 2002, 2004). Meanwhile, we focused on identifying an equation suitable for predicting relative and average tree radial growth of mixed-wood boreal forest at a large spatial scale, since some equations for the mixed chronology usually contained two or more species and the regression equations for multiple species chronology were often used to infer past climate records through response of tree growth of mixedwood forest mosaics at a large spatial scale (Miina 2000; Yeh and Wensel 2000; Cook et al. 2003; Touchan et al. 2003).

Through a wide literature review, we selected 30 dendroclimatic equations (Table 8.2). The average latitude (57°14′N), elevation (584.31 m a.s.l.), annual precipitation (510.5 mm), and yearly mean temperature (1.86°C) of the sampling sites of the 30 equations were close to the mean of three study sites (55°65′ N, 319.67 m asl., 496.6mm and 1.2°C) (Table 8.3).

Each equation was calculated with the seven scenarios. Total calculations for the 30 equations were as high as 23,040 estimates for all climate scenarios and sites in the study.

Table 8.2 Dendroclimatic equations selected for this study.

No.	Dendroclimatic equations[a]
1	$\mathrm{SP}_{\mathrm{Aug}(t-1)\text{ to July}(t)} = 220.83 * I_t + 140.22$[b]
2	$G_t = 2.33 - 0.0083Year - 0.0287FTC + 0.0191CMI_t + 0.0350CMI_{t-1} + 0.565G_{t-1}$
3	$G_t = -0.40 + 0.1453Year + 0.168CMI_{\mathrm{Aug}(t-1)\text{ to July}(t)} + 0.0152GD_{\mathrm{Mar}(t)\text{ to July}(t)}$
4	$\mathrm{Log}\left(\frac{G_t}{G_{t-1}}\right) = -1.244 - 0.0172FTC + 0.0078FTC_{t-1} + 0.0049FTC_{t-2} + 0.0117CMI_{\mathrm{Sep}(t-1)\text{ to Aug}(t)} + 0.00113GD_{\mathrm{Apr}(t)\text{ to Aug}(t)} + 0.0293SD_{\mathrm{Mar}(t)}$
5	$G_t = -0.652 + 0.0280P_{\mathrm{Aug}(t-1)\text{ to July}(t)} + 0.0230P_{\mathrm{Aug}(t-2)\text{ to July}(t-1)} + 0.0113P_{\mathrm{Aug}(t-3)\text{ to July}(t-2)}$
6	$G_t = -0.335 + 0.0326P_{\mathrm{Aug}(t-1)\text{ to July}(t)} + 0.0178P_{\mathrm{Aug}(t-2)\text{ to July}(t-1)}$
7	$G_t = -0.443 + 0.0246P_{\mathrm{Aug}(t-1)\text{ to July}(t)} + 0.0164P_{\mathrm{Aug}(t-2)\text{ to July}(t-1)} + 0.0133P_{\mathrm{Aug}(t-3)\text{ to July}(t-2)}$
8	$I_t = 0.67 + 0.0015P_{\mathrm{June}(t-1)\text{ to Aug}(t-1)}$
9	$I_t = 2.65 - 0.081T\max_{\mathrm{June}(t-1)\text{ to Aug}(t-1)}$
10	$I_t = -2.77 + 0.86T\min_{\mathrm{June}(t-1)\text{ to Aug}(t-1)} - 0.049T\min^2_{\mathrm{June}(t-1)\text{ to Aug}(t-1)}$
11	$\mathrm{RW}_t = -0.70 + 82.18T_{\mathrm{July}(t)} - 0.363T^2_{\mathrm{July}(t)}$
12	$\hat{y}_t = -0.01967278ES_{\mathrm{June}(t)} + 0.00315112P_{\mathrm{July}(t)} + 0.00260555P_{\mathrm{May}(t)} - 0.00110200T_{\mathrm{May}(t-1)} + 0.01199449ES_{\mathrm{Apr}(t-1)}$
13	$\hat{y}_t = 0.00151306P_{\mathrm{July}(t)} - 0.00902201ES_{\mathrm{July}(t)} + 0.00879209ES_{\mathrm{Apr}(t-1)} - 0.00091872T_{\mathrm{July}(t-1)} + 0.00123843P_{\mathrm{Aug}(t-1)}$
14	$\hat{y}_t = 0.00315403P_{\mathrm{July}(t)} + 0.00527618P_{\mathrm{Apr}(t)} + 0.00292561P_{\mathrm{May}(t)} + 0.00252929P_{\mathrm{Aug}(t-1)}$
15	$Y_t = 0.361 - 0.003T_{\mathrm{May}(t)} + 0.024T_{\mathrm{June}(t)} + 0.001P_{\mathrm{July}(t)} + 0.004P_{\mathrm{Aug}(t)}$
16	$I_t = 6.1947T_{\mathrm{July}(t)} + 0.0059P_{\mathrm{Sep}(t-1)} + 0.1422P_{\mathrm{June}(t)} + 0.0672P_{\mathrm{Oct}(t-1)} - 4.3694T_{\mathrm{Sep}(t-1)} + 2.9004T_{\mathrm{May}(t-1)} + 0.0532P_{\mathrm{Dec}(t)} + 0.0453P_{\mathrm{Jan}(t)} + 3.2289$
17	$I_t = 0.68I_{t-1} + C_t, \quad C_t = 50.72 + 1.19ST_{\mathrm{July}(t)\text{ to Aug}(t)} + 0.11P_{\mathrm{May}(t)} + 0.073SP_{\mathrm{June}(t-1)\text{ to July}(t-1)}$
18	$I_t = 0.64I_{t-1} + C_t, \quad C_t = 87.29 + 0.13P_{\mathrm{May}(t)} + 0.059SP_{\mathrm{June}(t-1)\text{ to July}(t-1)}$
19	$I_t = 0.44I_{t-1} + C_t, \quad C_t = -10.89 + 2.85ST_{\mathrm{July}(t)\text{ to Aug}(t)} + 0.13SP_{\mathrm{June}(t-1)\text{ to Aug}(t-1)}$
20	$C_t = -0.13 + 0.034T_{\mathrm{May}(t)} + 0.025ST_{\mathrm{July}(t)\text{ to Aug}(t)} + 0.00062SP_{\mathrm{June}(t-1)\text{ to Aug}(t-1)}$
21	$I_t = 0.65I_{t-1} + C_t, \quad C_t = 81.00 + 2.23T_{\mathrm{July}(t)} - 1.72T_{\mathrm{Aug}(t-1)} + 0.061SP_{\mathrm{June}(t-1)\text{ to July}(t-1)}$
22	$I_t = 0.60I_{t-1} + C_t, \quad C_t = 94.33 - 0.49T_{\mathrm{Feb}(t)} + 0.83ST_{\mathrm{May}(t)\text{ to July}(t)} - 1.01ST_{\mathrm{July}(t-1)\text{ to Aug}(t-1)}$
23	$I_t = 0.59I_{t-1} + C_t, \quad C_t = 121.80 - 0.51T_{\mathrm{Feb}(t)} + 1.48T_{\mathrm{June}(t)} - 1.55ST_{\mathrm{July}(t-1)\text{ to Aug}(t-1)}$
24	$I_t = 0.43I_{t-1} + C_t, \quad C_t = 100.90 + 1.67T_{\mathrm{May}(t)} - 0.92T_{\mathrm{Aug}(t-1)}$
25	$I_t = -0.24I_{t-1} + C_t, \quad C_t = 1.31 + 0.0064ST_{\mathrm{Feb}(t)\text{ to May}(t)} - 0.018T_{\mathrm{June}(t)}$
26	$I_t = 0.56I_{t-1} + C_t, \quad C_t = 92.21 - 0.45T_{\mathrm{Feb}(t)} + 0.96ST_{\mathrm{May}(t)\text{ to Aug}(t)} + 0.066P_{\mathrm{July}(t)} - 1.67ST_{\mathrm{July}(t-1)\text{ to Aug}(t-1)}$
27	$I_t = 0.0389 + 0.0272T_{\mathrm{July}(t)} - 0.0308T_{\mathrm{July}(t-1)}$
28	$RW_t = -15.44 + 2.984AT_{\mathrm{June}(t)\text{ to July}(t)}$

Continued

No.	Dendroclimatic equations[a]
29	$I_t = 0.8583 + 0.1830T_{\mathrm{May}(t)\ \mathrm{to\ July}(t)} - 0.3523P_{\mathrm{Sep}(t-1)\ \mathrm{to\ Oct}(t-1)} +$ $0.1885T^2_{\mathrm{May}(t)\ \mathrm{to\ July}(t)} - 0.3669P^2_{\mathrm{Sep}(t-1)\ \mathrm{to\ Oct}(t-1)} +$ $0.0086T_{\mathrm{May}(t)\ \mathrm{to\ July}(t)}P_{\mathrm{Sep}(t-1)\ \mathrm{to\ Oct}(t-1)}$
30	$I_t = 0.409 + 0.00935T_{\mathrm{June}(t)} + 0.00124T_{\mathrm{July}(t)}$ for general climate scenario or $I_t = 0.409 + 0.00935(T_{\mathrm{June}(t)} + 5) + 0.00124(T_{\mathrm{July}(t)} + 5)$ for "sharp" climate change scenario

[a]Independent variables: I = ring width index (dimensionless), t = the current year (i.e., year t), t-1 = the previous year (i.e., year t-1), $Year$ = number of years since 1950 (assumed to be 25 years, i.e., up to 1975, the midpoint between 1961 and 1990), FTC = forest tent caterpillar (*Malacosoma disstria* Hbn.) defoliation index, CMI = climate moisture index (cm) (annual precipitation-annual potential evapotranspiration), G = annual tree basal area increment (cm,yr per tree), GD = numbers of growing days during a season, ES = evaporative stress (miles per hour times inches of mercury), P = monthly total precipitation (mm or inches), T = monthly mean temperature (°C), C = climatic signal residuals of tree ring growth (dimensionless), ST = sum of monthly mean temperature (°C), SP = sum of monthly precipitation (mm), and AT = mean air temperature (°C) in June and July. Dependent variables: SP = sum of monthly precipitation (mm), G = annual tree basal area increment (cm,yr per tree), $\hat{y}$= mean total wood growth (sum of ring widths of the whole tree [inches]), Y = annual increment according to perimeter (cm), I = ring width index (dimensionless), C = climatic signal residuals of tree ring growth (dimensionless), and RW = ring width (mm).

[b]The inverse transformation was used to predict tree growth rate.

8.2.4 Dimensionless analysis and identification of dendroclimatic equations

To amplify the climate signal and consistently compare tree growth rate estimates of each function, a dimensionless transformation for the selected equations was implemented by:

$$F_{ij} = \hat{Y}_{ij}/Y_{i1} \tag{8.2}$$

where F_{ij} is the relative climate response rate (RCR) of tree growth, that is, the ratio of tree growth estimates from Eq. i at the jth level of a climate change scenario to the estimate at level one; $\hat{Y}_{ij}$ is tree growth estimates from Eq. i at the jth level of a climate change scenario; and Y_{i1} is the estimates from Eq. i at level one.

The dimensionless transformation highlighted response of tree growth rate to climate changes, reduced the impacts of non-climate change factors on predictions and removed dimensions of dependent variables of the selected equations. The RCR was, in effect, a relative index of tree growth rate in response to climate change and could be used to quantify the potential of existing equations for predicting tree-radial growth increments under the conditions of climate change.

We defined a 95% confidence interval for the means of the RCR estimates to identify the average equations for predicting tree growth rate at the three sites. The 95% confidence interval for the means of the RCR estimates was

Table 8.3 Features of the dendroclimatic equations used in this study.

Function no.	Tree species	Location	Altitude (m a.s.l.)	Mean annual temperature (°C)	Mean annual precipitation (mm)	Sample size (no. of chronology cores)	Sample age range (yr)	R^2 or Pr	Source
1	White spruce (*Picea glauca* (Moench) Voss.), lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.)	50°00′N, 109°00′W[a] (Saskatchewan, Canada)	1500	5.5	322.6	43	$\leqslant$ 200	R^2= 0.365	Sauchyn and Beaudoin 1998
2	Trembling aspen (*Populus tremuloides* Michx.)	53°51′N, 109°08′W (Saskatchewan, Canada)	680	0.1-1.0	390.0-449.0	138	46-72	R^2= 0.832	Hogg and Schwarz 1999
3	White spruce	53°51′N, 109°08′W (Saskatchewan, Canada)	680	0.1-1.0	390.0-449.0	12	18-98	R^2= 0.602	Hogg and Schwarz 1999
4	Trembling aspen	60°41′N-62°03′N, 135°04′W-136°07′W (Yukon Territory, Canada)	610-730	-1.0	269	50	84-149	R^2=0.569, P <0.05	Hogg and Wein 2005
5	Regenerating trembling aspen	60°41′N-62°03′N, 135°04′W-136°07′W (Yukon Territory, Canada)	610-730	-1.0	269	54	84-149	R^2=0.324, P <0.05	Hogg and Wein 2005

Continued

Function no.	Tree species	Location	Altitude (m a.s.l.)	Mean annual temperature (°C)	Mean annual precipitation (mm)	Sample size (no. of chronology cores)	Sample age range (yr)	R^2 or Pr	Source
6	White spruce	60°41′N-62°03′N, 135°04′W-136°07′W (Yukon Territory, Canada)	610-730	-1.0	269	54	84-149	R^2=0.451, P <0.05	Hogg and Wein 2005
7	Trembling aspen	55°10′N, 118°47′W (Alberta, Canada)	669	1.85	453	42	43-99	R^2=0.758, P <0.001	Hogg et al. 2002
8	Black spruce (*Picea mariana* (Mill.) B.S.P.)	55°20′N, 114°34′W (Alberta, Canada)	400-800	-2.0-1.0	71-424.7	51	87	R^2=0.23, P <0.01	Dang and Lieffers 1989
9	Black spruce	55°20′N, 114°34′W (Alberta, Canada)	400-800	-2.0-1.0	71-424.7	50	87	R^2=0.29, P <0.01	Dang and Lieffers 1989
10	Black spruce	55°20′N, 114°34′W (Alberta, Canada)	400-800	-2.0-1.0	71-424.7	50	87	R^2=0.30, P <0.01	Dang and Lieffers 1989
11	White spruce	65°20′N, 138°20′W (Yukon Territory, Canada)	1230	-3.3	326	10-67	-	R^2=0.40, P <0.01	D'Arrigo et al. 2004
12	Red oak (*Quercus rubra* L.)	43°08′N, 89°20′W (Wisconsin, USA)	271-287	6.67	787.4	25	75-139	Pr <0.05	Shulman and Bryson 1965
13	Bur oak (*Quercus macrocarpa* Michx.)	43°08′N, 89°20′W (Wisconsin, USA)	271-287	6.67	787.4	5	78-139	Pr < 0.05	Shulman and Bryson 1965

Continued

Function no.	Tree species	Location	Altitude (m a.s.l.)	Mean annual temperature (°C)	Mean annual precipitation (mm)	Sample size (no. of chronology cores)	Sample age range (yr)	R^2 or Pr	Source
14	Ash (*Fraxinus pennsylvanica* var. *subintegerrima* (Vahl) Fem.)	43°08′N, 89°20′W (Wisconsin, USA)	271-287	6.67	787.4	13	10-56	$Pr < 0.05$	Schulman and Bryson 1965
15	Mixedwood	56°05′N, 21°50′E[a] (USSR)	105	-	559.0	-	-	$R^2 = 0.998$	Kairiukstis and Stravinskiene 1992
16	White fir (*Abies alba* Mill.)	44°57′N, 6°08′E[a] (France)	1520-1820	7.5	713.0	302	-	$R^2 = 0.894$	Rolland 1993
17	Earlywood and latewood, Scotch pine (*Pinus sylvestris* L.)	63°60′N, 30°40′E (Finland)	106	2.0	583.0	116	6-112	$R^2 = 0.490$	Miina 2000
18	Earlywood, Scotch pine	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	6-112	$R^2 = 0.420$	Miina 2000
19	Latewood, Scotch pine	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	6-112	$R^2 = 0.340$	Miina 2000

Continued

Function no.	Tree species	Location	Altitude (m a.s.l.)	Mean annual temperature (°C)	Mean annual precipitation (mm)	Sample size (no. of chronology cores)	Sample age range (yr)	R^2 or Pr	Source
20	Ratio of latewood ring-width to earlywood ring-width, Scotch pine	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	6-112	$R^2 = 0.320$	Miina 2000
21	Earlywood and latewood, Scotch pine	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	N, A	$R^2 = 0.500$	Pasanen 1998; Miina 2000
22	Earlywood and latewood, Norway spruce (*Picea abies* (L.) Karst.)	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	141	11-289	$R^2 = 0.620$	Miina 2000
23	Earlywood, Norway spruce	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	11-289	$R^2 = 0.560$	Miina 2000
24	Latewood, Norway spruce	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	11-289	$R^2 = 0.430$	Miina 2000
25	Ratio of latewood ring-width to earlywood ring-width, Norway spruce	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	11-289	$R^2 = 0.400$	Miina 2000

Continued

Function no.	Tree species	Location	Altitude (m a.s.l.)	Mean annual temperature (°C)	Mean annual precipitation (mm)	Sample size (no. of chronology cores)	Sample age range (yr)	R^2 or Pr	Source
26	Earlywood and latewood, Norway spruce	63°60′N, 30°40′E (North Karelia, Finland)	106	2.0	583.0	-	11-289	$R^2 = 0.560$	Pasanen 1998; Miina 2000
27	Scotch pine	67°22′N, 29°39′E (Finland)	120	-1.5-1.5	404.0-537.0	191	$\leqslant 79$	$R^2 =$ 0.180-0.254	Nöjd and Reams 1996; Nöjd and Hari 2001
28	Stone pine (*Pinus cembra* L.)	49°19′N, 19°57′E[a] (Poland)	1675	2.0-5.0	NA	-	$\leqslant 170$	$R^2 = 0.772 \pm 0.052$	Bednarz 1982
29	Engelmann spruce (*Picea engelmannii* (Parry) Engelm.)	40°3′N, 105°35′W (Colorado, USA)	3388.33	-1.57	930	40	-	$R^2 = 0.482$	Hansen-Bristow et al. 1988
30	Cajander Larch (*Larix cajenderi* Mayr.)	69°07′29″N to 70°32′23″N, 143°56′18″E to 150°16′13″E (Northern Yakutia, Russia)	105	-9.00	200-700	355	> 300	$R^2 = 0.65$	Hughes et al. 1999; Vaganov et al. 2006

[a]The latitude and longitude of the sampling sites were estimated by the authors using World Sites Atlas, and the elevations were obtained by personal communication with the original authors, because they were not reported in the original literature.

Note: asl = above sea level; NA = not available.

calculated as follows:

$$\bar{x} \pm t_{n-1,1-\alpha/2} * \frac{\sigma}{\sqrt{n}} \tag{8.3}$$

where $\bar{x}$ is the mean of the RCR estimates of the 30 equations, σ is the standard deviation of the RCR estimates, n is the sample size (herein n=30), and $t_{n-1,1-\alpha/2}$ is the upper 1- $\alpha/2$ critical point from student's t distribution with n-1 degrees of freedom. Herein, t=2.045 for two tails with 29 degrees of freedom for the 95% confidence interval. The equation whose RCR estimates fell within the confidence interval was considered to be the least-risk equation for a given scenario and site. For all scenarios and sites, the dendroclimatic equation with the highest frequency of RCR estimates falling within the confidence interval was approximated as the best equation, i.e., the lowest biased equation for the large-spatial scale boreal forest.

8.2.5 Validation of the selected equations

We used 11 standardized tree-ring width index datasets of west-central Canada to validate applicability of the selected equations of tree-radial growth increments to our study area. The tree-ring network datasets were excerpted from the International Tree-Ring Data Bank and World Data Center for Paleoclimatology (http://hurricane.ncdc.noaa.gov/pls/paleo/fm_createpages.treering). According to locations of sampling sites, we grouped tree ring data of white spruce from Chenal des Quatre Fourches (58°47′N, 111°27′W), Claire River (58°53′N, 111°00′W) and Revillon Coupe (58°52′N, 111°18′W), Peach River (58°59′N, 111°30′W) and Athabasca River (58°22′N, 111°30′W) for our study site of Fort McMurray A, Alberta, white spruce and balsam fir from Boreas SSA (55°07′N, 105°09′W), tamarack and black spruce from Boundary Bog (53°57′N, 106°20′W) and Esker (57°42′N, 105°16′W) for the study site of La Ronge A, Saskatchewan, and black spruce from Gunisao Lake (53°30′N, 96°23′W and at) for Thompson A, Manitoba, respectively.

8.3 Results

This section will show that the response of tree growth rate to climate change was positively correlated with both precipitation and temperature in west-central Canada. A dendroclimatic equation associated with current summer temperature and current May and previous summer precipitation can be identified as having average predictions of tree growth rate for the boreal forest at the landscape scale.

8.3.1 The relative climate response rate (RCR) of tree growth

The mean RCR estimates of the 30 selected equations were positively correlated with annual precipitation and yearly mean temperature, for all climate change scenarios and sites. Correlation between RCR estimates and precipitation all was at confidence level of $P < 0.01$, which was generally higher for La Range A than for Fort McMurray A and Thompson A. For Fort McMurray A and Thompson A, temperature was more important than precipitation except for scenario I (Table 8.4). For scenarios I, II, III, IV, and V, for example, the correlation between RCR estimates and precipitation ranged from 0.28 to 0.95 for Fort McMurray A, 0.69 to 1.00 for La Ronge A, 0.40 to 0.90 for Thompson A, and 0.48 to 0.97 for the three-site mean, while the corresponding values between RCR and temperatures varied from 0.16 to 0.95 for Fort McMurray

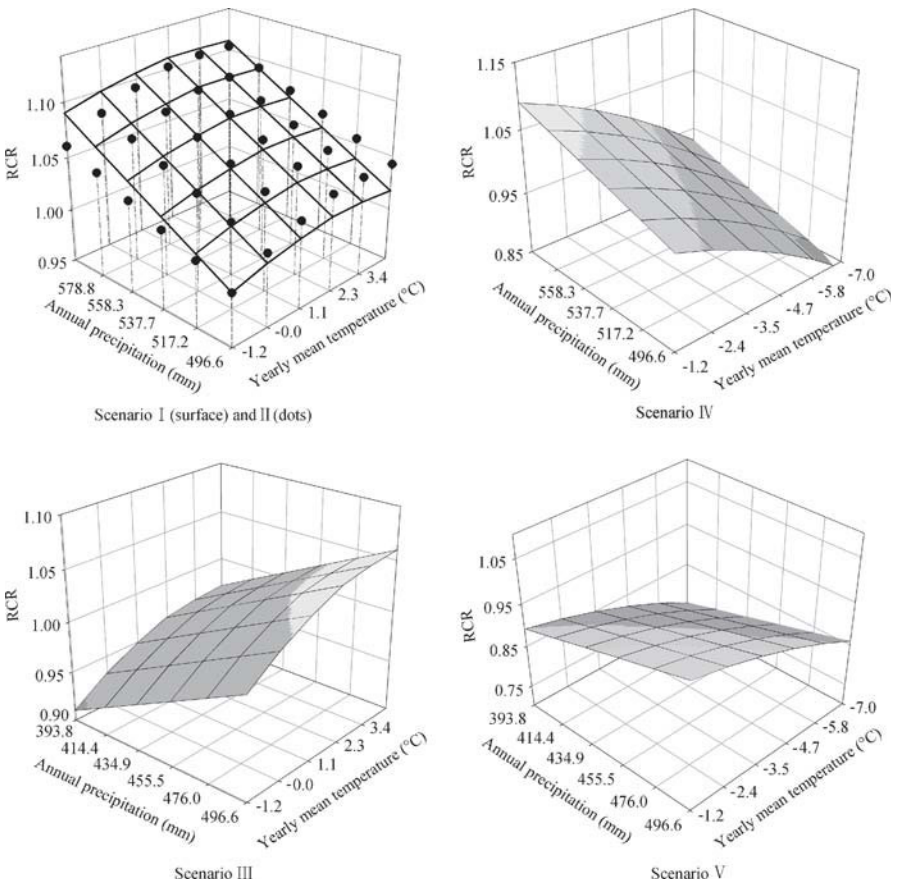


Fig. 8.2 Response of tree growth rate to climate change expressed as relative climate change rate (RCR) for scenarios I-V

Table 8.4 Correlation coefficients for the means of the relative climate response rate (RCR) estimates and climatic variables.

Climatic	Scenario													
variables	I	P	II	P	III	P	IV	P	V	P	VI	P	VII	P
Fort McMurray A														
Temperature	0.16	0.3656	0.65	< 0.0001	0.89	< 0.0001	0.95	< 0.0001	0.88	< 0.0001	-[b]	-	0.78	0.0681
Precipitation	0.95	< 0.0001	0.57	0.0002	0.42	0.0105	0.28	0.0945	0.45	0.0053	1.00	< 0.0001	-	-
La Ronge A														
Temperature	0.02	0.9207	0.28	0.1036	0.18	0.2891	0.72	< 0.0001	0.62	< 0.0001	-	-	0.85	0.0324
Precipitation	1.00	< 0.0001	0.95	< 0.0001	0.97	< 0.0001	0.69	< 0.0001	0.78	< 0.0001	1.00	< 0.0001	-	-
Thompson A														
Temperature	0.42	0.0107	0.76	< 0.0001	0.61	< 0.0001	0.91	< 0.0001	0.85	< 0.0001	-	-	0.96	0.0023
Precipitation	0.90	< 0.0001	0.62	< 0.0001	0.77	< 0.0001	0.40	0.0146	0.52	0.0012	1.00	< 0.0001	-	-
3-site mean[a]														
Temperature	0.20	0.2458	0.56	0.0004	0.55	0.0006	0.87	< 0.0001	0.79	< 0.0001	-	-	0.92	0.0105
Precipitation	0.97	< 0.0001	0.79	< 0.0001	0.82	< 0.0001	0.48	0.0029	0.61	< 0.0001	1.00	< 0.0001	-	-

[a]RCR estimates obtained with three-site average climate change scenario in Table 8.1.
[b]- = not applicable.

A, 0.02 to 0.72 for La Ronge A, 0.42 to 0.91 for Thompson A, and 0.20 to 0.87 for three-site means, respectively. Fig. 8.2 also illustrates that the RCR surfaces all were positively related to changes in precipitation and temperature in the seven scenarios (Fig. 8.2a, 2b, 2c and 2d).

The roles that changes in precipitation and temperature played in predicting tree growth rate were site-specific. At Fort McMurray, the correlation between RCR estimates and temperatures was 0.16, 0.65, 0.89, 0.95 and 0.88 for scenarios I, II, III, IV, and V, which were similar to Thompson A (0.42, 0.76, 0.61, 0.91 and 0.85) but higher than La Ronge A (0.02, 0.28, 0.18, 0.72 and 0.62). By contrast, the correlation coefficients between RCR estimates and precipitation at Fort McMurray varied from 0.28 to 0.95 for scenarios I to V, which were lower than those at La Ronge and Thompson (Table 8.4).

The RCR estimates had a more significant relationship with the changes of precipitation and/or temperature in summer than in winter. For example, with precipitation fixed at level 1, as yearly mean temperature increased from level 1 to 6, the mean RCR of the three-site mean increased from 1.00 to 1.04 for scenario II, which was higher than that of scenario I (1.00 to 1.02) (Fig. 2a). Similarly, when temperature was fixed at level 1, and precipitation increased from level 1 to 6, the mean RCR estimates increased from 1.00 to 1.09 for scenario I, which was higher than that of scenario I (1.00 to 1.04)(Fig. 8.2a).

8.3.2 Variability of the RCR estimates

Coefficient of variation (CV%) of the RCR estimates was positively correlated with the magnitude of temperature and precipitation change, for all scenarios and sites. As temperature and precipitation increased from level 1 to 6, the CV% for the three-site means increased from 0.00 to 24.49%, 38.27%, 43.48%, 52.83%, 58.59%,12.24%, and 39.15%, respectively, for scenarios I to VII (Fig. 8.3).

Temperature took a greater part in variability of RCR estimates than precipitation did. For the three-site mean, the correlation coefficients of between CV% of RCR estimates and temperature were up to 0.89, 0.96, 0.93, -0.96, -0.93, and 1.00 for scenario I, II, III, IV, V, and VII, respectively, while the CV% for precipitation was 0.34, 0.12, -0.28, 0.12, -0.22, and 1.00 for scenario I, II, III, IV, V, and VI, respectively (Table 8.5).

The changes of temperature and precipitation in summer contributed more to CV% of RCR estimates than changes in winter. For example, in scenario I (characterized by an increase of summer precipitation), the CV% was 0.00% to 12.62% for three-site means, which was higher than 0.00% to 12.24% of scenario II because of increase in winter precipitation (Fig. 8.3).

CV% of the RCR estimates was site-specific. As temperature and precipitation changed from level 1 to 6, the maximum CV% of RCR estimates

Table 8.5 Correlation coefficients between CV% of the relative climate response rate (RCR) estimates and climatic variables.

Climatic variables	Scenario													
	I	P	II	P	III	P	IV	P	V	P	VI	P	VII	P
Fort McMurray A														
Temperature	0.95	< 0.0001	0.97	< 0.0001	0.96	< 0.0001	-0.97[b]	< 0.0001	-0.94	< 0.0001	-[c]	-	0.99	< 0.0001
Precipitation	0.22	0.2060	0.10	0.5676	-0.18	0.2984	0.12	0.4885	-0.21	0.2082	1.00	< 0.0001	-	-
La Ronge A														
Temperature	0.68	< 0.0001	0.86	< 0.0001	0.80	< 0.0001	-0.79	< 0.0001	-0.63	< 0.0001	-	-	0.99	< 0.0001
Precipitation	0.66	< 0.0001	0.38	0.0231	-0.54	0.0007	0.55	0.0005	-0.72	< 0.0001	1.00	< 0.0001	-	-
Thompson A														
Temperature	0.81	< 0.0001	0.96	< 0.0001	0.93	< 0.0001	-0.97	< 0.0001	-0.95	< 0.0001	-	-	1.00	< 0.0001
Precipitation	0.48	0.0028	0.11	0.5223	-0.31	0.0616	0.05	0.7774	-0.15	0.3864	1.00	< 0.0001	-	-
3-site mean[a]														
Temperature	0.89	< 0.0001	0.96	< 0.0001	0.93	< 0.0001	-0.96	< 0.0001	-0.93	< 0.0001	-	-	0.99	< 0.0001
Precipitation	0.34	0.0447	0.12	0.4954	-0.28	0.1001	0.12	0.4773	-0.22	0.2019	1.00	< 0.0001	-	-

[a] RCR estimates obtained with three-site average climate change scenario in Table 8.2.
[b] The negative values of correlation coefficients between CV% and climatic variables showed positive correlation for scenario III, IV and V, because temperature and/or precipitation were decreased for those test scenarios.
[c] - = not applicable.

ranked as follows: Thompson A (98.21%)>La Ronge A(62.77%)>Fort McMurray A(44.67%).

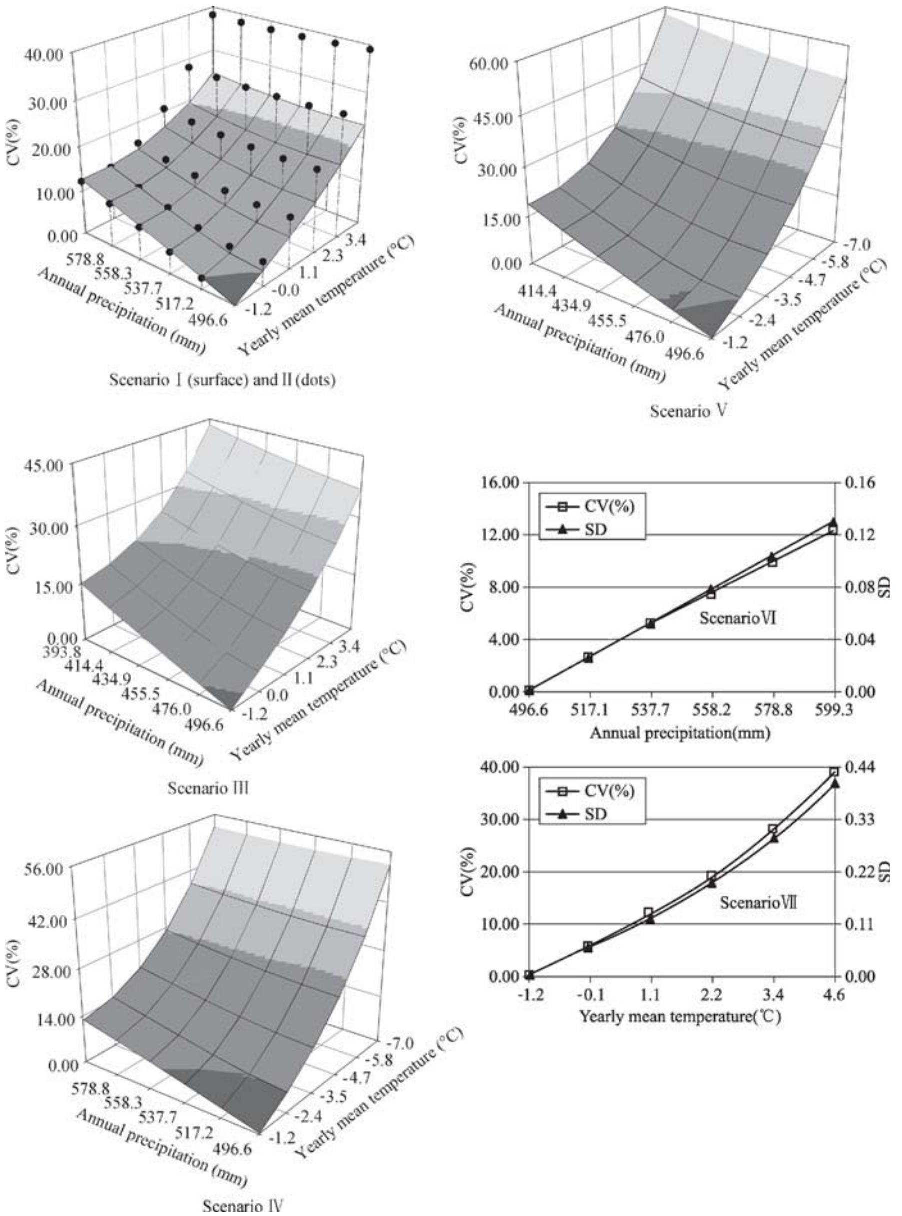
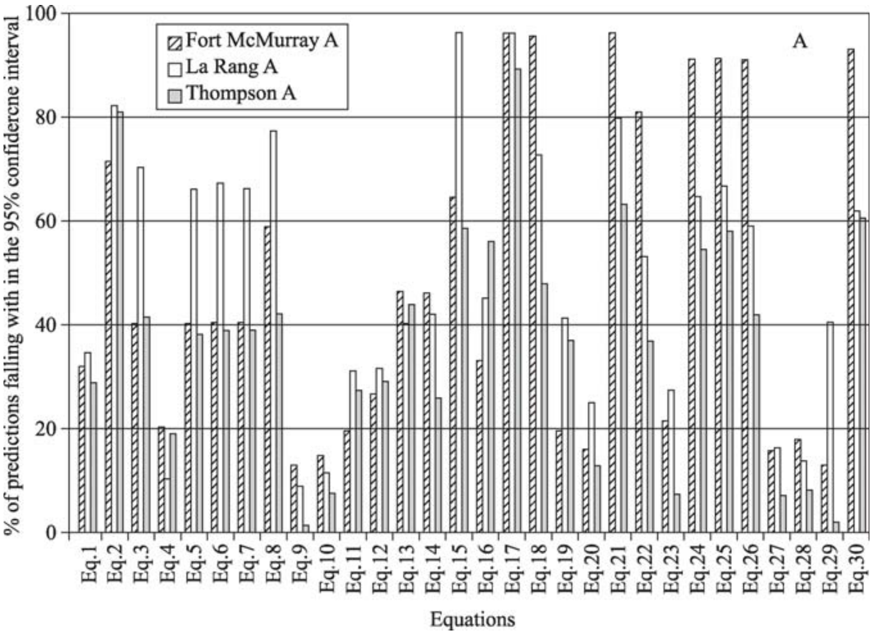


Fig. 8.3 Variations of RCR estimates expressed by coefficient of variation (%) of the three-site RCR mean estimates for seven scenarios.

8.3.3 Identification and validation of the selected equations

Because of the large variability of RCR estimates of the selected equations, we introduced a 95% confidence interval for the mean of the RCR estimates to identify the best equation. Fig. 8.4 illustrates frequencies of the RCR estimates of the selected equations fall within the confidence interval of 95% for individual site (i.e., 192 predictions per equation under the seven test scenarios) and sum for all three sites (i.e., 768 predictions for each function under the test scenarios). For Thompson A, probability of RCR estimates of Eq. 17, which fell within the confidence interval, was the highest among the 30 selected equations. For La Range A, however, frequencies of RCR estimates of Eq. 2 falling within the confidence interval were the highest. For Fort McMurray A, the top four probabilities of RCR estimates falling within the confidence interval were Eq.30> Eq.24> Eq.17=Eq.21. For the 3-site mean and sum of three sites, RCR estimates of Eq.17 ranked the highest probability whose RCR estimates fell within the confidence interval.

Estimates of tree-ring width index by Eq.17 were comparable to the averages of standardized tree-ring index network data. The predictions of Eq. 17 with 30-yr mean climate variables (1961-1990) (i.e., level 1 in Table 8.1) were 1097.19, 755.12, 746.15 and 796.0 for Fort McMurray A, La Ronge A, Thompson A and 3-site mean, respectively, while 30-yr averages of the measured tree-ring width index were 1432.40, 951.04, 949.93 and 1013.89 for the above sites. The root mean square error (RMSE) was 22.53% between the



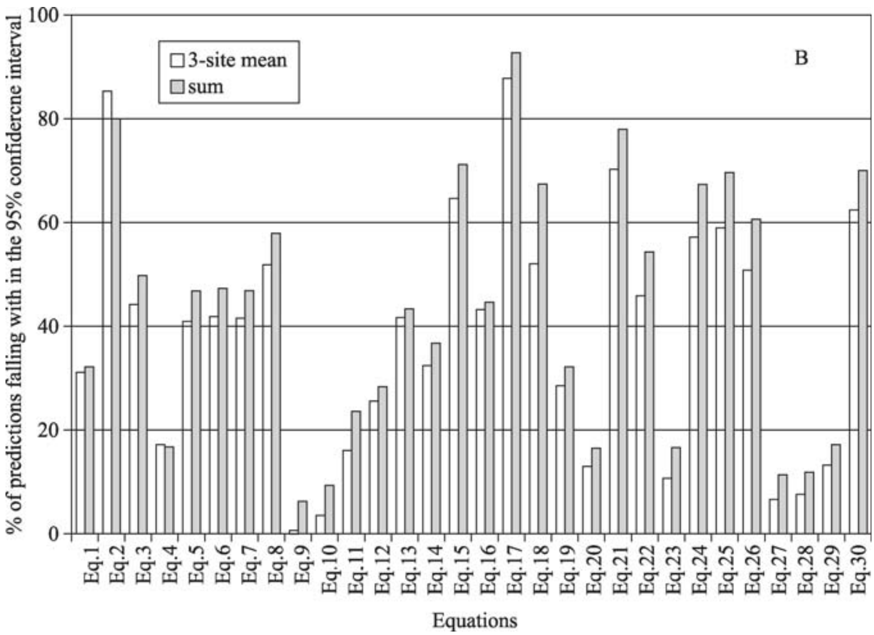


Fig. 8.4 Probabilities of the RCR estimates of the selected equations falling within the confidence interval of 95% for individual site (A) and the 3-site means and sum (B).

predictions and measurements. Thus, Eq. 17 can be approximated as a proper equation for the three sites.

8.4 Discussion

In this section we discuss the role of climate factors in predicting tree growth dynamics of boreal forest in response to climate change.

8.4.1 Major climate predictors of boreal forest in west-central Canada

For dendroclimatic equations across northern United States and Canada, site differences are usually more significant than species differences, and for a particular site, the limiting climate factors usually play a leading role in fitting these equations (Fritts 1974; Gray and Pilcher 1983; Cook et al. 1987; St. George and Luckman 2001; Hogg et al. 2002; Watson and Luckman 2002, 2004; Hogg and Wein 2005). In the southwestern Canadian plains, for example, precipitation is the major variable in the chronology equations because forest vegetation showed a more significant relationship to moisture availability than to temperature (Sauchyn and Beaudoin 1998; Hogg and Schwarz 1999). In the Northwest Territories, however, temperature is the main predictor in the dendroclimatic equations of tree-radial growth increments because in that area, temperature is too low and warm temperature becomes more important than precipitation to tree growth (Watson and Luckman 2002; D'Arrigo et al. 2004). Our results demonstrate that both temperature and precipitation positively contribute to the growth rate of trees for the three study sites, no matter how temperature and precipitation will change in the future. Temperature takes a crucial part in tree growth for Fort McMurray A and Thompson A, but precipitation plays an important role for La Ronge A where temperature is higher. Changes in summer temperature and precipitation will have more influence on tree growth rate than changes in winter temperatures and precipitation. For this reason, the appropriate equations should contain variables for summer temperature and precipitation for the study sites. These results confirm to previous studies showing that the degree of correlation of temperature with the tree-ring parameter was generally higher than corresponding values for precipitation for white spruce, black spruce, jack pine, and Engelmann spruce (*Picea engelmannii* Parry) in the Prairie provinces, Yukon and the Northwest Territories (Jozsa et al. 1984; St. George and Luckman 2001).

Therefore, when selecting the best dendroclimatic equations for predicting tree growth rate at different sites, users should take into account the differences in the major climatic stress factors at primary and target sites. For a large area, the selected equations should come from a low-elevation area and a multiple-species chronology of mixedwood rather than from a high-elevation area and single species, because the forest ecotones and tree lines in low-elevation areas usually represent the latitudinal ecological and climatic features of the ecozone, whereas in mountainous areas, local climate is more predominant because slope orientation, sun exposure, local air drainage, and elevation all influence the distribution of tree species (Brubaker 1982; Cook et al. 1987). In this study, Eq.17 could comprehensively predict tree growth rates under the impacts of temperature and precipitation change, because it included two independent variables: current summer temperature and current May and previous summer precipitation (Miina 2000). These results are consistent with previous studies for the Prairie provinces that both low precipitation and low temperatures stress tree growth (Case and MacDonald 1995; Sauchyn and Beaudoin 1998; Hogg and Schwarz 1999; St. George and Luckman 2001; Watson and Luckman 2002). The sampling site conditions for Eq. 17 (580 mm annual precipitation, 2.0°C yearly mean temperature and 106 m a.s.l. elevation) are similar to the mean for the study sites (496.6 mm annual precipitation, 1.2°C yearly mean temperature and 319.67 m a.s.l.). The tree-ring samples also contained earlywood and latewood, and the sample tree age included young to old trees (6 and 112 years) (Miina 2000).

Eqs.1-10 were not included within the confidence interval, though they were developed from sample sites in west-central Canada. Within these equations, tree growth is mainly associated with moisture availability or temperature rather than both temperature and precipitation variables in the testing scenarios for the large spatial scale area. We thus recommended Eq.17 for all three sites, while Eq. 2 including only moisture variable is more applicable to La Range A site where temperature is higher.

For a given equation, we derive the contribution of each climate variable to tree growth rate by using partial derivatives: Let $RCR = f(T, P)$, then $\dfrac{\partial(RCR)}{\partial T} = \dfrac{\partial f(T,P)}{\partial T}$ and $\dfrac{\partial(RCR)}{\partial P} = \dfrac{\partial f(T,P)}{\partial P}$, where RCR= relative climate response rate, T= mean or accumulative temperature (℃) of a certain months of current or previous year, P= precipitation (mm) of a certain months of current or previous year, $\dfrac{\partial(RCR)}{\partial T}$ and $\dfrac{\partial(RCR)}{\partial P}$ represent contributions of temperature and precipitation to RCR. For Eq.17, for instance, $\dfrac{\partial(RCR)}{\partial ST_{\text{Jul}(t)\text{ to Aug}(t)}} = 1.19$, $\dfrac{\partial(RCR)}{\partial P_{\text{May}(t)}} = 0.11$ and $\dfrac{\partial(RCR)}{\partial SP_{\text{Jun}(t-1)\text{ to Jul}(t-1)}} = 0.073$. This suggests that the contribution of changes in temperature to RCR is much higher than that in precipitation. The prerequisite that the derivatives are meaningful is that the equations of tree-radial growth increments must be applicable to the target study area.

8.4.2 Uncertainties of the predictions of tree growth in response to climate change

Variations of the predicted response of tree growth rate to climate change increase as the magnitude of climate change rise (Fig. 8.3). For the three study sites, variation in the climate response estimate was caused mainly by changes in temperature. The changes of temperature and precipitation in summer had contributed more to the uncertainty of equation predictions than changes in winter, particularly for the scenarios with a decrease of temperature and/or precipitation in summer (e.g., scenarios IV-V). Further work may be needed to evaluate the selected equations with nonlinear climate change scenarios.

For the dendroclimatic equations, improvement of climate signal-to-noise ratio is necessary to evaluate response predictions to climate change. Apart from climate variables, uncertainty of predictions may also be from non-climatic variables and the intrinsic flaws in the equations. The dendroclimatic equations were developed using historical climate records and tree ring chronology, underlining that the relationship between tree growth and climate is stable as climate changes, and does not reveal the ongoing and future relationship between tree growth and climate change. Moreover, the linear equations do not include the ecophysiological cause and relationship of climate change and tree growth, and do not consider interrelations and interactions between plant developmental processes (e.g., photosynthesis, assimilation, respiration, etc.) in a multitude of possible impacts such as adaptability and feedback of tree to external micro-and macro-environment (Bednarz 1982; Brubaker 1982; Rathgeber et al. 2000; Girardin et al. 2008). For these reasons, some authors have adopted aggregated equations of tree-radial growth increments to reveal nonlinear ecophysiological effects and nonclimatic factors (Pan and Raynal 1995a, 1995b; Rathgeber et al. 2000; Yeh and Wensel 2000). Hence, the RCR dimensionless transformation will be required to amplify climatic signal-to-noise ratio of tree growth.

8.4.3 Identification of the dendroclimatic equations

The identification method of a confidence interval for the mean of the RCR estimates could be used to search the average and low-risk equations for a large spatial area because of the difficulties of the construction, calibration, and verification of a high-quality equation for various sites. The rationale of the evaluation method is supported by previous researches that regional tree-ring width or climate reconstruction from two or more equations of different structure could be averaged, the point-based estimates within homogeneous climate regions could be averaged, and averaging may increase the signal-to-noise ratio of reconstruction by reducing the randomly related differences and errors of tree-ring measurement (Fritts 1991; Hughes et al. 1999; Naurzbaev et al. 2004; Vaganov et al. 2006). This method is relatively low-cost, labor- and time-saving, and may provide an alternative method of identifying the existing dendroclimatic equations for users who wish to approximately predict tree growth rate in response to climate change at a large spatial scale rather than accurately estimate tree growth for a specific local site. For example, Eq.17 recommended in our study can give an average and low-risk approximation of tree growth rate responding to climate change for the three sites.

The fundamentals of the evaluation method are to establish the testing climatic scenario datasets for a study area and adequately collect the existing dendroclimatic equations whose sampling sites are similar to the target study area. Unfortunately, the collection of existing dendroclimatic equations is impeded because they are often published in graphic format of correlation coefficients or principle components, and are impossible to apply accurately (Brubaker et al. 1982; Jozsa et al. 1984; St. George and Luckman 2001; Briffa et al. 2002; Watson and Luckman 2002, 2004). We suggest that researchers should publish the raw data from which their regressions were developed, as well as the equations themselves rather than graphs in the future, so that users can synthesize the dendroclimatological knowledge-based equations to predict tree growth dynamics responding to climate change.

8.5 Conclusions

In this study, we developed a method to produce an approximation of growth dynamics response of boreal forest to climate change by using existing knowledge from the research field of dendrochronology. The main steps of the approach included collection of existing dendroclimatic equations from the sampling sites which are similar to target study area, amplification of climate signal of the equations, development of testing of climate change scenarios based on existing climate change projections, evaluation of the equations at a certain confidence level and validation of the equation with tree-ring network datasets. The dendroclimatic equations should contain major climate variables of the target study area. By taking an example of the boreal forest of west-central Canada, our results suggested that response of tree growth rate to climate change is positively correlated with both temperature and precipitation, especially the changes of temperature and precipitation in summer. Through the 95% confidence interval for the mean of tree growth rate estimates and validation with tree-ring network data, the following equation has been identified as having average predictions of tree growth rate that are appropriate for landscape scale simulations in the boreal forests of west-central Canada:

$$I_t = 0.68I_{t-1} + 50.72 + 1.19ST_{\text{July}(t)\text{ to Aug}(t)} + 0.11P_{\text{May}(t)} + 0.073SP_{\text{June}(t-1)\text{ to July}(t-1)}$$

where I is ring width index, P is monthly precipitation (mm), T is monthly mean temperature (℃), ST is sum of monthly mean temperature (℃), SP is sum of monthly precipitation (mm), and subscripts t and $t-1$ are years t and $t-1$, respectively.

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Part III
Emerging Approaches in Forest Landscape Conservation

Chapter 9 The Next Frontier: Projecting the Effectiveness of Broad-scale Forest Conservation Strategies

Janet Silbernagel*, Jessica Price, Randy Swaty and Nicholas Miller

Abstract

Conservation and land management organizations such as The Nature Conservancy are developing conservation strategies to distribute protection efforts over larger areas and a broader range of ownership and management techniques. These "distributed conservation strategies," such as working forest conservation easements, are based on the premise that blending resource extraction, such sustainable timber harvest, and conservation should yield greater socio-economic benefits without significantly compromising the conservation of biodiversity or the sustainable provisioning of ecosystem services. However, it is unknown how well these strategies will compare to traditional conservation preserves or if they will be robust to climate change and resource demand over the coming centuries. Due to scarce financial resources and the relative difficulty of negotiating easement acquisitions, it is important for forest conservation and management organizations to know which strategies most effectively meet conservation goals. Meanwhile, the long duration required to evaluate most monitoring questions leads to a lag in knowledge transfer and delayed adaptive management. In this chapter, we discuss the challenges and constraints to measuring conservation effectiveness and illustrate a scenario-building approach that we are applying to understand and compare the conservation effectiveness of various conservation strategies in two large conservation acquisitions in the Great Lakes region of the United States. We show how this approach can be used to evaluate

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potential outcomes for biodiversity and the provision of ecosystem services resulting from varying conservation strategies and discuss implications of this approach for the future of forest conservation.

Keywords
Conservation effectiveness, expert knowledge, landscape modeling, scenario-building, spatial narratives, biodiversity, ecosystem services, climate change.

9.1 Introduction

In the face of a rapidly changing world that includes globalization, climate change, trends in population growth, and the accompanying increase in resource and energy demands, innovative forest conservation strategies could play an important role in how land is allocated and used. However, the typical size, costs, lack of historical examples, and local or regional implications make development and implementation of innovative management and conservation options particularly challenging. Additionally, the conservation effectiveness for broad-scale forest conservation actions depends largely on their social legitimacy. That is, persons that may be affected by or are responsible for implementing these actions must be allowed to have a voice in the decision-making process (Daniels and Walker 2001). Moreover, the public at large—stakeholders, community groups, indigenous peoples, and local experts—are becoming more connected to conservation decision-making for several reasons, including the cross-boundary requirements of many conservation targets and strategies, ease of communication through information technology advances, and heightened interest. Thus, the trend toward participatory decision-making in conservation has contributed toward investment in sustainable forest management options that balance the interests and needs of multiple stakeholders.

After setting the context of historical and traditional conservation thought in the United States, we will discuss scenario-building and modeling approaches designed to evaluate the conservation effectiveness of emerging strategies.

9.1.1 A brief history of conservation

Forest conservation has a rich global history, with ideologies and practices simultaneously evolving in different geographical and cultural contexts. While important for understanding and applying conservation today, detailed recounting of this history is beyond the scope and purpose of this chapter. To situate our work within a historical context, we focus on the roots of forest conservation in the United States, where two prevailing ideologies concerning nature have informed forest conservation—the preservationist and conservationist perspectives.

The preservationist perspective grew out of the broader romantic-transcendentalist cultural movement of the 19th century, in which nature was viewed as an intrinsically valuable and inspirational part of divine creation. Importantly, this perspective placed humans outside of "nature", meaning that utilization and intervention in nature by humans was unnatural and destructive. Formative works that articulated and shaped the preservationist perspective include the writings of Ralph Waldo Emerson (*Nature*, 1863) and Henry David Thoreau (*Walden*, 1854). Naturalist and founder of the Sierra Club, John Muir also played a pivotal role in the preservation movement through his writings and advocacy, especially for the protection of the Yosemite Valley. Preservationist philosophy provided the basis for Muir's argument for preservation of natural areas irrespective of economic valuations.

Contemporary with the development of the preservationist perspective and in many ways a response to its ideology, the conservationist perspective viewed nature as useful for the provisioning of resources and materials for human consumption and to fuel economic growth. As a result, early conservation was largely aimed at the sustained harvest of particular species. This anthropocentric view was popularized largely by Gifford Pinchot, the first chief of the United States Forest Service (USFS), and the ideology of efficient and multiple uses of public lands, such as timber harvest, recreation, and hunting, remains a mandate of both the USFS and the Bureau of Land Management (BLM) today. Though President Theodore Roosevelt, a friend of Pinchot, was credited with nationalizing the conservation effort, Roosevelt was deeply concerned with species protection and also considered the preservationist perspective promoted by John Muir (Fig. 9.1).

The early dialogue between preservationists and conservationists inspired extensive research and discussion among both scientists and land managers. A synthesis of the preservation and conservation perspectives emerged in the mid-twentieth century. This "Ecological Land Ethic" was put forth most clearly in Aldo Leopold's *A Sand County Almanac* (1949), which describes nature as a system of interdependent components, some useful for human use and some not, all of which are required for proper functioning of the system. This "systems view" reflects the sophisticated understanding of both evolutionary and ecological processes that result in the functioning of ecosystems and their provisioning of goods and services. Importantly, from this perspective, humans are considered a component of the ecosystem whose influence, both positive and negative, must be understood and acknowledged in land management and conservation decision-making.

Fig. 9.1 President Theodore Roosevelt and John Muir on Glacier Point in Yosemite Valley, California in 1903. Photo courtesy of the Library of Congress.

9.1.2 Traditional conservation approaches

Just as the theoretical foundations of conservation have evolved, so have the goals of conservation and the strategies utilized to accomplish these goals. Conservation approaches have consistently been expanding in scale both spatially and ecologically. Advances in scientific methodology have expanded the scale at which humans are able to perceive and understand the environment, revealing that species and ecosystems require resources beyond a single preserve.

Early naturalists first observed ecological degradation on a relatively fine scale, noting the decline of individual species or natural areas, and linked this degradation with human presence and activity. As a result, ecological studies and conservation management were conducted at a local scale, with the establishment of nature reserves being aimed at excluding human activity. Also, conservation efforts often focused on the protection of individual species, as embodied by the Endangered Species Act of 1973. This approach was supported by the static equilibrium view of ecosystems, where human activities were viewed as unnatural and destructive. However, single species approaches to conservation can divorce the species from its ecological context.

Advancing ecological understanding and technology prompted conservation planning and approaches to expand to broader landscape scales.

Ecological research revealed that ecosystems were, in fact, dynamic, open systems that change over time in response to natural and anthropogenic disturbances. In parallel, ecological research and technology (computing power, remote sensing, and GIS) expanded the spatial scale at which ecosystems and processes could be investigated and understood. Subsequently, the subdiscipline of landscape ecology developed (Troll 1950; Turner et al. 2001). As a result, ecologists and conservation practitioners were able to understand the broad-scale dynamics of ecosystems and recognized that successful conservation efforts would need to be larger in scope and broader in scale to ensure the persistence of these important dynamics (Boutin et al. 2002).

9.1.3 Changing conservation

The broadening of conservation efforts in both scope and scale has forced conservation practitioners and land managers to address the important issue of defining the proper scale and boundaries of conservation units. Historically, political boundaries were the default boundaries of conservation units. These boundaries mostly followed a "defensible perimeter" without consideration of non-human issues unless they were of strategic importance with regard to resources or protection (e.g. rivers or cliffs). However, Lopez-Hoffman et al. (2009) noted that many species of animals regularly migrate across international borders; the same is likely the case for county and state borders. One tool that conservationists use to plan across political boundaries and define conservation units at the landscape scale is thematic maps focused on the biotic and abiotic properties that are "the basic units of nature on the face of the earth" (Tansley 1935).

A commonly used type of thematic map is an ecoregion map, which shows the Earth's surface subdivided into identifiable areas based on macroscale patterns of ecosystems, that is, areas within which there are associations of interacting biotic and abiotic features. These ecoregions delimit large areas within which local ecosystems recur more or less throughout the ecoregion in a predictable fashion on similar sites. In other words, there is relative homogeneity in the properties of an area (Omernick et al. 1997). While a number of scientists have mapped ecologically relevant characteristics, such as life zones (Holdridge 1967; Merriam 1898) and biotic provinces (Dasmann 1974), ecoregions are necessarily interdisciplinary due to the relationships between abiotic and biotic properties including geology, soils, climate, and nutrient cycling (Loveland et al. 2004). Bailey's ecoregions distinguish areas that share common climatic and vegetation characteristics (Bailey 1998, 2005). Ecoregion maps are useful in land management and conservation in a number of ways. For example, The Nature Conservancy combines ecoregion maps with information about the distribution of species, communities, and ecosystem functions and processes to assess the biodiversity and conservation importance of areas within an ecoregion, providing a working blueprint for long-term management and conservation.

Even with improved technologies and methods, scientists and land managers have found several challenges to developing conservation strategies at landscape scales. For example, most landscapes are divided into small parcels each with different owners. In this situation, gaining the support of enough landowners to implement broad-scale conservation strategies may be difficult. Alternatively, in landscapes with relatively few landowners, changes in land ownership may affect cooperative efforts over a large proportion of the project area. Also, voluntary landscape planning and management efforts are often difficult to fund and maintain and can be temporary as a result.

Despite these challenges, there are a growing number of compelling reasons to continue with landscape scale conservation. First, conservation opportunities are arising at unprecedented spatial scales, such as large corporate timber divestments (e.g. International Paper in the eastern and central United States). Second, while investments may be viewed as opportunities, there is great potential for accelerated landscape fragmentation if divested lands are not purchased as a whole or placed under a conservation easement that significantly limits subdivision. In addition, the successful conservation of species with large home ranges, such as many carnivore species, and species that require large, continuous forested areas, also depends on ecoregional or landscape scale strategies. Finally, climate change science suggests a need to conserve larger areas and connectivity to enable adaptation and ecosystem resilience (Millenium Ecosystem Assessment 2005b).

Not only has the scale of conservation efforts increased spatially to incorporate larger areas, but also conservation efforts are expanding in scope. Ecosystem services are increasingly recognized as an important basis and catalyst for conservation. Ecosystem services are the conditions and processes through which natural ecosystems, and the species that comprise them, sustain and fulfill human life (Daily 1997). More simply, they are the benefits that people obtain from nature, which range from aesthetic pleasure and recreation to pollination of crops to water and nutrient cycling (Diaz et al. 2005). "Provisioning" ecosystem services include resource extraction, such as harvest of timber or non-timber forest products. Recently, there has been an interest in forest areas that can supply woody biomass for energy production.

Additionally, conservation decision-making is engaging a broader range of stakeholders. Where government agencies had previously taken the lead in land management and protection, conservation organizations are more actively participating in and leading conservation efforts today, partnering with local, regional, and federal governments as well as land owners and land users to achieve conservation goals. Today, community-based and participatory decision-making in conservation are more common, where stakeholders, community groups, indigenous peoples, and local experts are significantly involved in conservation planning and decision-making. In fact, many conservation practitioners are looking to traditional or local ecological knowledge to inform plans and strategies (Agrawal et al. 1999). Public participation may not be appropriate to all conservation decision-making. Instead, many conservation practitioners collaborate with local experts to ensure locally and socially relevant decisions (Gustafson et al. 2006).

9.1.4 New directions in conservation

Conservation strategies are evolving in response to this expansion in scale and scope toward what we term “distributed conservation.” This approach spreads the economic and human resources available for conservation more thinly and across larger areas, as opposed to concentrated conservation efforts that focus on providing higher levels of protection to a smaller area. A concentrated conservation approach might purchase forest land to protect species of interest in a “reserve”, setting land aside from any extractive or working lands management. This may be optimal for some biodiversity targets, such as species relying exclusively on core habitat or species that are extremely sensitive to anthropogenic disturbance. However, strict preservation of relatively small areas is not effective for other targets, including wide-ranging species, landscape matrix species, species dependent on large-scale disturbances, and other non-species specific biodiversity targets, such as community-level targets and ecosystem services. On the other hand, a distributed conservation approach could protect forest land by investing in specific land resource rights. For example, the international market for forest carbon credits invests in the carbon resource of a forest while allowing continued sustainable uses (Millennium Ecosystem Assessment 2005b; O’Connor 2008). Conservation easements also offer distributed conservation, a way to protect biodiversity, especially from fragmentation, by taking land out of development while still allowing sustainable uses (e.g. resource management or harvest, some recreation). However, easements may also be seen as a compromise, and the implications of management restrictions on landowners must be taken into account.

Many of the assumptions that underlie distributed conservation strategies, such as working forest conservation easements (WFCEs), are untested and include risks, such as ecological, social, public relations, and economic risks. It is unclear if blending resource extraction (e.g. provisional ecosystem services) with conservation will yield a net conservation gain, that these broader, distributed strategies will more efficiently spread resources, or that today’s conservation strategies will be robust to climate change impacts over the coming centuries.

Ideally, all conservation actions are monitored over time, and insights provided by monitoring are integrated into the management regime. This adaptive management allows the conservation strategy to remain flexible and effective in the face of new information, disturbances, and unanticipated dynamics (Gregory et al. 2006; Moore et al. 2008). Both on-the-ground and remote sensing methods are an integral part of management and monitoring at the landscape scale and are often coupled to provide an understanding of conservation over the long term. However, a more comprehensive understanding of conservation effectiveness often requires monitoring efforts that span decades, likely exceeding the duration of current trends in forest divestiture or funding opportunities as well as the timeframe for effective mitigation of external disturbances such as climate change. Therefore, there is a clear need to incorporate methods that inform current conservation opportunities by providing insight into the potential future outcomes of conservation strategies for both biodiversity and ecosystem services.

9.1.5 Scenario-building and landscape modeling: an integrated approach

Scenario analysis offers environmental planning and monitoring a glimpse into the potential future outcomes of decision-making and external change. A scenario is an account of a plausible future (Peterson et al. 2003a). Scenarios have been used at least since WWII as a way of strategizing responses to opponents' actions. In the 1960's and 70's, scenario approaches were adopted as a business planning tool, particularly by the oil industry facing a rapidly changing global market (Mahmoud et al. 2009). In the context of this paper, a scenario represents, describes, and accounts for the conditions that lead to one or more alternative futures (Fig. 9.2). Rather than relying on predictions, which are quite uncertain under complex changing conditions, scenarios "enable a creative, flexible approach to preparing for an uncertain future," and recognize that several potential futures are feasible from any particular point in time (Mahmoud et al. 2009). Among the most well-known applications, the Millennium Ecosystem Assessment used scenario analysis to understand the consequences of global ecosystem change for human well-being (Millenium Ecosystem Assessment 2005a; Carpenter et al. 2006; Cork et al. 2006).

In regional environmental applications, scenario analysis is often integrated with landscape modeling to create spatially explicit alternative futures resulting from land management, policy, climate change, and resource or energy demand alternatives (Baker et al. 2004; Gustafson et al. 1996; Nassauer et al. 2007; Peterson et al. 2003a; Provencher et al. 2007; Sala et al. 2000; Santelmann et al. 2006; Santelmann et al. 2004; Schumaker et al. 2004; Sturtevant et al. 2007; Tilman et al. 2001; White et al. 1997; Wilhere et al. 2007; Zollner et al. 2008). More specifically, a *landscape* scenario refers to the different possible conditions and accounts that underlie landscape change (Nassauer and Corry 2004), where the alternative futures are *spatially explicit* representations of plausible landcover patterns (often generated by using landscape modeling). Thus in this context, scenario-building is the collaborative learning *process*

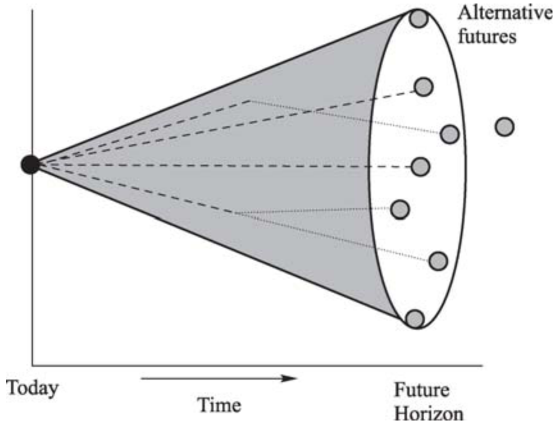

Fig. 9.2 Conceptual diagram of the use of scenario analysis to generate alternative futures (Mahmoud et al. 2009, adapted from Timpe and Scheepers 2003).

by which a team that includes stakeholders and/or experts defines the sets of conditions that will be used to generate future landscapes, and then simulates possible future land cover patterns based on those conditions. This synthesis can provide conservation practitioners and land managers with insight into the possible future landscape resulting from each scenario, enabling them to evaluate and compare the effectiveness of different strategies at achieving specific goals.

Approaches to scenario analysis vary broadly, and Mahmoud et al. (2009) provided a comprehensive review of the types and applications of scenario approaches. Generally, we talk about two types of scenarios: *exploratory* scenarios describe the future according to known process of change and extrapolations from the past. They can project forward using past trends (as with climate change), or anticipate upcoming change that significantly varies from the past (e.g. new demands for woody biomass for energy production). As an example, Metzger et al. (2006) considered vulnerabilities of ecosystem services across regions in Europe under various land use change scenarios. Their assessment showed, for example, that southern Europe may be particularly vulnerable to land use change. On the other hand, when alternative scenarios are developed to depict a desired or feared outcome and are utilized to develop strategies to achieve or avoid that outcome, respectively, they are referred to as *normative or anticipatory* scenarios (Mahmoud et al. 2009; Nassauer and Corry 2004). For example, normative scenarios were applied in an iterative, interdisciplinary process for visioning alternative agricultural futures in watersheds of the Upper Mississippi River valley. This team looked at water quality, biodiversity, farm economics, and aesthetics under three leading constituency goals: a) maximizing agricultural commodity production, b) improving water quality and reducing downstream flooding, and c) enhancing biodiversity within agricultural landscapes (Nassauer et al. 2007; Santelmann et al. 2004).

In either case (exploratory or anticipatory), scenarios can be developed through a collaborative process among various stakeholders and experts (Hulse et al. 2004; Peterson et al. 2003a; Theobald et al. 2005). In the case of forest landscape scenarios, the input of stakeholders and experts, such as landowners, foresters, and ecologists, can be used to set up the conditions of various strategies and to understand the alternative futures and contrasting trends that might result from those strategies. This participation can continue beyond scenario development to inform the iterative evaluation and implementation stages. For example, three alternative scenarios of varied ecosystem service use through 2025 were developed for a northern Wisconsin (USA) lake region. These scenarios sparked a discussion of alternative futures and helped local people consider how the region might develop (Peterson et al. 2003b). The collaborative learning process (Daniels et al. 2001; Gustafson et al. 2006) builds trust among diverse groups, lends social legitimacy to the outcomes of the process, and takes advantage of the place-based knowledge provided by these stakeholders. Put together, this approach recognizes that no amount of quantitative data or modeling alone can predict the dynamic behavior of complex natural systems (Fig. 9.3). Yet, teams working in specific places or systems can build scenarios informed by years of practical knowledge along with empirical and simulated data. Scenario analysis offers a framework for developing more resilient conservation policies when faced with uncontrollable, irreducible uncertainty (Peterson et al. 2003a).

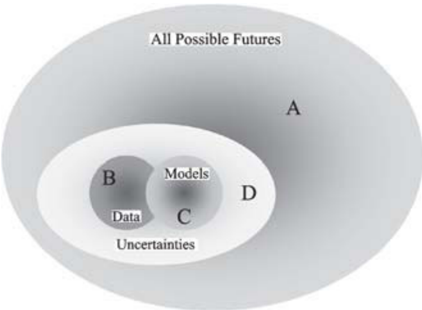

Fig. 9.3 The full set of possible futures (A) is only partially represented in available data (B) and models (C). Together, the data and the models allow us to project the uncertainties, or knowable unknowns (D). But there remain many unknown futures that may exist beyond our estimation of uncertainties (large grey ellipse). The probability of any model projection depends on the full set of possible futures, most of which are unknown (Carpenter et al. 2006, based on the ideas of L. A. Smith 2002).

Concerns about scenario analysis tend to center on the validity of the experts' knowledge and the selection of experts and stakeholders to be included in the process. Scientists at a recent landscape ecology workshop (US-IALE 2009) commented that if scenarios are built as stories without empirical data, the public will "think we don't know what we are doing." A related concern is that scenarios are not probabilistic, as they can include unlikely events or events to which a probability cannot be assigned. Indeed, sometimes scenarios with highly unlikely but very impactful events can be quite informative. For example, at the time of the oil embargo (1973-1974), scenario planning previously undertaken by Shell Oil helped the company to respond quickly to maintain stability in an unpredictable market (Mahmoud et al. 2009). Still, while scenarios can address many of the uncertainties in a system, they cannot necessarily be quantified (Fig. 9.3). Thus, a stigma or misunderstanding about how scenarios are formed, their purpose, and their credibility may still persist.

The other key component to building integrative landscape scenarios is the selection of appropriate landscape modeling software. In a review and classification of forest landscape models, Scheller and Mladenoff (2007b) provided a valuable classification based on three criteria. The first criterion is whether the model includes or excludes spatial interactions, referring to whether or not the model represents the movement of energy, matter, or information across the landscape (Reiners et al. 2001). The second criterion asks whether or not the software uses static or dynamic ecological communities. A particular model may keep an ecological community intact over time (static models), or the communities may shift to include or exclude new members (dynamic models). For example, Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies Ltd. 2009), an open-source state and transition model, has static successional classes that are user-defined communities. The amount of each successional class on the landscape can change, but the species composition will not. The third criterion is whether the model includes ecosystem processes. Modeling software that simulates ecosystem processes follows changes in net growth, biomass accrual, and decomposition. An example of such modeling software is LANDIS-II (Scheller et al. 2007a). But, with the addition of spatial interactions, dynamic communities and tracking of ecosystem processes comes increased complexity and inputs.

The process of selecting landscape modeling software can help to refine research objectives, define the audience, and set realistic goals (Sturtevant et al. 2007). For example, if the objective of the modeling exercise is to inform stakeholders of the potential outcomes of landscape scenarios, then the ability to explain the outputs and process in a meaningful way is important. This suggests working in a less complex modeling environment. Alternatively, if the audience for the modeling exercise is more academic in nature and the questions involve factors such as ecosystem processes, then selection of a more robust software package is warranted, if possible.

Like any approach to understanding complex systems, landscape modeling efforts present complexities and challenges. For example, obtaining reliable, correctly scaled inputs can be difficult and sometimes impossible. Ecological systems are driven by processes that are the foundation of ecological modeling software. For example, VDDT requires that probabilities be entered for each disturbance (transition) per time period (e.g. if the mean fire return interval is 100 years, then the annual yearly probability is 0.01). Often this information is lacking or is from a particular study site that may or may not be representative of the landscape under consideration. Sometimes it is necessary to make assumptions about particular disturbances or management actions. In a ecological modeling exercise, Provencher et al. (2007) were uncertain about the effectiveness of particular invasive treatments. In this situation, modelers are required to make assumptions based on best information or model multiple scenarios (e.g. treatments are 25%, 75% and 100% effective).

9.2 Template project: Wild Rivers Legacy Forest and Two Hearted River Watershed

We are applying scenario analysis coupled with landscape modeling to evaluate and compare the conservation effectiveness of both concentrated and distributed conservation strategies. These strategies include: 1) no conservation action, 2) persistence of current management strategies in the study areas, 3) all land in the study areas managed as a protected reserve aimed at biodiversity conservation, 4) all land in the study areas managed under a WFCE. An example of a distributed conservation strategy, WFCE's are based on the premise that sustained timber harvest and recreation activities should yield greater socio-economic benefits (ecosystem services) without significantly compromising the conservation of biodiversity. The possible future landscapes and potential outcomes for biodiversity and the provision of ecosystem services are evaluated for each alternative conservation strategy in the presence of external drivers of landscape change, including various climate change projections, development pressures, and demand for woody biomass for energy production in the Great Lakes region of the United States.

We focus on two study areas (Fig. 9.4): 1) the Wild Rivers Legacy Forest (WRLF) area in northeastern Wisconsin encompasses 26,300 ha and contains both state-owned and managed forests as well as lands that are owned and managed by Timber Investment Management Organizations (TIMOs) with state-held WFCEs; 2) the Two Hearted River (THR) Watershed in Michigan's Upper Peninsula encompasses 46,538 ha and contains a mix of TNC and state-owned land under WFCEs and TNC-owned land that will be managed under Forest Stewardship Council certification (Forest Stewardship Council 2009). These two areas are similar in forest and landscape composition (riparian systems and hemlock-hardwood forest types predominate) and are typical of the adjacent Great Lakes and Superior Mixed Forest ecoregions. These two sites are regionally important for conservation due to the variety of biodiversity targets addressed and the landscape scale effort to abate the threat of subdivision as large landowners divest. Other examples of similar WFCEs

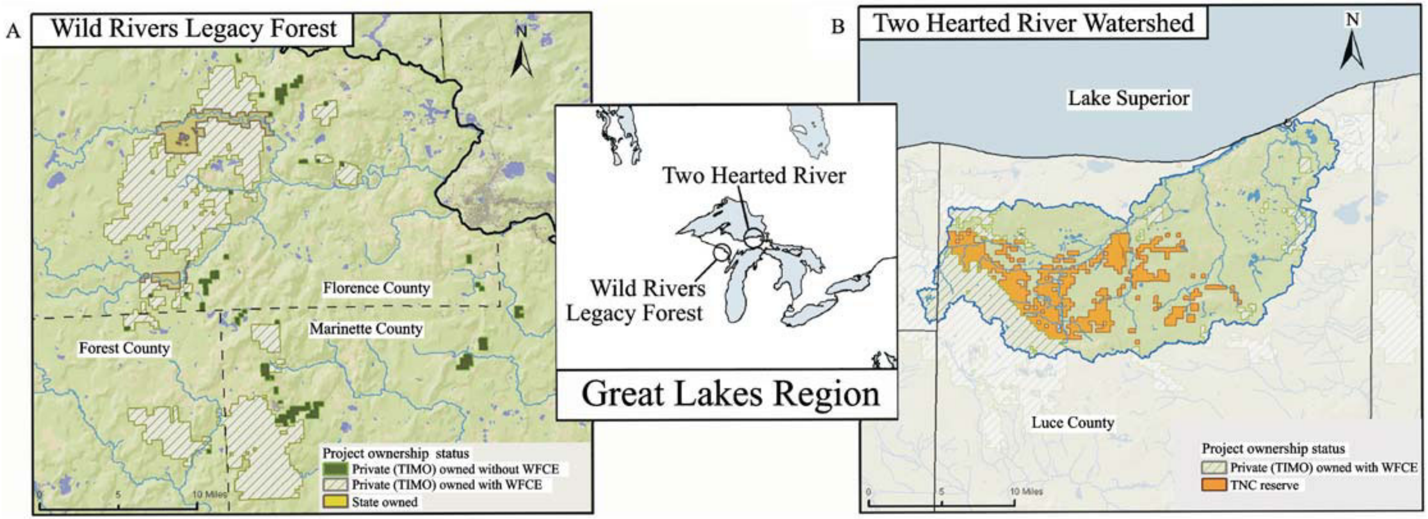

Fig. 9.4 Maps showing the Wild Rivers Legacy Forest in northeastern Wisconsin (A) and the Two Hearted River Watershed in Michigan's Upper Peninsula (B). Maps courtesy of John Wagner, The Nature Conservancy in Wisconsin.

occur in Maine with the Pingree Forest Easement implemented in 1999 by the New England Forestry Foundation (NEFF 2009) and in Minnesota with the Koochiching WFCE implemented in 2007 (TNC 2007). These sites exemplify the innovative landscape scale forest conservation strategies at work today, with many organizations and stakeholders at work on the landscape.

The scenario-building process we use (Fig. 9.5) is distilled into five general, iterative stages: 1) information gathering and scenario development, 2) target selection, 3) determining model parameters, 4) spatially explicit landscape modeling, and 5) synthesis of spatial narratives. Each stage is informed by our core team, consisting of conservation professionals and landscape ecologists, as well as local and regional experts via four interactive in-person and web-based workshops (dark grey boxes, Fig. 9.5). We have considered these partners into two groups: an Expert Group that has site- or subject-specific expertise and

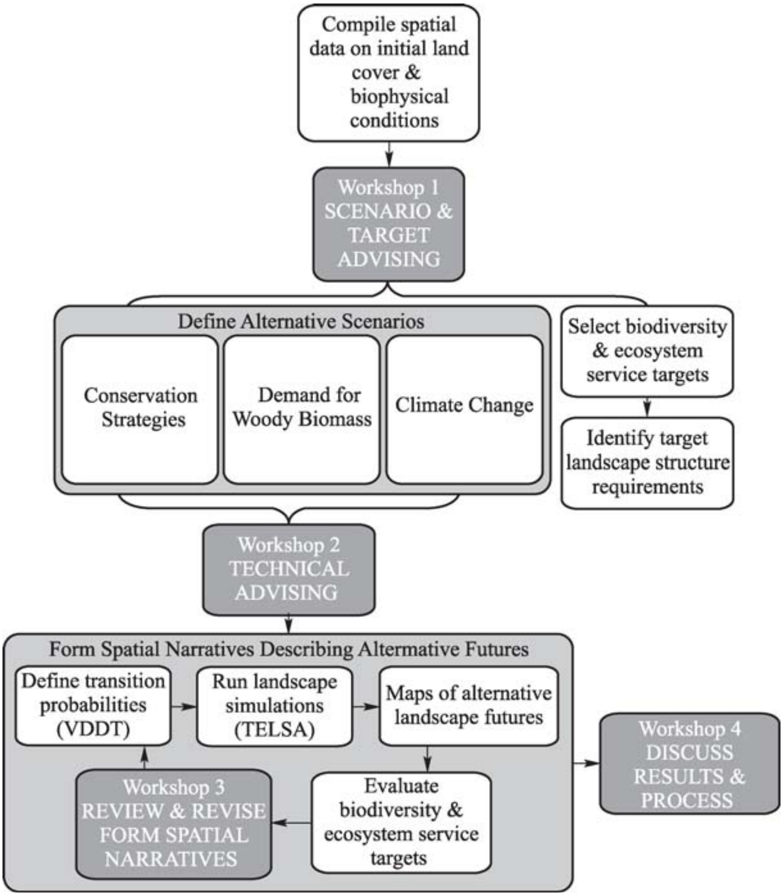

Fig. 9.5 Flow chart of the scenario-building process, infused with local and regional expert knowledge during four workshops (dark grey boxes).

participates in Workshops 1, 3, 4; and a Steering Group with regional expertise to ensure alignment with TNC goals and to consider our project within the broader forest management and monitoring context, whose role is focused on Workshops 2-4.

9.2.1 Information gathering and scenario development

The first stage focuses on developing the scenarios or different possible conditions that may drive landscape change in our study areas. These are exploratory, rather than normative, scenarios. Scenario development requires an understanding of the initial state of each study area as well as the dynamic biotic and abiotic processes affecting these areas. First, initial maps of the two study areas are constructed by using land cover data and setting biophysical conditions. Initial landscape structure (composition and configuration) of the study areas is quantified by using spatial landscape metrics and indices. These initial landscape maps and indices provide the baseline from which alternative future landscapes diverge during the modeling process.

Once the baseline status of the study areas is established, the next step is to define the landscape scenarios for which we will model possible future landcover. Each scenario is composed of a set of conditions that influence landscape change. Here, each scenario is a combination of a conservation strategy, a level of demand for woody biomass for energy production, and selected climate change variables (Fig. 9.5). The Expert Group provides crucial input for defining these scenarios in Workshop 1, including details about the alternative conservation strategies and demand for woody biomass that might be applied in each of our study areas.

Climate change projections are also a key component of each scenario. Specifically, we use data on climate change variables and rates for Great Lakes terrestrial ecosystems projected with Climate Wizard software developed by TNC, the University of Washington, and the University of Southern Mississippi (TNC 2009) and informed further by work of the Wisconsin Initiative on Climate Change Impacts (WICCI) Forestry Working Group (pers. comm., Sep. 2009). We then migrate selected climate output variables (e.g. change in temperature, precipitation rates) at defined time steps into model definition as described next.

9.2.2 Target selection

Input from the Expert Group is also integral to selection of biodiversity and ecosystem service targets for each study area, the other component of Workshop 1 (Fig. 9.5). Because the possible conservation outcomes for both biodiversity and ecosystem service targets are evaluated based on maps of possible land cover for each alternative future, all targets must have specific landscape structure or forest composition requirements. For example, biodiversity targets common to both areas include pine marten, red-shouldered hawk, and a suite of rare understory plants, including trillium, bunchberry, dogwood, and fringed-polygala, as well as communities such as Great Lakes Beachgrass Dune, Bog Birch-Leatherleaf Poor Fen, Jack Pine - Red Pine Barrens, Great Lakes White Pine - Hemlock Forest (TNC 2000), and fishless lakes. For each of those targets, we draw from known occurrences, existing studies, and expert knowledge about habitat and landscape structure requirements, especially in terms of spatial pattern and forest composition. We also relate the targets to indicators of forest health that TNC maintains. Then current and projected future habitat under different scenarios can be mapped based on measured landscape and forest health indices.

Ecosystem service targets for this area fall primarily in the provisioning (e.g. forest products – timber, game, jobs) and cultural services (e.g. recreation, bird-watching) categories (Diaz et al. 2005). For example, trout fishing is an ecosystem service important in these areas. As with biodiversity targets, landscape structure and forest composition requirements will be determined for each of the selected ecosystem services, and measured landscape cover in each of the different scenarios will be used to estimate their ability to provide the selected ecosystem services.

9.2.3 Determining model parameters

The next step is to determine the parameters for the landscape model for each study area with the input of both the Expert and Steering Groups in Workshop 2. Model parameters, including ecological pathways of disturbance and succession and how these pathways will be influenced by projected climate variables and demand for woody biomass, must be defined and incorporated into the model interface. Though these parameters are grounded in the principles of forest and landscape ecology, expert input and local knowledge about the dynamics of our study areas refine the landscape modeling process.

9.2.4 Spatially explicit landscape modeling

We are using spatially explicit landscape modeling to simulate forested landscape configurations for each combination of conservation strategy, climate change projection, and demand for woody biomass. Our primary modeling tool is the VDDT/TELSA suite developed by ESSA technologies, which has been grouped with models that include spatial interactions among static communities but exclude ecosystem processes (Scheller et al. 2007b). The Vegetation Dynamics Development Tool (VDDT) has been used extensively by the LANDFIRE program and other projects with TNC involvement. This low-cost and relatively user-friendly tool provides a state and transition landscape modeling framework for examining the role of various disturbance agents and management actions in vegetation change. We are using VDDT to build transition diagrams with succession, management, and disturbance pathways and transition probabilities. These transition diagrams are further informed by data on climate change and woody biomass demand gathered in Workshop 1 as well as by expert input in Workshop 2 (Fig. 9.5). Once the diagrams are built for particular ecological systems and management strategies, the model is run to obtain expected proportions of the landscape that will be in specific successional classes (states).

To generate spatially explicit landscape maps, the state and transition models developed with VDDT are linked to the Tool for Exploratory Landscape Scenario Analyses (TELSA). TELSA projects multiple states for multiple ecological systems across the landscape to produce spatial data. TELSA is polygon-based, requiring that specific geographic areas be assigned to an ecological system and an age class. VDDT is the foundation for the spatial modeling in TELSA, and thus its non-spatial models serve as major inputs to guide the spatial modeling.

For each alternative conservation strategy, management regimes are assigned by area and parameters, based on input from the Steering Group. Then, the TELSA main model is used to simulate land cover changes at 25-, 50-, 100- and 200-year time steps under each of the four conservation strategies and with various degrees of climate change and demand for woody biomass. The results from the TELSA modeling yield simulated landscape maps for each time step under each combination of conservation strategy, climate change, and demand for woody biomass, for a total of 24-32 initial simulations (more with additional iterations). Using TELSA we can evaluate some of the landscape requirements determined for each selected biodiversity and ecosystem service target. For additional metric analysis, raster output maps from these modeling runs can be used as input layers in FRAGSTATS. Map and graphic output from TELSA and FRAGSTATS allow us to compare and communicate the potential outcomes and landscape indices resulting from different scenarios.

9.2.5 Synthesis of spatial narratives

Participants at Workshop 3 review and consider the series of landscape simulation outputs. Using their combined knowledge of the systems, they identify which scenarios are plausible and build spatial narratives, or storylines, around those alternative landscapes to describe human-ecological dynamics behind the visible landscape change. Input from this workshop also guides us in modifying the model and running additional iterations to produce more plausible simulations.

Finally, these scenarios are disseminated to TNC's forest conservation leaders in Workshop 4, a conference-style workshop at a central location within the upper Great Lakes region, to review lessons learned about various protection strategies. We invite an open discussion of the spatial narratives that emerged from the study and evaluation of the maps and graphics that convey how the two landscapes might look and function in the future. As a group, we reflect on implications of these scenarios considering, for example, whether TNC made the right decisions with these conservation strategies.

9.3 Conclusions and implications: Pushing the frontier

Given the context of global change, innovative forest conservation strategies will be critical to future ecosystem health and biodiversity as well as the quality of life provided by ecosystem services. However, the success of these strategies depends on their ability to address very challenging issues: making decisions with incomplete information, working across multiple political boundaries, limited resources, and varied vulnerabilities and needs of conservation targets. While there will never be a perfect "toolset" to address all of these issues for each stakeholder, we suggest that by creative use of new and existing approaches we can advance conservation.

Here, we have presented scenario-building as a flexible tool for informing and optimizing landscape scale forest conservation efforts. This integration of scenario analysis and landscape modeling enables scientists and conservation practitioners to understand the potential outcomes of the complex and simultaneous interactions of the diverse milieu of processes that influence landscape change over time, including ecological processes, climate change, and interactions of humans and the environment. We have demonstrated how the scenario-building approach can be used with teams of local and regional experts to explore and model and understand these complex dynamics in forested ecosystems in North America, and we expect that this approach can be tailored to provide insight into other conservation settings and drivers of landscape change. For example, this scenario-building approach (Fig. 9.5) could provide insight into the possible futures of grasslands given various climate change and grazing pressures, or it could be used to understand the possible response of salt marshes to rising sea levels and development pressures.

Scenario-building complements both monitoring and adaptive management of ongoing conservation efforts. Areas revealed as vulnerable under a particular conservation strategy may warrant more intensive monitoring. And, by suggesting how different parts of the landscape could plausibly respond under various scenarios, adaptive management can be considered to redirect landscape change. Target ecosystems that respond poorly under changing climate scenarios might be candidates for a modified conservation strategy. Additionally, while the scenario-building process suggests plausible landscape outcomes, we expect that it will also lead to enhanced shared conservation management. Involving local experts and managers in defining the models and visioning futures will likely lead to more realistic outcomes (as opposed to black box models) and increased cooperation in conservation strategies (Gustafson et al. 2006).

Scenario-building also facilitates conservation planning. By comparing the potential outcomes and conservation effectiveness of different conservation strategies in an area of interest, conservation practitioners can make informed decisions about how to best utilize scarce financial resources and reduce the risks associated with the implementation of innovative strategies. In other words, this approach can be used to determine when and where concentrated versus distributed conservation may be most effective. These outcomes can inform the processes of negotiating easement acquisitions, arranging conservation strategies on the landscape, and maximizing return on conservation investments.

If successful, scenario building projects should result in decisions that respond better to a changing environment and socioeconomic conditions. Only through long-term monitoring and landscape scale experiments can this metric be truly assessed. However, it is clear from our past experiences, and from literature (Mahmoud et al. 2009) that scenario-building promotes discussion and a more thorough consideration of potential complications and benefits of innovative landscape scale conservation strategies. In addition, we have learned that often the best way to communicate is to consider how various strategies may affect local ecosystems. The perspectives gained from scenario-building are often provocative, leading to engaging discussions and a better understanding of the system(s) of interest. It is clear that only through cooperation and constructive communication can conservation be successful at broad scales. Scenario-building provides a framework for both.

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Chapter 10 Forest Avian Species Richness Distribution and Management Guidelines under Global Change in Mediterranean Landscapes

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Abstract

Determining forest bird responses to environmental factors may represent a keystone to disentangle how forest management could mitigate the current and expected impacts of global change in Mediterranean biodiversity. We analyzed the spatial variation of the relationships between bird species richness (specialist and generalist birds) and forest landscape features, fires and climate in order to provide specific forest management guidelines in the Mediterranean region of Catalonia (NE Spain). We performed Geographically Weighted Regression (GWR) models, an extension of the standard regression approach that accounts for non-stationary processes in the analyzed relationships. Climate warming would negatively affect forest bird diversity, particularly in the southern part of Catalonia where the higher temperatures and lower precipitations occur. However, the key role of forest landscape characteristics to explain the distribution of bird species richness suggests that forest management could buffer the negative impacts of climate change. Management should also avoid landscape homogenization and an excessive fuel accumulation that can boost the increasing wildfire occurrence, which has been here shown to negatively impact forest bird species richness in the region.

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Keywords
Catalan Breeding Bird Atlas, Catalonia (NE Spain), climate warming, fires, forest canopy cover (FCC), forest cover, Geographically Weighted Regression (GWR), Mediterranean forest landscapes, non-stationary processes, specialist and generalist birds.

10.1 Introduction

In the current context of global change, we assessed forest bird responses to environmental factors from Mediterranean forest ecosystems, particularly from the region of Catalonia (NE Spain). The large spatial extent covered made us consider non-stationary processes by means of Geographically Weighted Regressions (GWR).

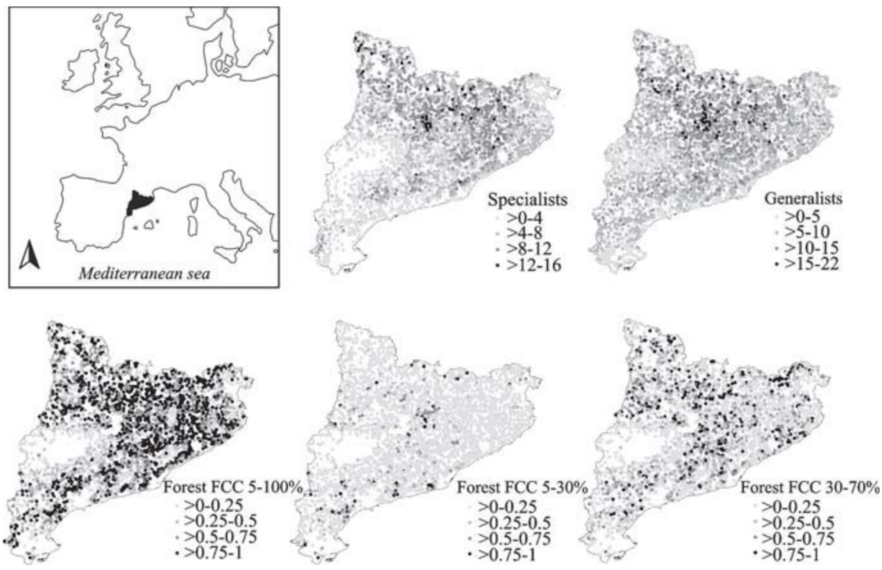
10.1.1 Global change and Mediterranean forest ecosystems

Mediterranean forest landscapes have been altered by a long-lasting anthropogenic pressure and fire events (Blondel and Aronson 1999), and hardly any virgin forest remains in the region. In the last decades there has been a progressive reduction in traditional forestry activities due to rural depopulation and the associated change of human dependence on forest products (Fabbio et al. 2003). Furthermore, rural land abandonment has also boosted afforestation in former agricultural areas (Debussche et al. 1999; Poyatos et al. 2003), which is counteracting with a greater incidence of wildfires due to an increase in forest fire occurrence and extent in large regions of the Mediterranean in the last years of the 20th century (Mouillot and Field 2005), mainly associated with landscape homogenization, fuel accumulation, socioeconomic interests and climate warming (Pausas 2004).

In this context, current and expected impacts of climate change in the Mediterranean have been anticipated to be large due to warmer conditions, a potential expansion of desert habitats, increased risk of forest fires and drought events (Metzger et al. 2008). Specific types of changes for Mediterranean environments have been predicted depending on their location; for example southern Mediterranean strata are projected to expand northwards while Mediterranean mountain environments will decline dramatically (Metzger et al. 2008). In fact, Mediterranean forests appear to be one of the most threatened habitats because of climate warming and other large-scale disturbances (De Dios et al. 2007). Forest management can have a key role to mitigate the effects of climate change by means of carbon sequestration and by improving habitat spatial cohesion and other favorable landscape features (De Dios et al. 2007). Birds are often considered a good forest biodiversity indicator (Sekercioglu 2006), and may be particularly suited to filling knowledge gaps about the influence of forest management on forest biodiversity. As in other regions of the world (Mitchell et al. 2001; Westphal et al. 2003; Radford et al. 2005; Caprio et al. 2009), forest birds in the Mediterranean have been shown to respond to forest features at the landscape level (Gil-Tena et al. 2007, 2008). In this respect, the increasing availability of biodiversity data at large regional extents can be useful for evaluating the relative importance of the landscape factors influencing forest biodiversity, and providing valuable recommendations for forest and landscape managers in a cost-effective way (Gil-Tena et al. 2008).

10.1.2 Catalonia: A Mediterranean heterogeneous region

Catalonia (NE Spain) (Fig. 10.1) is a heterogeneous region comprising a wide range of habitats from mountainous areas in the Pyrenees and inland chains (up to 3,143 m) to a long coastline along the Mediterranean Sea. The climate is mainly Mediterranean temperate, with a maritime influence in the coast and a cold influence in the Pyrenees. Forests represent about 38% of Catalonia, with an increasing trend (Gil-Tena et al. 2009) because of the afforestation of former agricultural lands (Poyatos et al. 2003). Nevertheless, between 1975 and 1998 fires have burned approximately 240,000 ha (Díaz-Delgado et al. 2004). About a hundred of different tree species can be found, although 90% of the total number of trees is from the 14 most common tree species (mainly



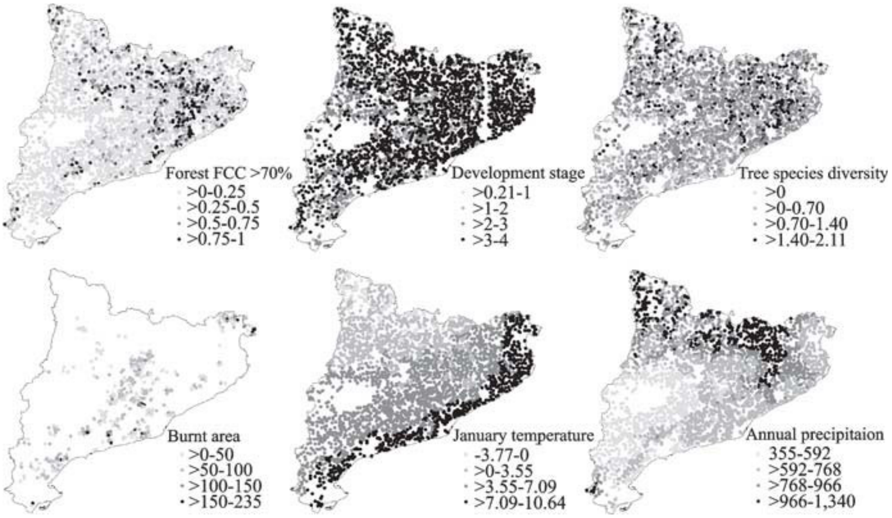


Fig. 10.1 Geographic location of the study area (Catalonia, shown in black color in the upper left chart) and representation of the dependent and independent variables. See M*aterial and Methods*' section for the variables' descriptions and units.

from the *Pinus* and *Quercus* genera), with the average stand age being under 50 years for most of the forest types (Gracia et al. 2004).

10.1.3 Geographically weighted regression: A spatial regression technique to model non-stationary processes

Stationary processes (those that are constant over space) rarely prevail in real landscapes (Wagner and Fortin 2005), particularly when considering large spatial extents (Fortin and Dale 2005). Therefore, non-stationarity should be included in models characterizing ecological processes in order to avoid incorrect inferences that may not apply to the whole study region (Wagner and Fortin 2005). A range of approaches may be used to deal with non-stationary processes in studies of wildlife distribution (Osborne et al. 2007). Among them, Geographically Weighted Regression (GWR) has been suggested to be particularly useful to complement global regression modeling (Osborne et al. 2007) and widely used to model ecological processes, such as determinants of bird distributions (Foody 2004, 2005; Osborne et al. 2007) and forest ecosystem relationships (Zhang et al. 2004, 2005; Guo et al. 2008).

GWR is an extension of the standard regression framework that allows the modeling of processes that vary over space, resulting in a set of local regression models that enable to assess the local performance of predictors (Fotheringham et al. 2002). The distance (search radius or bandwidth) within which surrounding observations are included in the analysis can be defined according to a fixed Gaussian kernel, or by using an adaptive kernel that alters the inclusion distance to encompass a defined number of data points. The observations' influence is weighted with a distance-decay-function from the location being predicted (see Fotheringham et al. 2002 for an extensive mathematical overview). Besides, as GWR generates a regression coefficient for each variable at each data point, it is possible to assess the stationarity of the analyzed processes and relationships (Fotheringham et al. 2002).

10.1.4 Study aim

In order to provide specific forest management guidelines within the study region to better face up to the current context of global change, we analyzed the spatial variability of the relationships between forest bird species richness (differentiating between specialist and generalist birds) and forest landscape features, forest fires and climate (at 1 km $\times$ 1 km) in the Mediterranean region of Catalonia using GWR models (Fotheringham et al. 2002). Then, we compared these GWR models to those developed by Gil-Tena et al. (2007) in Catalonia. We here improved the analyses by Gil-Tena et al. (2007) by means of the non-stationary approach and considering forest fires and climate variables. As we do not pretend to compute a predictive model, but to better understand species responses to habitat characteristics in our region, for each bird specialization group we computed three different GWR models, with each one assessing separately the influence of three aspects that can mediate forest birds' diversity in the current and future context of global change. We studied the influence of forest management on forest bird species richness by means of forest landscape factors, of fires by means of the accumulated burnt area during 1980-2000 and of climate by means of precipitation and temperature variables. In this respect, it is worth noting that at finer scales vegetation is likely to have an effect on breeding bird distribution closer to causality than climate by providing breeding substrates and foraging habitats (Seoane et al. 2004), whereas climate would be a major determinant of species' distributions at broad biogeographical scales (González-Taboada et al. 2007). Therefore, at the scale of our study we assume that climate may be acting more as a surrogate for one or more factors that relate to space and co-vary with climate and are thought to directly influence species richness. Furthermore, we did not model together the influence of forest landscape, fires and climate on forest birds since the relationships established between them and species richness cannot be constant in time and we want to provide straightforward forest management guidelines.

10.2 Material and methods

The procedure for obtaining the data on forest birds and environment (forest landscape, fire and climate variables) as well as the analysis performed are listed below.

10.2.1 Forest bird data

The Catalan Breeding Bird Atlas (Estrada et al. 2004) includes information about the distribution of breeding birds in Catalonia during the period of 1999-2002. We estimated the forest bird species richness from the census bird data collected by volunteers within the Atlas work in a sample of 3,038 1 km × 1 km UTM cells throughout Catalonia. Two 1-hour surveys (between sunrise and 11 am, and between 6 pm and sunset) in the period of March-July were conducted in each 1 km × 1 km UTM cell within that atlas.

We selected 53 forest breeding bird species recorded in the 1 km × 1 km UTM cells (Table 10.1) that were classified either as specialists (22 species) or as generalists (31 species) according to differences in the species forest and agricultural habitat selectivity indices (Gil-Tena et al. 2007, 2008) derived from the bird atlas data (Estrada et al. 2004). Forest specialists were characterized by higher selectivity of forested landscapes and avoidance of agricultural dominated landscapes, whereas generalist species, despite showing positive selection of forested landscapes, did not clearly avoid agricultural landscapes (Table 10.1).

Table 10.1 Forest breeding bird species selected for the analysis.

SPECIALISTS	*Accipiter gentilis, Accipiter nisus, Aegithalos caudatus, Certhia familiaris, Coccothraustes coccothraustes, Dendrocopos major, Dendrocopos minor, Dryocopus martius, Erithacus rubecula, Fringilla coelebs, Garrulus glandarius, Loxia curvirostra, Parus ater, Parus caeruleus, Parus palustris, Phylloscopus collybita, Regulus ignicapilla, Regulus regulus, Sitta europaea, Sylvia atricapilla, Tetrao urogallus, Turdus philomelos*
GENERALISTS	*Anthus trivialis, Buteo buteo, Carduelis spinus, Certhia brachydactyla, Circaetus gallicus, Columba palumbus, Corvus corax, Corvus corone, Cuculus canorus, Emberiza cia, Emberiza citrinella, Falco subbuteo, Ficedula hypoleuca, Hieraaetus pennatus, Lullula arborea, Milvus milvus, Oriolus oriolus, Parus cristatus, Parus major, Pernis apivorus, Phylloscopus bonelli, Picus viridis, Prunella modularis, Pyrrhula pyrrhula, Serinus citrinella, Sylvia borin, Sylvia cantillans, Troglodytes troglodytes, Turdus merula, Turdus torquatus, Turdus viscivorus*

10.2.2 Environmental data

Forest landscape and climate variables were computed for 2,497 1 km × 1 km UTM cells in order to compare the models assessing the relationships of forest landscape and climate and bird species richness. Forest landscape characteristics were obtained from the Spanish Forest Map (SFM) at the scale of 1:50,000 (created within the Third Spanish National Forest Inventory; Ministerio de Medio Ambiente 2006). According to the SFM, we considered those UTM cells with presence of forest [defined as the land with a forest canopy cover (FCC) above 5%] that were completely inside Catalonia, and for which the SFM data were entirely available and updated (excluding areas affected by the large wildfires during 1998). We selected those forest landscape variables that were shown to be more biologically meaningful to bird species richness in previous studies (Gil-Tena et al. 2007, 2008), and that at the same time expressed sufficiently distinct aspects of the forest landscape (with $r < |0.5|$):

- Area covered by forests with three different ranges of FCC [FCC from 5 to 30% (Forest FCC 5-30%), from 30 to 70% (Forest FCC 30-70%), and >70% (Forest FCC>70%)], expressed as the proportion of the total cell area.
- Mean forest development stage (Development stage), computed as the area-weighted average for each forest patch in the 1 km × 1 km UTM cell. We assigned a numerical value for the four different development stages discriminated in the SFM, that is, from recently regenerated to canopy closure (1), from thicket to natural pruning (2), trees with diameter at breast height (DBH)⩽20 cm (3) and trees with DBH>20 cm (4).
- Forest tree species diversity (Tree species diversity), quantified through the Shannon-Wiener index for the proportion of forest land area covered by each tree species.

The Climatic Atlas of the Iberian Peninsula (Ninyerola et al. 2005) was the data source for the mean annual values of temperature and precipitation, the mean temperature during the coldest and hottest months (January and July, respectively) and the summer precipitation. The best climate variables were selected by means of a principal component analysis (PCA) with a varimax normalized rotation, which maximizes the correspondence between the factors and the original variables. The selected descriptors in the PCA were the mean temperature of January (January temperature) and the mean annual precipitation (Annual precipitation).

Forest fires were assessed from the Catalan government data (Departament de Medi Ambient i Habitatge 2007) and were only studied in 509 1 km × 1 km UTM cells that had been affected by fires. We estimated the accumulated burnt area (in hectares; Burnt area) in each 1 km × 1 km UTM cell in the 20-year period before the Atlas (1980-2000).

All the variables were standardized to zero means and unit variances to eliminate the effect of differences in the measurement scale.

10.2.3 Analysis

To detect the spatial variation of the relationships between bird species richness and forest landscape features, forest fires and climate in Catalonia, GWR calculations were performed using the GWR 3.0 software (Charlton et al. 2003), applying both standard ordinary least squares and Geographically Weighted regressions. We incorporated species spatial ecology by using a bandwidth that matches species richness spatial range (Guo et al. 2008) based on Moran's I zone of influence (Fortin and Dale 2005). The lag distance defined to calculate Moran's I coefficients of spatial autocorrelation was 10 km, which is the minimum distance at which it is possible to detect a true spatial pattern from the Atlas sampling design (for more details see Estrada et al. 2004). As we had four different datasets of species richness according to their specialization group (one dataset for specialists and the other for generalists) and the model to be performed [one dataset for forest landscape and climate (n=2,497) and the other for fires (n=509)], we computed a Moran's I correlogram for each one. For each variable we took the distance at which the value of spatial autocorrelation crosses the expected value $\mathrm{E}(I)$ for the absence of spatial autocorrelation, indicating the spatial range of the pattern (Fortin and Dale 2005). The resultant spatial range (bandwidth) determined for the species richness subsets to be modeled by forest landscape and climate factors was of 90 km for specialists and of 70 km for generalists, whereas for forest fire subsets was of 70 km for specialists and of 60 km for generalists.

To compare if GWR improved the standard regression techniques (hereafter called Standard) we used the Akaike's Information Criterion (AIC) since R^2 is not a meaningful metric for comparing GWR and Standard regression models because a model with many parameters will have a very good fit to the data but also few degrees of freedom. The best model will be the one with the smallest AIC. We also used an approximate likelihood ratio test, based on the F-test, to compare the ability of the models to replicate the observed data (Fotheringham et al. 2002). This test is based on a comparison of the residual sum of squares of the two regression models and tests the null hypothesis that the local model (GWR) represents no improvement over the global model (Standard). In order to assess the parameter significance, we considered a more restrictive p-value ($p < 0.01$ instead of $p < 0.05$) to avoid potential problems caused by multiple hypothesis tests (for more details see Fotheringham et al. 2002). A Monte Carlo significance testing procedure was selected to examine the significance of the spatial variability in the local parameter estimates (Fotheringham et al. 2002). Furthermore, mapping the local estimates of R^2 may allow one to assess how stationary may be the processes and to know the capacity of replication of the local model in the vicinity (Fotheringham et al.

2002).

Detectable levels of spatial autocorrelation in model residuals would indicate that some part of the spatial pattern of the dependent variable was not considered by the model and, therefore, the model estimates are probably unreliable (Cliff and Ord 1981). The spatial autocorrelation of model residuals was investigated using global and local Moran's I coefficients. Local Indicators of Spatial Association (LISA) (Anselin 1995) quantify locally the spatial dependence and can identify local pockets of spatial autocorrelation. LISA of Moran's I coefficients were computed since a global assessment of spatial dependence may be misleading if the assumption of stationarity is violated (Fortin and Dale 2005). The bandwidth defined to compute the GWR models was the lag distance used to calculate the global and local Moran's I coefficients for the residuals. The Rookcase excel add-in was used for exploring global and local spatial autocorrelation (Sawada 1999).

10.3 Results and discussion

In all the cases, GWR models significantly improved the Standard regression results, producing better-fitting models with lower AIC values (Table 10.2). These results agreed with former studies that have applied this spatial regression technique (Foody 2004, 2005; Zhang et al. 2004, 2005; Osborne et al. 2007). This implies that some of the relationships between forest bird species richness and the factors considered may not be constant across Catalonia, and that the model parameters may vary from sector to sector within the study region. The explanatory power of the GWR models varied spatially in terms of local estimates of R^2 (Fig. 10.2). Particularly, for the models assessing the role of forest landscape features and climate, there was a general trend of

Table 10.2 Comparison between model-fitting of Standard and GWR models assessing the factors behind forest bird species richness according to their specialization group. RSS: Residual Sum of Squares, ENP: Effective Number of Parameters, AIC: Akaike's Information Criterion. According to the F-test, in all the cases the GWR models significantly improved the Standard models (p <0.0001).

		Standard Specialists	GWR Specialists	Standard Generalists	GWR Generalists
Forest landscape model	RSS	18,533	16,889	23,344	21,803
	ENP	6	10.77	6	14.14
	AIC	12,105	11,883	12,682	12,528
Climate model	RSS	21,863	20,326	24,150	22,636
	ENP	3	4.77	3	6.03
	AIC	12,512	12,334	12,760	12,605
Fire model	RSS	4,475	3,865	5,360	4,430
	ENP	2	4.45	2	5.27
	AIC	2,557	2,487	2,649	2,559

increasing R^2 southwards, quite independent of the specialization group (Fig. 10.2). Thus, the adjusted local models appeared to reduce their explanatory power in areas with higher forest species richness (northern part of Catalonia) and with a more heterogeneous pattern (Fig. 10.1). Nevertheless, we found that some of the significant relationships regarding species richness and forest landscape features were stronger towards the North (development stage in

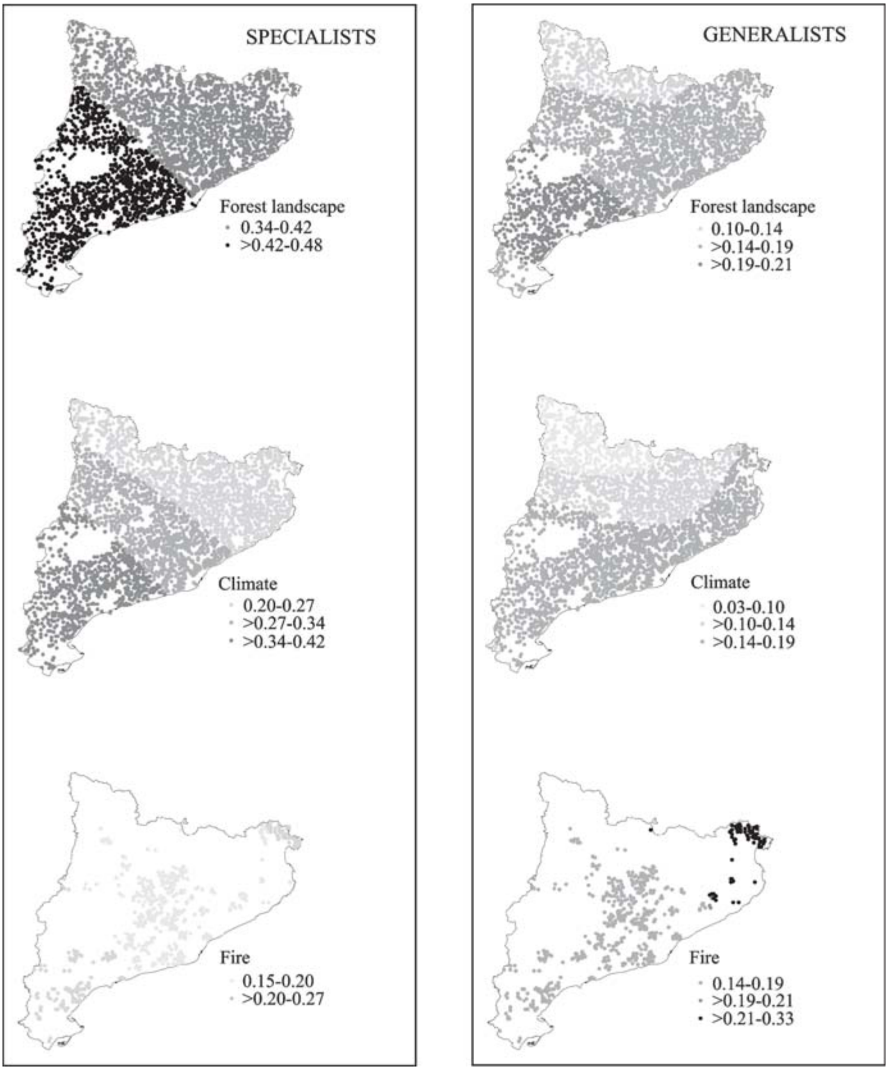


Fig. 10.2 Local R^2 representation of the GWR models assessing the factors explaining the distribution of forest bird species richness according to their specialization group. The sample size depends on the models, being n=2,497 for the forest landscape and climate models and n=509 for the fire model.

general and forest area with FCC 30-70% for specialists) (Fig. 10.3).

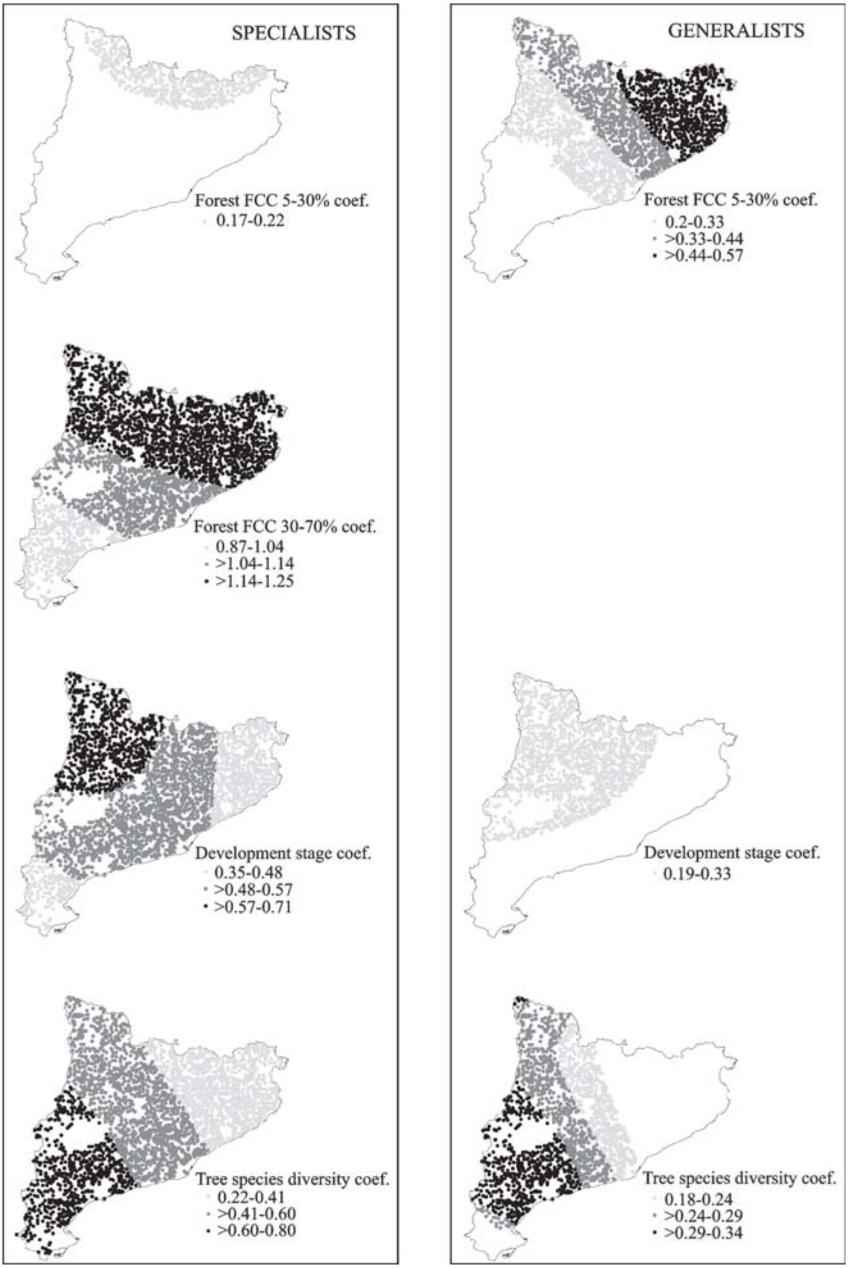


Fig. 10.3 Spatial variation of the significant (p <0.01) parameters of the GWR models assessing the forest landscape factors explaining the distribution of forest bird species richness according to their specialization group (n=2,497). coef: regression coefficient.

GWR models based only on forest landscape variables had better fits (lower AIC values) than those regarding climate (Table 10.2). Yet the difference was less for generalists (Table 10.2) which may indicate that specialists are comparatively more affected by forest features (Mitchell et al. 2001; Gil-Tena et al. 2007, 2008; Caprio et al. 2009) because of the higher preference of specialists for forests rather than for non-forested habitats.

When focusing on the GWR models assessing the forest landscape factors behind forest bird species richness, we found that the results were quite in accordance with those by Gil-Tena et al. (2007) in terms of variable importance. In addition, GWR approximations represent an improvement with regard to the results by Gil-Tena et al. (2007) because there were some parameters that were not stationary across Catalonia (Table 10.3), particularly for specialists. For both specialist and generalist birds a forest cover with different ranges of FCC was shown to be their main habitat requirement, although forest specialists appeared to be more determined by more closed canopies (FCC>70%) than generalists. It is worth noting that independently of the specialization

Table 10.3 GWR models assessing the factors explaining the distribution of forest bird species richness according to their specialization group.

SPECIALISTS		Min. β	Mean β	Max. β	$t-$values	Spatial variability
Forest landscape model	Intercept	4.77	5.83	6.36	***	***
	Forest FCC: 5-30%	-0.05	0.12	0.22	depending	*
	Forest FCC: 30-70%	0.87	1.13	1.25	***	***
	Forest FCC>70%	1.64	1.72	1.78	***	n/s
	Development stage	0.35	0.53	0.71	***	***
	Tree species diversity	0.22	0.49	0.80	**	***
Climate model	Intercept	5.27	5.93	6.23	***	***
	Mean January temperature	-1.03	-0.32	0.06	depending	***
	Mean annual precipitation	1.25	1.62	1.91	***	***
Fire model	Intercept	2.76	4.51	5.20	***	***
	Burnt area	-1.08	-0.83	-0.63	**	n/s
GENERALISTS						
Forest landscape model	Intercept	7.98	8.95	9.57	***	***
	Forest FCC: 5-30%	-0.03	0.33	0.57	depending	***
	Forest FCC: 30-70%	0.67	0.92	0.99	***	n/s
	Forest FCC>70%	0.73	0.88	0.98	***	n/s
	Development stage	0.13	0.21	0.33	depending	n/s
	Tree species diversity	0.02	0.20	0.34	depending	**
Climate model	Intercept	8.26	9.10	9.86	***	***
	Mean January temperature	-1.45	-0.96	0.41	depending	***
	Mean annual precipitation	-0.29	0.18	0.58	depending	***
Fire model	Intercept	6.94	8.62	9.93	***	***
	Burnt area	-0.62	-0.16	-0.03	depending	n/s

$*p < 0.05, **p < 0.01, ***p < 0.001$; depending: lack of complete significance ($p < 0.01$) all over Catalonia; n/s: not significant.

group, the parameter estimates of the relationship between species richness and forests with FCC>70% were stationary, and the same was true regarding generalists and forests with FCC 30-70%, while specialists were more associated with the latter in northern regions (Fig. 10.3). In general, more open forests (FCC from 5 to 30%) showed weaker relationships with bird species richness, particularly for specialists, and were not always significant throughout Catalonia (Table 10.3 and Fig. 10.3). The large contribution of forest cover variables explaining bird species richness agreed with previous results on bird distribution at different landscape scales (Westphal et al. 2003; Radford et al. 2005; Gil-Tena et al. 2007, 2008). At the same time, forests with different proportions of FCC seem to be indispensable for forest bird species richness; this agreed with the results by Gil-Tena et al. (2007, 2008) who found that forest landscapes with only very closed canopies harbor less species. Similarly, Archaux and Bakkaus (2007) pointed out that forest birds may prefer a developed understory rather than a closed canopy. Forests with too closed canopies are usually very dense and may present lower forest bird species (Tellería and Santos 1994) because of lack of development of an understory that affects negatively the availability of feeding and foraging substrates or nest sites (Gil-Tena et al. 2007, 2008 and references therein).

Forest development stage and tree species diversity were positively but less associated with forest bird species richness, especially for generalists. The lack of stationarity between specialist species richness and forest development stage allowed us to conclude that forest development stage was more determinant for specialists in the northwestern sector of Catalonia and less in the South and Northeast (Fig. 10.3). The relationship between generalists and development stage was stationary (Table 10.3) but only significant in a northwestern sector (Fig. 10.3). In general, older forests are often associated to harbor more forest bird species (Barbaro et al. 2005; Díaz 2006; Gil-Tena et al. 2007 but see Gil-Tena et al. 2008) by providing more complex structures with more vegetation strata (Venier and Pearce 2005). Besides, most of the forest birds in the Iberian Peninsula, as in Catalonia, are from a non-Mediterranean biogeographic origin (Ramírez and Tellería 2003; Carrascal and Díaz 2003) and have been shown to be more related to more advanced development stages (Súarez-Seoane et al. 2002). Nevertheless, in the case of generalists the lack of significant association with forest development in great part of Catalonia agreed with the results by Archaux and Bakkaus (2007), who found that the vertical stratification associated with stand age has a mitigated effect on forest birds in the French Alps (depending on the study site). Tree species diversity presented a non-stationary relationship with forest bird species richness (Table 10.3) that was stronger southwards, although in the case of generalists the former trend was not significant all over Catalonia (Fig. 10.3). More diverse forest landscapes, with a high number of tree species, usually supply greater variety of potentially suitable niches for different bird species than forests with homogeneous characteristics (Hobson and Bayne 2000; Díaz 2006; Gil-Tena et al. 2007, 2008). Nonetheless, the stronger relationships in the South between forest tree species diversity and bird species richness (Fig. 10.3) appeared to indicate that forests with lower species diversity coincided with poorer bird assemblages in that sector of Catalonia (Fig. 10.1).

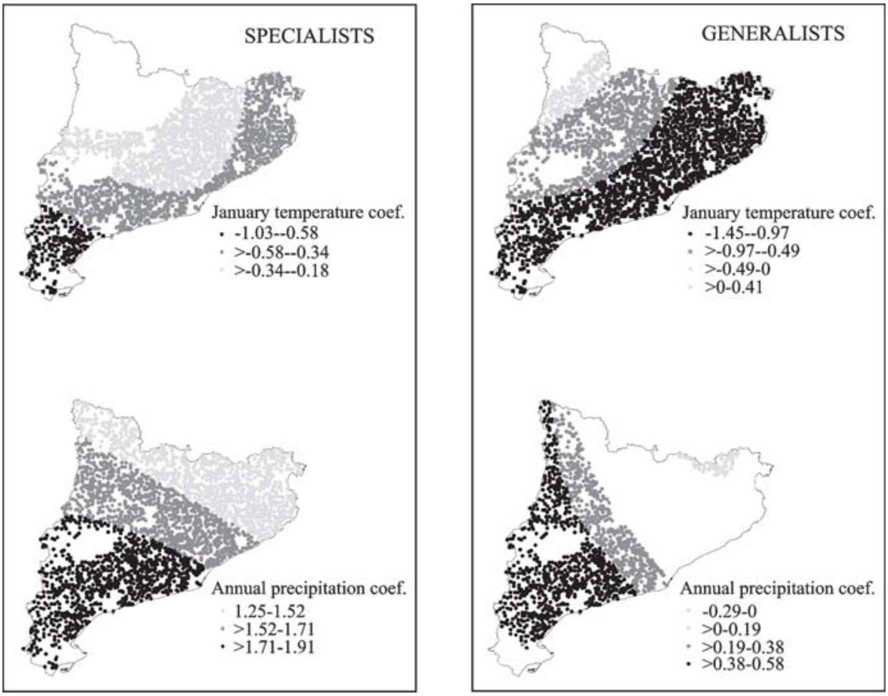


Fig. 10.4 Spatial variation of the significant (p <0.01) parameters of the GWR models assessing the climate factors explaining the distribution of forest bird species richness according to their specialization group (n=2,497). coef: regression coefficient.

As previously pointed, at the scale considered, climate variables would not have a direct effect on forest breeding bird species' distributions, but are good surrogates of factors limiting Mediterranean tree forests in the region (Thuiller et al. 2003) and other factors affecting breeding birds as environmental productivity or water stress during the breeding season (Tellería and Santos 1994; Carrascal and Díaz 2003). In this respect, the lack of stationarity in the relationships between climate and species richness (Table 10.3) may allow one to describe sectors where climate is more limiting. The mean temperature in January was significantly negatively related to species richness in almost all Catalonia, particularly in the southern part, with the highest January temperatures, and for generalists (Fig. 10.1 and 10.4). The northwestern and coldest sector presented non-significant relationships with species richness or even positive for generalists (Fig. 10.1 and 10.4). Specialists were more positively associated with annual precipitation in the southern and driest part of Catalonia, although the relationship was significantly positive throughout the study region in contrast to generalists (Fig. 10.1 and 10.4). Largely, our results agreed with those by Ramírez and Tellería (2003) who showed a decreasing trend of forest bird species richness with increasing temperature and xericity in the Mediterranean sector of the Iberian Peninsula. Nevertheless, in our case there was a stronger relationship between precipitation and bird species richness southwards (Fig. 10.4) which may indicate that in the southern xeric parts of Catalonia there are also few forest bird species (Fig. 10.1) and this had been better reflected in the GWR modeling. According to Carrascal and Díaz (2003) it seems that in Spain most of the forest bird species are more associated with the moistest and most forested northern areas than with the driest southern parts. The differences between the strength and patterns of association between specialists and generalists and the climate factors may be due to our coarse classification of the species that enclosed forest birds of different ecological requirements and biogeographic origins. Although we cannot unambiguously conclude that differences between forest bird specialization groups may be somewhat linked to the type of forests at which they are associated, Thuiller et al. (2003) suggested that Mediterranean forest tree species are distributed mainly in relation to temperature and secondly to precipitation, which seemed to correlate well with the pattern of generalists, whereas sub-Mediterranean and Eurosiberian forest tree species would appear to be influenced mainly by precipitation and secondly by temperature, thus matching the specialist pattern. The specialization of forest bird species in their environmental preferences on the Iberian Peninsula appeared to be also significantly associated with their biogeographic origin (Carrascal and Díaz 2003) and birds from diverse biogeographic origins have been shown to respond differently to climate in the Iberian Peninsula (Tellería and Santos 1994; Carrascal and Díaz 2003; Ramírez and Tellería 2003). Therefore, the different patterns observed between specialists and generalists and climate may be also associated with the fact that generalists represent a more heterogeneous biogeographic group (Holartic, Paleartic and Mediterranean species) than specialists (Holartic and Paleartic species).

GWR models including the influence of fires on species richness apparently improved the modeling of the species richness since the AIC values and the F-test indicated that GWR models performed better (Table 10.2). However, the relationships between burnt area and species richness were stationary and only the intercepts showed spatial variability (Table 10.3). Burnt area during the period between 1980 and 2000 appeared to be only significantly negatively affecting generalists on the most northeastern part of Catalonia (results not shown), while specialists were everywhere negatively associated with the amount of burnt area (Table 10.3). Fire is in the Mediterranean often associated with increases in landscape heterogeneity and enhanced opportunities for several bird species (Brotons et al. 2004), although the effects of this disturbance have been shown to be more deleterious for forest birds in a coastal woodland of Mediterranean central Italy with an apparent increase of edge/generalist species (Ukmar et al. 2007).

Spatial analysis of the GWRs residuals, by means of global Moran's I, showed the presence of significant spatial autocorrelation within the neighborhood corresponding to the bandwidths (Table 10.4), being all the Moran's I scores positive. Still, there was also a considerable amount of pockets of deviant residuals detected by LISA (most of them positive), particularly for forest landscape models when comparing to those of climate and in general for specialists (Table 10.4). Although GWR was shown to produce more desirable spatial distribution of the model residuals than standard regressions (Zhang et al. 2005), our GWR models still had a high amount of residual deviance which indicated the need to improve the analytical approach. This could be achieved by performing species or guild level analysis (but see Caprio et al. 2009), using independent variables related to biological processes (e.g. dispersal) or other factors more suitable to them (e.g. new descriptors of forest heterogeneity at the stand and landscape scale). Testing the role of other bandwidth distances may also improve our approach in terms of spatial autocorrelation (Guo et al. 2008) and help to determine at which spatial scale the relationships between the response and the predictor variables become stationary, thus allowing us to know if the ecological processes considered operate at different spatial scales (Foody 2004, 2005; Osborne et al. 2007).

Table 10.4 Global and local tests for spatial autocorrelation in the residuals of the GWR models by means of Moran's I coefficients. The table shows Moran's I for different neighborhood distances according to the bandwidth of each GWR (in brackets). In all the cases the significance of the global Moran's I was p <0.001 assessed through a randomization test with 999 runs. The central columns in the table show the percentage of the sample size with significant local spatial autocorrelation at the 0.05 and 0.01 levels. The two right-hand columns show the percentage of locations that are significant at the 0.05 level with positive (+) or negative (−) local Moran's I scores.

	Global I	p <0,05	p <0,01	(+)	(−)
SPECIALISTS					
Forest landscape model (90 km)	0.045	69%	63%	65%	35%
Climate model (90 km)	0.022	64%	57%	62%	38%
Fire model (70 km)	0.052	57%	47%	69%	31%
GENERALISTS					
Forest landscape model (70 km)	0.051	58%	50%	64%	36%
Climate model (70 km)	0.023	49%	41%	60%	40%
Fire model (60 km)	0.060	49%	40%	66%	34%

10.4 Concluding remarks and forest management guidelines

According to our results, to preserve forest bird diversity in Catalonia it would be desirable to promote forest landscapes with different proportions of FCC, with developed maturation stages (particularly in the Northwest) and with a higher tree species diversity (especially in the South). These specific recommendations represent a call for a proactive forest management in the region. In addition, due to the negative effect of fires on specialist forest bird species richness and the predicted deleterious effects of large and severe wildfires on forest biodiversity in the Mediterranean (Moreira and Russo 2007), the former concise guidelines should conciliate with the silvicultural treatments focused on fire prevention in order to create forest landscapes less prone to burn that simultaneously allow harboring Mediterranean forest bird diversity (see also Camprodon and Brotons 2006). Another key aspect to preserve forest bird diversity in managed forests should be to assess if features affecting birds are seasonal-dependent (Caprio et al. 2009).

In a context of global change, climate warming is expected to negatively impact forest bird diversity since species richness appeared to be related to low January temperatures and high precipitation, particularly in the southern part of Catalonia where the former associations were stronger. Forestry practices favorable to some forest birds will not necessarily be effective for species very sensitive to climate in comparison to forest structure and composition (Archaux and Bakkaus 2007). Nevertheless, although climate change is pointed out as a big threat for forest birds by numerous reasons (see Leech and Crick 2007 for a review), other processes related to global change such as land use changes are known to be driving recent changes in forest bird distributions in the Mediterranean (Sirami et al. 2007; Gil-Tena et al. 2009). Hence, to some extent land use changes may buffer negative impacts of climate change on birds as in the case of regrowth forests in former agricultural lands (Bowen et al. 2007). Forest management plans should consider these dynamics that can enhance forest biodiversity but also boost landscape homogeneity and other associated hazards (e.g. fires).

Considering specialization traits of specialist and generalist birds, Jiguet et al. (2007) pointed out that generalists would overcome better the effects of climate change because they are more tolerant to different environmental conditions (e.g. land uses), and fragmentation and habitat disturbance would affect more negatively specialists (Devictor et al. 2008). Despite the greater dependence shown here for specialists on forest landscapes compared to generalists, Gil-Tena et al. (2009) did not find significant differences in Catalonia between the associations of forest dynamics and the variation of forest bird species richness during the last 20 years of the 20th century according to their specialization degree. Consequently, special attention should be paid to conserving overall forest bird species diversity and forest management should be integrated with other kinds of landscape management and conservation biodiversity plans considering other aspects of forest biodiversity and planning scales.

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Chapter 11 Development of a Forest Network System to Improve the Zoning Process: A Case Study in Japan

Ken Sugimura* and Theodore E. Howard

Abstract

The Japanese forest zoning process used to be simple when society treated wood production as the primary objective. However, due to little attention paid to biodiversity as well as decreasing dependence on the domestic wood supply in the market, the government introduced the current system that requires zoning for a variety of uses and environmental services. In this procedure the site quality of a forest block is first evaluated based on natural conditions but without paying sufficient attention to social factors. In the next step they attempt to determine the primary management objective of a forest block without incorporating public views into the process. Therefore, we propose a new forest network system, using a set of social factors to revise the site quality assessment (SQA) in the first step and propose a method that integrates SQA scores and social evaluations (SE) that were obtained through public involvement in the next step. As a case study, we selected a region that is close to large cities and another in which forestry is one of the major industries. First, we examined some influences of social factors on the evaluation of SQA scores. Then, employing the SQA scores and SE for forest service categories, we determined the area to which each category was assigned as the primary management objective. As a result, incorporating social factors and public views yielded significant effects on the site quality assessment as well as providing a larger area for biodiversity in the forests near large cities.

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Keywords

Amenity, Analytic Hierarchy Process, Arnstein's ladder, biodiversity, forest planning, forestry, forest service, landscape, natural forest, primary management objective, public involvement, public opinion, public survey, site quality assessment, social evaluation, stakeholder, Thurstone's law of comparative judgment, tree plantation, water supply, wood production.

11.1 Background of the Japanese forest policy and methodological problems

As we indicate below in more detail, the current Japanese forest zoning process was fundamentally based on a system that had been developed when foresters were primarily interested in whether a forest block was suitable for wood production or not. Such a system could be dependent solely on natural sciences that assessed potential productivity. In this chapter we demonstrate how incorporating social factors can amend some drawbacks that the current system maintains.

Forests and tree plantations cover 25 million ha, which is 67% of the land area in Japan. They are generally located in mountains with ca. 2,000 to 4,000 mm of annual precipitation and are home to diverse fauna and flora. These forests extend from boreal forests that are dominated by coniferous trees in Hokkaido to the subtropical rain forests in the west of Okinawa Island (Fig.11.1). Altitudinal variation is also great, extending from timberline below the alpine tundra to lowland broad-leafed forests in the central parts of Honshu Island. On the other hand, population density is generally high in Japan. As of 2005, the population density was 343 persons per square kilometer and 1,881 persons per square kilometer excluding forested areas (calculated from Japanese Statistics Bureau 2006). Urban populations are much denser and located not very far from forests or tree plantations and these forests are intimately connected to people's daily lives by providing a variety of services, such as recreational activities or hazard protection. By contrast, rural areas have suffered from ever decreasing population due to migration to urban areas and a diminishing labor force due to aging. The variability in natural conditions and the dichotomy between urban and rural areas should have large effects on determining the relative importance of forest services.

When the Pacific War ended in 1945, economic recovery had far greater priority than the other objectives. By the same token, the Japanese forest planning process used to be simpler when society treated wood production as the primary objective. The government did not change their policies until the early 1990s, so that wood production remained the highest priority in Japanese forest planning. These policies applied to all the national forests that cover ca. 30% of the total forest area as well as most non-national forests.

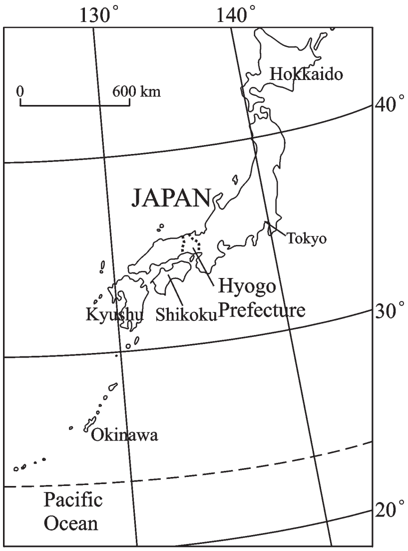

Fig. 11.1 Japanese Archipelago.

Exceptions were largely limited to special protection areas in national parks and ecosystem conservation areas in national forests. These exceptions covered less than 3% of the whole forest area as of 2000. Cutting natural forests clear for timber and pulpwood, foresters planted large areas of Sugi *Cryptomeria japonica* in the areas where the soil is mesic and Karamatsu *Larix leptolepis*, where the climate is too cold for Sugi. Hinoki *Chamaecyparis obtusa* grows slowly but has better wood quality. The government employed economic incentives, so that the percentage of plantation forests increased from 22% in 1951 to 33% in 1970 (Japan Forestry Agency 1953; Japan Ministry of Agriculture and Fishery 1993). In 1974 the government introduced the forestland development permission system to sustain forest services as well as wood production (Anon 1992). This system requires that any development activity that clears a forest larger than one hectare obtain permission from the governor of the prefecture government. All the non-national forests but for protection forests, which cover about 70% of them or 50% of the whole forest land stay under this regulation. Yet, almost all the requests have been permitted formally, while regulations for protection forests remained unchanged including national forests. As such, throughout the 1970s and 80s, foresters cleared many of the last remaining old growth forests on marginal lands, primarily in higher elevation areas. Some citizens' groups strongly protested against such activities because these stands being the last nearly untouched forests in the locality were home to some endangered species and post-harvests sites were ugly. In 1983 the government acknowledged that about 70% of the forestland in the nation was appropriate for wood production and could also satisfy other uses, such as water supply, hazard protection and recreation, simultaneously. So plantations continued to expand and covered 43% by 1990 (Japan Ministry of Agriculture and Fishery 1993). Except for limited areas in national parks, other protection forests and forests unsuitable for plantation, the Japanese forest planning and management system paid little attention to biodiversity preservation or protection of endangered species, no matter whether the forests were national or non-national.

By way of contrast, statistics measured since 1989 show that the Japanese public has regarded “wood production” as less important than land and water conservation, preservation of wildlife habitat and recreational opportunities (Japanese Cabinet Office 2003) (Fig.11.2). Sugimura (1990) also recognized an upsurge of public environmental concern in general in the late 1980s, when Time Magazine named the endangered earth “the man of the year” indicating that the planet faced serious global environmental problems. Many mass media reported the event, shocking many Japanese people, who were largely unaware of these problems. Following the UN Earth Summit in Rio de Janeiro in 1992, Forestry Agency accepted the challenge that public concern implied. Domestic wood production gradually declined, supplying only 26% of the demand in 1990, slipping down to a minimum point of 18% in 2000 and increasing slightly to 23% in 2007, while, at the same time, imports of timber and pulpwood increased (Japan Forestry Agency 1997, 2002, 2008). In 1996 when the Agency realized that the new outlook on forest preservation and lower wood consumption meant plantation areas would not be expanded, they introduced a zoning system over national and non-national forest lands (Tsumoto 1997), which we describe briefly below.

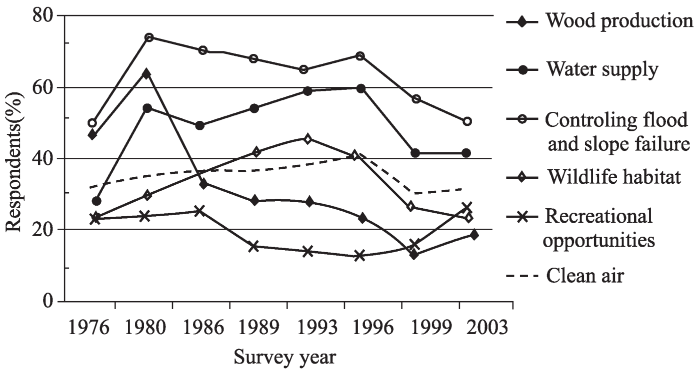

Fig. 11.2 Percentage of people who recognized a certain forest service is important in the National Statistical Survey.

In 2001, the Forestry Agency proposed a new plan to divide all of Japan’s forests and plantations, either national or non-national, into three zones: (a) wood production; (b) land and water conservation; and (c) amenity and biodiversity. The greatest difference is that in the newer system, a forest area has no more than two zones, whereas in the former system a plantation could also be water and wildlife protection areas. Now, each zone specifies the primary objectives for which any block of forest land should be managed in the following manner. (a) Tree growth should be enhanced by proper forest management such as thinning and by maintaining younger stands through an appropriate cutting cycle. (b) Forest management should emphasize soil conservation for water supply, flood control and alleviation of erosion. Development of forest floor vegetation and enhancement of tree growth are also important. It is necessary to maintain a relatively long cutting cycle and not to create an extensive clear-cut area. (c) Forest management should take care of stand structure for the conservation of biodiversity or scenic beauty. Such treatment as restoration of original vegetation, enhancement of diversified stand structure and protection of wildlife habitat corridor may be introduced. The agency instructed the local national forest offices and prefecture forestry departments to divide the forests they manage into those three zones, presenting guidelines for that procedure. One of the basic points in the guidelines was to evaluate the site quality of each block of land according to the following five services with three levels, L for low, M for medium and H for high. This evaluation was regarded as the most important information in the initial stage of the zoning system along with the existing status of protection, such as water conservation forests, wildlife protection areas, and natural parks. The five services were (a) wood production, (b1) water supply, (b2) flood control and land conservation, (c1) conservation of living environment, such as atmosphere purification, and noise reduction, and (c2) amenity and culture. For example, the evaluation of forest block "101a" may be H for (a), M for (b1), etc. The site quality level is determined primarily based on soil science, hydrology and biological sciences, so that natural conditions provide most of the criteria for judgment. It is difficult to determine the primary management objective (PMO) of a forest block simply by comparing these evaluations. The guideline suggests that (b) land and water conservation should have the greatest priority, and (c) amenity and biodiversity should be the second (Fig.11.3). It is ambiguous as to how to evaluate each block for the five services. Therefore, the local offices of the national forests and municipalities generally made subjective determinations. Then, prefectural departments reported to the National Forestry Agency how large area of forests is to be primarily managed for each service. The agency accordingly announced that 6.6, 13.0 and 5.5 million ha of plantations and natural forests should contribute to zones, a, b and c, respectively.

Although we support this change from overlapping forest-use to parallel use, we identify two potential problems with the current zoning system. One is that forest site quality is evaluated almost completely based on natural conditions without paying attention to how people use the forestland. The low level of consideration afforded social values is likely due to the educational background of foresters, which emphasizes silviculture and related natural

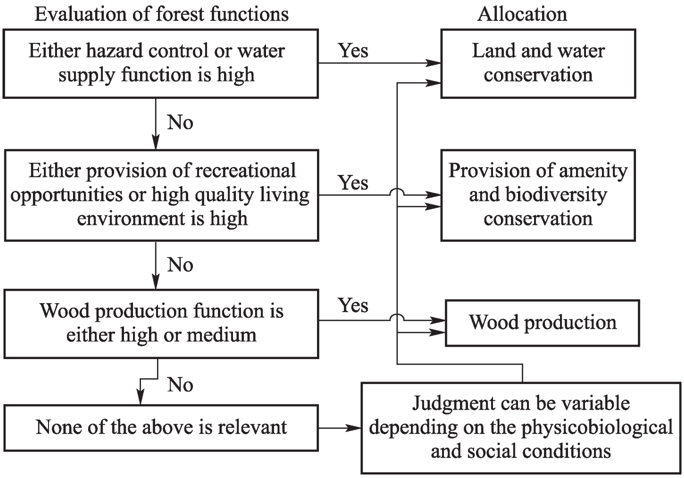

Fig. 11.3 The guideline as to how to determine primary management objective for a forest block, provided by the forestry department.

sciences. In other words, foresters are primarily concerned with what forests can provide in their professional mind instead of what kind of services people can obtain from the forests. Social factors may be significant in evaluating a forest for water supply, scenic beauty and recreation, as we demonstrate later in a case study, because such factors as how forests are utilized, population density in the vicinity or frequency of visits to forests vary considerably. If these social factors are ignored, a zoning process may yield an unreasonable allocation of land to specific management objectives.

In Japan, public involvement is not mandatory in the decision making processes for natural resource management. In the current forest planning process, the only forum for public opinion is the websites of the Japan Forest Agency and their local offices. On those websites, people can tell how large an area is to be allocated for each of the three functional zones for the 44 districts of the national forests. The figures may be changed, if anybody requests, for a smaller unit area of the national forests, other public forests and private forests. The agency or department is supposed to respond to public opinion. However, foresters make the final decisions and they do not need to clarify how they made their decisions. Citizens' groups may take a legal action to change the government's decisions but, in the Japanese context, such actions are not very effective.

11.2 State of the public participation

Arnstein's (1969) hierarchy describes the development of public participation from a level of nonparticipation to the highest level as citizen control. In the current Japanese forest planning system, the participation level would be classified as "consultation" in which they ask opinions and collect views but those who are doing the consulting make final decisions (Hislop et al. 2004). The Japanese mechanisms for public participation are well behind those of other advanced countries. It is comparable with those in the U.S.A. in the 1970s, when public participation consisted of asking the public to comment on a draft plan before the National Forest Service made a final decision (Leach 2006). By contrast, town hall meetings or round tables are held, for example, in a variety of local, regional, provincial and national processes in Canada (Bouin 1998; Lecomte et al. 2005). Government organizations started giving local partners support and encouragement in designing and facilitating new schemes for community forest management in Great Britain, no matter whether forests are public or private, after forest managers changed their attitude from arrogant forestry professionals to those who are prepared to be receptive of broader interests in the 1990s (Weldon 2004). The Finish National Forest Programme was approved in 1999, in which all stakeholders and interest groups were to be included in the preparation, in order for any conflicts to be identified and disputes settled (Primmer and Kyllönen 2006). Today, in the U.S.A., there are elaborate processes for public involvement in planning on national and state forests.

These more advanced systems invite limited number of participants who may be neither democratically authorized nor accountable to the population. The processes are time consuming for all participants (Santos et al. 2006; Elsasser 2007). In the present chapter we propose a method to incorporate public survey results in an earlier stage of the zoning process. There are potential advantages with this type of public involvement. Making use of social evaluations moves up Arnstein's metaphorical ladder, so that such advancement may bring about a decrease in the forest manager's own prejudices against other parties (Moote and McClaran 1997; Shindler and Neburka 1998; Wagner et al. 1998; Côté and Bouthillier 2002). If forest managers perceive wood production as far more important than biodiversity, for instance, but the general public or the majority of stakeholders have very different views, then public involvement may have a significant influence on the outcome of zoning. When the power holders retain the right to decide, even as the public survey results are effectively incorporated in the process of decision making, Arnstein (1969) called this "placation" but indicated that it is a higher level involvement than "consultation" in which the public has no assurance that their views are to be included in the process. In such a case, adding substantive information may contribute to a certain improvement of the quality of decisions, and public involvement may serve to reduce conflicts among stakeholders (Côté and Bouthillier 2002). A rational, technical-scientific model that does not incorporate public concerns makes it difficult to reach decisions that are acceptable to citizens (Shindler et al. 2002). Interested parties may oppose decisions less vehemently if they know that the general public has been involved in the process. Another advantage is that a parameter obtained by public involvement can alleviate unreasonable effects on the outcome of zoning that may result by a difference in how forest services are classified into various categories (Sugimura and Howard 2008). For instance, if planners intend to emphasize water and soil conservation, they can create a larger number of water and land conservation categories relative to other services. There is an educational dimension as an additional advantage (Buchy and Hoverman 2000). The general public may not realize how forests are related to their lives. Providing opportunities to participate in forest planning will certainly make more people consider the importance of forests.

There are various dimensions for public participation that have been practiced in forest planning processes. One that is closely related with this study is whether the general public or certain limited groups of stakeholders are to participate. Stakeholder involvement may be more appropriate if conflicts among interest groups are quite serious so that iterative and intensive exchanges of information, opinions and discussions are necessary. By contrast, involvement of the general public would suffice, if participation aims at collecting information from a broad perspective. In this chapter, we propose a method to determine a general goal of forest planning based on information from a social evaluation (SE). Therefore, we selected the general public approach instead of stakeholders, because we incorporate a wide range of forest services and broad representation is necessary for successful involvement (Bouin 1998; Shindler and Cheek 1999; Leach 2006). Another dimension is whether participation occurs in an early stage or in a later stage of the planning process. It has been suggested that the earlier the public gets involved, the more effective is its influence on the decision (Shindler and Cheek 1999; Appelstrand 2002). If the objective is to select the final plan from among specified alternatives (Martin et al. 1996; Leskinen 2004; Sawathvong 2004; Purnomo et al. 2005; Sheppard and Meitner 2005), participation must occur in a later part. We suggest in the present chapter to introduce public involvement in an earlier part and stakeholder participation later on a smaller regional scale, when more intensive and iterative communications are necessary.

We assume SE scores to represent social evaluation of a category of forest service relative to other categories. Such weighted scores can also be obtained through the Analytic Hierarchy Process (AHP) which has been utilized extensively for decision making in forest planning (Kangas 1994; Ananda and Herath 2003; Mau-Crimmins et al. 2005). However, Sugimura (1993) indicated that respondents had difficulty in comparing many pairs in AHP, when evaluating the importance of a forest service with a scale of intensity. Hajkowicz et al. (2000) also suggested that decision makers acknowledged that paired comparisons were difficult to use. By contrast, both of these studies suggested that ranking was easier to use. Likewise, Ananda and Herath (2003) suggest that, when there are a large number of criteria, pairwise comparison may become tedious to the respondent. Therefore, Leskinen et al. (2004) proposed a method that reduces the number of pairs to be compared by using regression technique. As another alternative to AHP, we present a method in which ranking was the core method and the number of AHP-type comparisons were reduced.

11.3 How to improve the current zoning process

We identify two fundamental elements for improving the current Japanese planning process in the previous section. One is the necessity to incorporate social factors when evaluating site quality (SQA) of a unit of forest land for a certain category of service; the other is to incorporate SE when the size of area allocated for each service category is to be determined. The objective is to give a category a higher priority in cases where local people value it greater than the others. To achieve this improvement Sugimura and Howard (2008) proposed a method that we call SQA-SE. It is designed in such a way that once the public places a relatively high value on one service, a larger area may then be assigned to it as primary management objective (PMO) than would be the case if the SQA alone is used to make allocations. Thus, it is important to consider how large area should be allocated for one service as PMO relative to the other services, before deciding which service is to be assigned to each forest block. Thus, SE modifies the area of forestland that SQA alone has allocated for each service. It may happen that such a modification appears to be too large for the existing capacity of a certain service category. In such a case the goal should be to attempt to enhance the capacity by changing overall strategies in the relevant planning area. For instance, the Japanese Forestry Agency encourages restoring tree plantations to natural forests, taking the current global economy and local social and ecological conditions into account.

The first step is to identify forest management objectives and classify them into categories. In this chapter, we classify forest services into eight categories that are relevant to zoning process and exclude such services as mitigation of global warming, atmosphere purification and noise reduction (Fig.11.4). As we indicate earlier, the outcome of the zoning process may be significantly affected by how the planners classify services into different categories. In fact, planners can even manipulate the outcome via their control of categories. Forest site quality assessment yields SQA scores that represent the degree to which a forest block satisfies each category of forest services. These scores may be based on natural conditions, which are adjusted later for social factors such as the population distribution, accessibility and the location of water sources.

The second step is to determine the size of the area where each service category is assigned as PMO in a given forest planning area (region) using the following methodology. Each region is divided into sub-compartments that correspond to the units of SQA. Each sub-compartment will have a certain

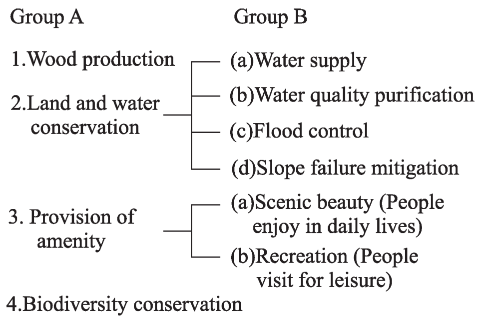

Fig. 11.4 Classification of the forest service categories; groups A and B are categorized to compare with each other for social evaluation.

forest service as the primary management objective (PMO) such that the sum of units assigned to each service yields the area to be assigned to that service. The SQA-SE method determines the size of area for each service, integrating SQA and SE as we describe below. By contrast, the conventional method determines area size solely based on SQA, assuming that all the categories have equal SE. It assigns a large area to a service if it has a high SQA. The SQA-SE method will provide an even larger area if local people indicate a high SE for that service. If they indicate a low SE, on the other hand, a smaller area will be allocated than determined by the conventional method.

Fig.11.5 illustrates the procedure of the SQA-SE and conventional methods, drawing SQA-cumulative area curves for the services. For each kind of forest service any sub-compartment has an SQA score and the scores are arranged on a curve, starting from the highest SQA on the lower left corner, adding the next highest SQA one by one until reaching to the lowest SQA on the upper right. Any point on a curve represents the total area of the sub-compartments that have higher SQA than that point on the horizontal axis. The point a_{ij} on the horizontal axis represents the jth compartment counting from the highest SQA for the service i. Note that the SQA scores are placed upside down on the vertical axis from the highest on the bottom to the lowest on the top. If SQA scores are generally high [e.g. $f(a_{hj})$ in the diagram], the curve takes a lower position.

The conventional method may be comparable with the following approach, in which a line e drawn parallel with the horizontal axis determines the size of area allocated to each category of service. The intersecting points of this line with the curve f (a_{ij}) give the area allocated for the service i, that is A_i' (Fig.11.5). The vertical position of the line e is determined, in such a way that the sum of A_i' is to be equal to the total plan area. So SQA determines A_i', indicating that this process is fundamentally similar with what has been practiced in the Japanese national forests in the sense that decision making was made primarily based on SQA.

In the SQA-SE method, a combination of the SQA curves and SE value for the service i, W_i, determines the area size, A_i, that is allocated for the

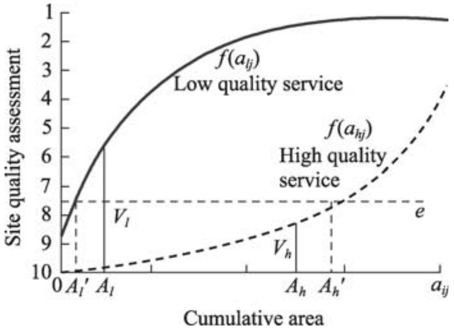

Fig. 11.5 Determination of the area size allocated for each service category based on the cumulative area curves plotted against the Site Quality Assessment (SQA) and social evaluation (SE) of the categories (V_l, V_h). The length of V_i is adjusted in proportion to W_i, the measurement of SE for the service i, so that the sum of A_i is equal to the size of the planning area. If SQA is generally high, the curve takes a lower position. A_i' and A_i denote how large area is allocated for the category i, according to the conventional and SQA-SE methods, respectively. Refer to the text for more details.

service i as PMO. A_i' is moved in accordance with W_i, adjusting the length of the vertical line from $f(a_{ij})$ down to A_i, that is V_i. In such a case as for the service l in Fig.11.5, if W_l is large enough, the area allocated for the service (A_l) increases by moving A_l' to the right, so that the length of the vertical line V_l becomes longer than that from A_l'. Supposing, by contrast, that W_h is low, then, the vertical line, V_h, should be shorter than that from A_h'. If we determine the length of V_i in proportion to W_i, both SQA and SE for the service i may influenceA_i. In Fig.11.5 A_h is larger than A_l due to higher SQA for the service h, while greater SE for the service l shortens the initial distance, i.e. between A_h' and A_l'. By adjusting all the V_i so that the sum of A_i is equal to the whole area, all the A_i can be determined.

The procedure to obtain SE is as follows: The respondents were asked to compare and ranked the four categories in an ordinal scale in groups, A and B (Fig. 11.4), independently. They were asked not to have any specific piece of land in mind, but asked to compare them in terms of their importance in the respondents' life. The results were transformed to pairwise comparisons. For example, if a respondent ranked four categories, land & water, biodiversity, amenity and wood, in this order, he or she evaluated land & water higher than biodiversity, and amenity higher than wood. Thurstone's law of comparative judgment (TLCJ), Case V [e.g. Nunnally (1978) for reference], provides an interval scale of evaluation through a statistical procedure for groups, A and B, individually. This procedure gives a score of zero to the category of the lowest evaluation and a higher score to any of the other categories according to the percentage of people who evaluated a category higher than the other as described below. It is assumed that the respondents' evaluation scores take a standard normal distribution curve with an equal variance for any service to compare with another. The larger the percentage of respondents who prefer a category to the other, the greater the distance between the two normal distribution curves. The percentage of the overlapping area under the two curves provides the z-score. The score can represent the evaluation for a category relative to another, giving the lowest evaluation a score of zero. The next step is to select two categories arbitrarily from each group and let the respondents compare them using the same scale with Analytic Hierarchy Process (AHP) (Saaty 1980). The scale varied from one denoting equal importance to nine denoting absolute importance. We needed this type of comparison because the lowest score TCLJ provides (zero) is not useful for the area allocation we described above.

Then, a combination of results from the two types of questions (TLCJ and AHP) produced a value on a ratio scale for each service, solving the following simultaneous equations for W_i.

$$\begin{aligned}
&\Sigma W_i = 1(i = 1, 2, 3, 4)\\
&W_i = W_1 + a * U_i; U_1 = 0\\
&W_4/W_1 = P
\end{aligned}$$

where W_i is the measurement of SE for the forest service $\underline{i}$ in either Group A or B (i.e. how high people evaluated each service); U_i is the value obtained through TLCJ; P is the value obtained from an AHP type comparison; "a" is a coefficient; and W_1 is the lowest weight of the four.

11.4 On the effective use of social backgrounds and evaluation

The Rokko Mountains are located in the southeastern part of Hyogo Prefecture, north of the Keihanshin Metropolitan Area which has the second largest population in Japan (Fig.11.6). It is heavily used for recreation and is home to some unique species of plants and animals. The area's forests were exploited for timber and fuel during the Pacific War, so most stands are young secondary growth; plantations cover only 7.3% (Japan Ministry of Agriculture and Fishery, 2002). Floods and landslides occurred frequently after the war. The Environment Agency designated 11% of the forests as strictly protected for wildlife. By contrast, Shiso District is remote from large cities and has a much lower population density. Because forestry is one of the major local industries, 76% of forests are plantations (Ibid). Shiso also has some pristine forests at its northern tip, which the National Forestry Agency had planned to log more than 30 years ago, but they decided not to do so after having heated debates with non-governmental organizations that wished to protect these areas. Due to much higher population densities, far more people visit the Rokko Mountains for recreation and observe the mountains in their daily lives than is the case in the Shiso region. Therefore, demand for recreation and scenic beauty should be much higher there than in Shiso.

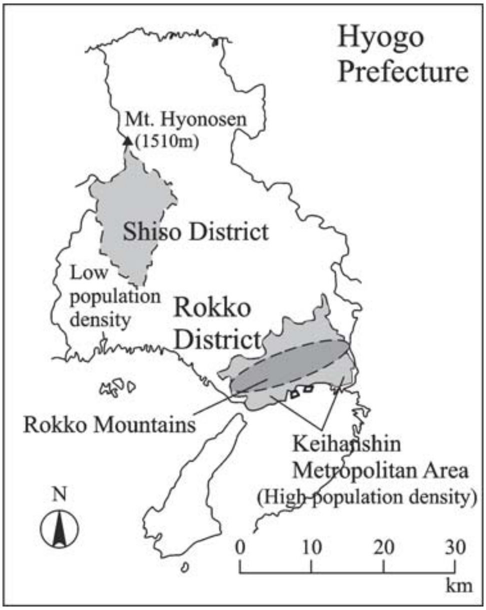

Fig. 11.6 Case study area.

Since the two regions have apparently different forest conditions with contrasting social backgrounds, we should not determine the SQA scores in each region separately. It is not reasonable, for instance, to deal with the highest score for scenic beauty in Rokko and that in Shiso equally, because the number of viewers in each region is very different. In another example, the people around the Rokko Mountains depend on the region for only 29% of their water supply; the balance comes from a river to the east. In Shiso, however, the region is very dependent on the forest for its water. These conditions are to be taken into account to manipulate the SQA scores in a certain manner. Therefore, incorporating social factors in Rokko yielded significant differences in the SQA for water supply and scenic beauty (Fig.11.6). Because most of the water supply in the region comes from outside the district, the scores for water supply were lower than the original SQA scores when we accounted for the percentage of supply from the Rokko forests. The scores for amenity were generally high when they were evaluated solely by natural conditions. Yet, when we took the very high population density in the south of Rokko into account, the Rokko forests away from this area did not attain high scores. Likewise because people visited a small number of places for recreation far more frequently than they did others, the heavily visited areas attained high scores.

As such, social factors tend to have significant effects on the evaluation of site quality when the demand for a certain limited area of forests is great as is the case for recreation and scenic beauty in Rokko forests that are located near very high population densities. A forest can reveal its potential value to a great extent when it provides any of its services effectively for the local people, possibly except biodiversity, for which existence value may be dominant (Stevens et al. 1991). To meet the large demand for water in the metropolitan area, for example, some Japanese municipalities decided to impose a tax for watershed management. Although the watershed is often far from the place where the users live, Kuriyama (1999) found that the majority of city residents agreed to pay such a tax. As for the evaluation of scenic beauty that people enjoy in their daily lives, it is not reasonable to evaluate a forest block very high in a case in which few people live in the area where the forest can be viewed. Likewise, forest planners should not value a forest block highly for recreation when few people visit it now or are likely to do so in the future.

When the SQA for two or more services in a forest block is nearly equally high, they cannot be used by themselves to determine PMO. A slight difference in SQA does not guarantee that one is more important than the others. In such a case, if the local people value one of them distinctively higher than the others, it would be reasonable to give the service with the highest social evaluation (SE) a higher priority than the others. If this process is applied to the whole area, a service that the local people give a high SE may be assigned to a larger area as the PMO than SQA alone would provide. We thought that it would be crucial to determine how large area should be allocated for one service as PMO relative to the others in a zoning procedure, before allocating a service on a certain forest block. To achieve this goal, we referred to the results obtained in a mail questionnaire study (Sugimura 2001) as SE for each service category, in which 444 and 128 residents in Rokko and Shiso participated, respectively (Table 11.1). In this study a brief description of a variety of forest services was given to the respondents, who were then asked to compare the categories with each other.

Table 11.1 Social evaluation (SE) values and area size allocated for each category of forest services.

Region	Wood	Slope failure	Water supply	Water quality	Flood	Scenic beauty	Recreation	Biodiversity
SE (W_i)								
Rokko	0.10	0.11	0.14	0.06	0.14	0.13	0.05	0.27
Shiso	0.20	0.11	0.13	0.07	0.13	0.13	0.03	0.21
Area size (km^2)								
Rokko	4	35	8	23	96	31	4	72
Shiso	502	17	139	0	26	0	0	21

SE has no unit and the values total one.

The outputs from the SQA-SE method revealed that incorporating SE had greater effects on the Rokko allocation. Taking SE into account caused a difference of 65% in Rokko and 11% in Shiso in terms of the area size allocated for each service as PMO. Two factors apparently brought about the difference. One is that SQA was generally low in Rokko so that the determination of PMO was more sensitive to the variance among SE scores than in Shiso. The other factor is the difference between high SQA and low SE for water quality and the opposite tendency for biodiversity in Rokko. Such a discrepancy between SQA and SE caused a greater influence on the SQA-SE method. Therefore, biodiversity, whose SE was 4.5 times higher than that of water quality, was able to attain a large share of forest. By contrast, wood production and water supply, which attained the greatest share of PMO in Shiso, had the second and third highest SE (Table 11.1).

The SE scores proposed in the present study can be used in a similar way to AHP. This approach reduces the number of pairwise comparisons in AHP, making it easier for participants to respond to questions (Sugimura 1993; Hajkowicz et al. 2000; Ananda and Herath 2003) and reducing survey cost because it uses simpler questionnaire composition than AHP. The scores can be used to yield general guidance about how large an area of forest for each service is allocated as PMO. If SE scores are used in an early part of a zoning process, it would be easier to compensate for any inadequacies in the outcome in a later part of the process. There are also some other ways to produce SE in a simple manner. It is also possible to reduce the number of pairs in AHP, using a regression technique (e. g. Alho and Kangas 1997; Tahvanainen et al. 2001; Leskinen et al. 2004). Although the number of comparisons is similar to each other, a combination of AHP and the regression uses more precise scale of measurement. Yet, as Sugimura (1993) and Hajkowicz et al. (2000) suggested, respondents might feel more comfortable with the method proposed in the present study which includes ranking order instead of paired comparisons with relative scale. Borda count is another method that may yield similar SE scores (Kangas and Kangas 2005). In this method, when there are n categories to compare, the most preferred obtains the score, $n-1$, from each voter and the least preferred obtains 1. The total of these scores from all the voters provides the evaluation for each category. The greater the number of categories to compare, the greater the difference in evaluation between the more preferred and the less preferred categories. Therefore, the results can be manipulated by including categories that voters would certainly rank very low (Kangas et al. 2006). SE can also be produced by voters giving utility values, which is similar to Contingent Valuation Method. Kant and Lee (2004) suggested that there are many fundamental problems with this technique. For instance, the value may be inflated enormously to give a significant bias to a certain alternative (Kangas et al. 2006). Besides, the value of forest services may be too ambiguous for the respondents to give confident utility values. Thus, there are some candidates for producing appropriate SE scores. Even though AHP appears to be most frequently used, Ananda and Herath (2003) suggested many more empirical studies would be necessary to examine its effectiveness. If the number of categories is not too great to compare, AHP would yield useful SE scores. We think the more important matter is that SE scores are to be used in an early stage of zoning process.

11.5 Experts vs. the general public

Forest science has diverse fields of disciplines and experts naturally emphasize the importance of their specialty. Alho et al. (1996) denoted that the views of experts can vary considerably depending on their specialty, whereas Steel et al. (1994) indicated that the general public has strong biocentric value orientations toward forests as Sugimura (2001) also suggested. Local people would expect forests to be managed in accord with their needs and preferences. Protesting against clear-cutting old growth forests in the 1970s and 80s was such a case, as noted previously. In another questionnaire study, Kuriyama (1999) reported that the respondents were willing to make a larger donation to managing forests for water supply than to recreation or wood production. The public's preferences were different from those of foresters, that resulted in significant changes in forest policy in some local areas, even though it is generally acknowledged that decreasing dependence on domestic wood supply has much more strongly affected Japanese national forest policies. Our experience suggests that many Japanese foresters have naturally developed emotional preferences toward a limited number of plantation tree species due to their educational backgrounds and professional experiences as guardians of tree plantations. Such foresters' preferences may be the reason why biodiversity has been neglected in the forest zoning process for a long period of time. In the meantime natural forests had been replaced with plantations extensively in the southwestern part of Honshu, Kyushu and Shikoku Islands (Fig. 11.1). Therefore, such public evaluations should be reflected in the forest planning process not only because foresters and the public have different values, but also because wood production is often not economically feasible even with financial incentives in Japan, where domestic production fell down in recent years despite such incentives. For instance, forests had been cut clear extensively with marginal profits, while receiving a large amount of government subsidies for wood production, despite the serious negative effects on some endangered species (Sugimura 1988). Such clear cutting lasted until a further decline in the price of imported wood made logging infeasible (Sugimura et al. 2003). Tanz and Howard (1991) argued that the public, rather than a forestry agency or an interest group, can make reasonable choices. Earlier in the present chapter, we proposed a method to take people's expectations and desires (SE) into account quantitatively in a zoning system, in which they gave a generic expression of human preferences irrespective of location. A combination of SE and experts' evaluations may yield an outcome, which is significantly different from the one that is based solely on experts' evaluations (Sugimura and Howard 2008). It should be noted, however, that people evaluate how they perceived the importance of the forest services and they do not have to know how effectively a forest would fulfill those services. Therefore, a weighted value, SE, for a kind of service is not necessarily proportional to the area apportioned to it.

The Japanese zoning system does not concern itself with how to classify the forest services. Dividing forests into three zones, (a) wood production, (b) land and water conservation, and (c) amenity and biodiversity, based on SQA alone means they treat these three equally. The Japanese Cabinet Office (2003), which conducted the most recent public survey on this matter, indicated that they considered the order of importance would be (b) land and water conservation, (c) amenity and biodiversity and (a) wood production. This result conforms to SE in the present study. The classification of forest services may significantly influence the outcome of zoning as indicated before, while the current classification does not consider SE in the zoning process. Decision makers, if they wish to enhance the importance of a certain type of forest service, would be able to manipulate the outcome by increasing the number of categories the forest service includes. However, if SE is taken into the procedure, this kind of manipulation would be less successful (Sugimura and Howard 2008).

Introducing SE through public involvement, therefore, would have several advantages. First, as Côté and Bouthillier (2002) suggested, public involvement should be able to give useful information to amend the bias toward wood production that the forest managers often maintain. A new parameter, SE, is able to alleviate this problem. Second, it is generally argued that public involvement would be more effective when it occurs in an early stage of planning (Shindler and Cheek 1999; Appelstrand 2002). We suggest that the general public participate to provide SE in an early stage and that stakeholders may participate in a later stage to select a specific alternative plan as many studies suggested. Third, incorporation of SE can move up Arnstein's (1969) ladder from "consultation" level in the current Japanese zoning process to "placation", which can be measured by the degree of power that citizens have over decision making (Beierle and Cayford 2002). As we presented in the case study, the SQA-SE method would have biodiversity allocated over 25% of Rokko's area, while currently only 11% of Rokko forests are protected strictly for biodiversity. Thus, incorporating SE can make some significant changes in the outcome, even though the participation level remains relatively low according to Parkins and Mitchell (2005). Foresters tend to think that they are experts and much more familiar with forest management. By contrast, the public is interested in a greater scope of benefits and welfare they gain from forests including existence values of endangered species (Jakobsson and Dragun 1996). Therefore, experts and the public are likely to experience tensions in the planning process (Mascarenhas and Scarce 2004). Citizens' groups protested against cutting the last remaining old growth forests in the past, so we expect that shifting toward biodiversity conservation according to SE would have mitigated previous conflicts between foresters and conservationists in Japan.

Only web-based public inputs are utilized in the current Japanese forest planning process and the central government provides too little time for each local office to include effective participatory processes in the decision making. Within that context, it is not presently possible to incorporate the methodology we have developed. Another problem is that no previous scientific study on forest zoning in Japan has proposed any systematic means of public involvement that significantly affects the outcome of a zoning process. Only a few exceptions can be observed in some local government processes where a form of consultation with stakeholders occurs near the final stage of planning (Nishikawa 2004). Therefore, available means for forest managers to employ such involvement appear to be quite insufficient. Yet, our proposed methodology can be carried out with relatively small costs by reducing the number of categories of forest services as well as simplifying SQA based on currently available information and professional expertise. By contrast, conflicts between foresters and conservationists are keen in some areas of Japan. These days, some citizens' groups have brought cases to court, which normally takes a long period of time before the final judgment is given. No matter whether they win or lose, the cost of not introducing SE can be enormous. Therefore, although the process may require more cost and time initially, the proposed methodology can contribute to more effective implementation of forest planning.

11.6 Perspectives of the future

At present the government seems to be promoting a longer cutting cycle and the mixing of broad-leafed trees with even-aged coniferous forest, whose effects have not yet been revealed. Larger areas of national forests are set aside for nature protection than before. Public opinion surveys imply that any shift in subsidies from wood production to watershed management or biodiversity conservation is likely to be acceptable. On the other hand, foresters have not developed a planning process in which the public participates in the decision making for using the nation's forests and plantations. A few fundamental adjustments to the current system would be necessary for the proposed system to be incorporated into the zoning procedure. First, site quality assessment should not only adopt social factors but also have as precise scale as possible. It can be scaled down to a practical level later, while an evaluation with too simple a scale cannot be transformed into finer one later. Secondly, since social evaluation changes with time as the past national public survey suggests, monitoring SE is essential for adaptive forest management. Third, after determining the general goal of zoning in terms of how much area may be allocated for each category of forest service, PMO must be assigned to each forest sub-compartment. We propose that stakeholders should participate in the latter stages of zoning process. Where special forest products are regarded as valuable resources, the local community maintains a traditional system of participatory decision making in their community forest management. Yet, this system has never been incorporated into the forest planning policy for the national and local governments. All these conditions support the need for development of a forest network, in which planners work with experts and the general public fitting information given by them into a decision making system.

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Chapter 12 Forest Fragmentation: Causes, Ecological Impacts and Implications for Landscape Management

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Abstract

In order to enable the development of appropriate landscape management plans, the causes and impacts of fragmentation should be fully understood. A new definition, incorporating the key aspects cited in landscape ecological literature since the 1980s, is proposed in order to shed light on the matter of fragmentation. By means of two case studies in the Democratic Republic of the Congo (Oriental Province) and in North Benin, the key role of anthropogenic activities in landscape fragmentation is evidenced; the spatial dispersion of forest vegetation is linked to population density and land use change. The potential impact of fragmentation on biodiversity is shown by an analysis of forest diversity in Ivory Coast (Tanda region), and by a study of edge effects on two rodent species in the Democratic Republic of the Congo (Kisangani). The chapter is concluded by an study on how planned corridors, assuming a spatial regrouping of existing teak plantations, could contribute to the conservation and management of remaining natural forest patches in the Atlantic Department in Benin.

Keywords

Corridor, edge effect, forest degradation, fragmentation, land cover change, landscape connectivity, landscape metric, population density, transition matrix.

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12.1 Fragmentation: A plenitude of definitions

The process of forest fragmentation due to human activities such as logging or conversion of forests into agricultural areas and suburbanization (Forman 1995) has been identified as the most important factor contributing to the decline and loss of species diversity worldwide (Noss and Cooperrider 1994). Forest fragmentation occurs when a large region of forest is broken down, or fragmented, into a collection of smaller patches of forest habitat (Wilcove et al. 1986; Collingham and Huntley 2000; Fahrig 2003). The outcome of fragmentation can be considered as a 'binary landscape' in the sense that the resulting landscape is assumed to be composed of spatially dispersed forest fragments with a non-forest matrix between them (Franklin et al. 2002).

Defining fragmentation is crucial in evaluating its effects on species in the forest ecosystem and at the landscape level (Bogaert 2003; Lafortezza et al. 2008). A spectrum of definitions has been cited in landscape ecological literature since the 1980s, of which a representative sample is listed below:

Fragmentation . . .

... is the process whereby a large, continuous area of habitat is reduced in area and divided into two or more fragments (Wilcove et al. 1986);

... is an alteration of the spatial configuration of habitats that involves external disturbance that alters the large patch so as to create isolated or tenuously connected patches of the original habitat (Wiens 1989);

... is an event that creates a greater number of habitat patches that are smaller in size than the original contiguous tract(s) of habitat (Bender et al. 1998);

... is habitat loss and isolation (Collinge 1996);

... refers to the patchiness of a landscape (De Santo and Smith 1993);

... produces a series of remnant vegetation patches surrounded by a matrix of different vegetation and/or land use (Saunders et al. 1991);

... is the process of breaking up continuous habitats, resulting in reduced area, increased edge, reduced interior area, increased isolation of patches and possibly increased number of patches and decreased average patch size (Davidson 1998); an increase in the total boundary length is also observed (Forman 1995);

... is the breaking up of a habitat, ecosystem or land use type into smaller parcels (Krebs 1994; Forman 1995); it is considered as a spatial process of land transformation (Forman 1995; Bogaert et al. 2004);

... is heterogeneity in its simplest form: the mixture of habitat and non-habitat (Franklin et al. 2002);

... refers to an increase of the number of patches in a landscape (Goodwin and Fahrig 2002);

... is the breaking up of extensive landscape features into disjunct, isolated or semi-isolated patches as a result of land use changes (Heywood and Watson 1995);

... is the breaking apart from habitat, and does not refer to habitat loss (Fahrig 2003; Yaacobi et al. 2007);
... is the disruption of continuity; when defined in this manner, the concept can be applied to any domain in which continuity is important to the functioning of ecosystems (Lord and Norton 1990);
... is a particular form of human-induced environmental degradation (Haila 2002);
... is a process of spatial landscape transformation, characterized by habitat loss, and an increase of the number of patches (Forman 1995; Jaeger 2000; Bogaert et al. 2004);
... is the complement of connectivity (Riitters et al. 2000);
... is both a state (or outcome) and a process; the process of habitat fragmentation is the set of mechanisms leading to a state of discontinuity of resources and conditions (Franklin et al. 2002);
... is conversion from natural vegetation to new land uses; the remaining habitat is inevitably divided into increasingly smaller parts (Groom and Schumaker 1993).

Although the limitations of the overview should be considered, four main features can be identified in these definitions:

- a continuum of habitat or vegetation is reduced to a discontinuum, composed of at least two ('more than one') patches;
- habitat destruction or loss is observed;
- spatial pattern is characterized by patch isolation due to the loss of the connecting habitat;
- habitat-matrix interactions are changed by an increase in cumulative patch perimeter, reducing total interior area (edge effect).

Taking these four key elements of fragmentation into account, a comprehensive definition could be proposed, i.e. *fragmentation is the process of breaking up continuous habitats and thereby causing habitat loss, patch isolation and edge effects* (Bogaert 2000). Some authors emphasize the distinction between the concepts of habitat loss and fragmentation (Franklin et al. 2002; Haila 2002; Fahrig 2003; Yaacobi et al. 2007), mainly due to the impact on diversity. Since many landscape ecologists accept both concepts to be inextricably related, we suggest to consider habitat loss as a component of fragmentation. For a more complete overview of existing definitions and views on fragmentation, and on its effects, the reader is referred to Haila (2002) and Fahrig (2003).

For Cadiz Township (WI, USA), the textbook example of fragmentation (Curtis 1956; Burgess and Sharpe 1981; Shafer 1990; Forman 1995) referring to land cover changes between 1831 and 1950 during the period of European settlement, the aforementioned four pattern features have been observed. Bogaert et al. (2004) confirmed this observation when the entire period was considered, but showed also that this sequence of land cover change could be disentangled into three distinct phases, in which only the first one (between

1831 and 1882) corresponded to fragmentation; the first subsequent phase (1882-1935) was characterized by patch attrition; patch shrinkage concluded the observed dynamics between 1935 and 1950.

In their overview of habitat fragmentation experiments, Debinski and Holt (2000) emphasize the wide range of species responses to fragmentation. Species can show highly disparate responses to fragmentation, including lack of response (Davies and Margules 1998). It should be noted that the aforementioned list of definitions only refers to nonspecific definitions of habitat or forest fragmentation, i.e. definitions independent of any particular species. This approach fits in with an aspiration to develop universal theories, applicable at the landscape as an entity and suitable for a variety of species, i.e. the entire landscape (eco)system (Bogaert 2000). This view is, however, considered also an ambiguity of the fragmentation concept (Haila 2002) contested by ecologists occupied by species-driven research (Franklin et al. 2002).

Within the emerging issue of landscape management and conservation, this contribution aims to explore forest fragmentation through the analysis of the main causes and ecological impacts. To achieve this objective, a deliberate choice is made for an approach based on case studies, referring to local or regional land cover dynamics in West and Central Africa (Benin, Ivory Coast, Democratic Republic of the Congo). This type of approach is justified by the variability of the ecosystems and landscapes subject to fragmentation worldwide, and by the ongoing discussion and controversy on the (ecological) consequences of habitat fragmentation (Bogaert 2003; Fahrig 2003; Ewers and Didham 2006; Yaacobi et al. 2007).

By means of simple fragmentation metrics such as the Monmonier index (Monmonier 1974), average patch size and the index of the largest patch (McGarigal and Marks 1995), it is shown how fragmentation affects biodiversity and how anthropogenic pressure (measured directly by land cover change or indirectly by means of the population density) fragments natural land covers. The use of simple metrics, instead of a long series of complex and often correlated metrics, is also a deliberate choice, an issue still subject to debate in landscape ecological literature (Bogaert and Hong 2004; Li and Wu 2004; Bogaert and Mahamane 2005).

The first two case studies described in this chapter focus on the drivers of fragmentation. The first study investigates how population density leads to a lower presence of dense forests in the Oriental Province of the Democratic Republic of the Congo. An increasing degree of forest fragmentation is observed for increasing population densities. The second study deals with land cover change in North Benin, where agricultural development, cotton production in particular, has substituted the original forest and savannah vegetations.

After these two examples, the focus of the chapter is moved towards the impacts of fragmentation on biodiversity. The first study, situated in a forest-savannah transition zone in Ivory Coast (Tanda region), links the degree of landscape fragmentation to forest diversity itself and to the presence of heliophylic species; the latter species group is accepted to be an indicator of habitat disturbance and canopy openness. The second study, based on data collected in the Democratic Republic of the Congo (Kisangani), discusses the impact of edge habitats, a direct consequence of landscape patchiness, on the abundance of two rodent species.

Finally, a fifth case study is included in this chapter in order to illustrate how a fragmented landscape could be restored by means of planned, continuous corridors. The resources for these corridors, teak plantations, are actually already present but scattered throughout the landscape. A spatial regrouping of these patches in a network connecting valuable forests could contribute to a better management and conservation of the remaining diversity.

12.2 Demographic development and anthropogenic activity as drivers of fragmentation

In this section, two case studies illustrate the central role of anthropogenic activities and demographic pressure in land cover dynamics.

12.2.1 Forest fragmentation and population density in the Democratic Republic of the Congo (Oriental Province)

12.2.1.1 Context, data set and methods

In the tropical zone, forest cover is assumed to be a direct consequence of population density (Williams 2000). In Central Africa, for example, Bogaert et al. (2008) have explored this pattern and found a negative relationship between population density and forest cover. Other studies have considered the key role of the political and socio-economical context (e.g. land tenure system) in determining forest cover in the tropics (Hecht 1985; Angelsen and Kaimowitz 1999; Geist and Lambin 2001).

We investigated the relation between forest fragmentation and population density for a study area in the Oriental Province of the Democratic Republic of the Congo. The Congo Basin, the second largest forest area worldwide, is threatened by an increasing population density and agricultural development, mining industry, urbanisation and deforestation (PFBC 2006). The study area has been defined using a Landsat ETM+ scene (March 2001; path/row 176/060) covering a portion of 185×185 km^2 of the Congo Basin. The central point of the study area is situated at $0°0'$ N and $25°0'$ E; the city of Kisangani ($0°31'$ N, $25°11'$ E) is situated in the northern part of the scene. The scene has been divided in 266 grids of 10×10 km^2. Population density has been calculated by means of the Africa Population Distribution Database (UNEP

2004) which consists of a population density grid-map of 2.5 km spatial resolution. An upper threshold of population density of 100 habitants/km^2 was applied in a grid to be included in this study; 246 grids corresponded to this criterion.

Forest pattern has been quantified using two fragmentation metrics: the number of patches, and patch dominance D_j also known as the index of the largest patch (McGarigal and Marks 1995); the latter was defined as the proportional area (%) taken by the largest patch ($a_{\max,j}$) in the patch type j with total area $a_{t,j}$:

$$D_j = \frac{a_{\max,j}}{a_{t,j}} \tag{12.1}$$

12.2.1.2 Results and discussion

Figs.12.1 and 12.2 show the correspondence between the fragmentation metrics (number of patches, index of the largest patch) and population density.

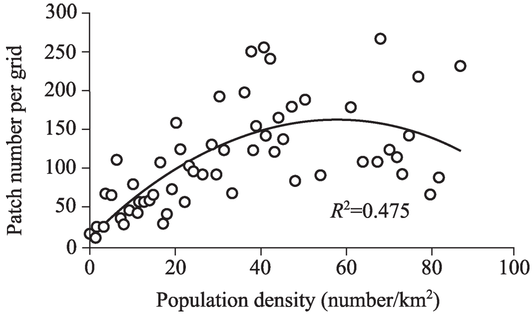

Fig. 12.1 Impact of population density on forest fragmentation in the Congo Basin (Oriental Province, Democratic Republic of the Congo). Fragmentation is measured by means of the number of forest patches. Data are presented on a grid base (grid size equal to 10 × 10 km^2). Graph based on index averages for every population density.

A significant ($p < 0.01, R^2 = 0.48$) nonlinear relation is observed between the number of patches and population density, indicating that forest fragmentation is the highest for intermediate population densities. Patch number increases with population density up to ~60 inhabitants/km^2. Afterwards, patch number decreases again. This example underlines the complexity of landscape dynamics which are often characterized by a sequence of land transformation processes (Forman 1995; Bogaert et al. 2004) as already mentioned in section 12.1. The initial patch number increase can be interpreted as forest fragmentation; the second part of the curve corresponds to forest attrition, in which the initially created patches disappear.

The link between forest fragmentation and population density is confirmed by the index of the largest patch; this index decreases when population density increases ($p < 0.01, R^2 = 0.67$): the higher the population density the lower the proportion taken by the largest patch, indicating that patches become smaller and are of comparable size when the number of inhabitants increases. Fragmentation and attrition lead to landscapes characterized by scattered land cover classes without dominant patches.

Both regression coefficients suggest that demographic factors play a principal role in changing forest cover pattern, which confirms the hypotheses of Williams (2000) and Bogaert et al. (2008). In the study area, this finding was expected due to the subsistence-type economy, in which people are often obliged to draw their daily needs from natural resources, leading to degradation of the forest resources (Bamba et al. 2008).

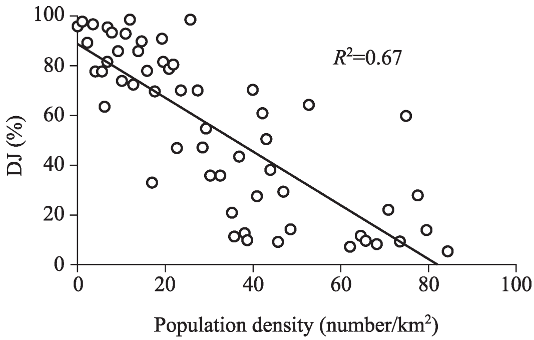

Fig. 12.2 Impact of population density on forest fragmentation in the Congo Basin (Oriental Province, Democratic Republic of the Congo). Fragmentation is measured by means of the index of the largest patch (D_j). Data are presented on a grid base (grid size equal to 10×10 km^2). Graph based on index averages for every population density.

12.2.2 Forest and savannah fragmentation as a consequence of cotton production in North Benin

12.2.2.1 Context, data set and methods

Since its introduction in Benin in 1946, cotton production (*Gossypium spp.*) has known a large expansion in the north of the country and has become the main source of income of its population (MAEP 2000). Cotton represents about 97% of the receipts of agricultural exportations (FAO 2004). As a consequence of cotton culture expansion, forest and savannah area have decreased and the concomitant ecosystems have been fragmented. This relationship between agricultural development, landscape dynamics and habitat fragmentation has been studied in detail for the Banikoara region, the principal zone of cotton production in Benin.

In Banikoara, a study area of 192 km^2 has been chosen, with its centre situated at approximately 11°13′ N and 3°54′ E. Three Landsat TM/ETM+ images (1972, 1986 and 2006) have been classified into four main land cover classes: forest, savannah, agriculture (fallow/field complexes) and built-up. The complexes of fallow lands and fields were mosaics of old and new cotton fields. Image analysis (classification, class area analysis) was executed with ENVI 4.2 and ArcGis 9.2. Landscape dynamics was analyzed by a comparison of the area of each land cover class between the images. A transition matrix, revealing the area exchanges between the classes considered, was composed to understand the dynamics observed. Landscape fragmentation was measured by the average patch size per type (McGarigal and Marks 1995):

$$a_{av,j} = \frac{a_{t,j}}{n_j} \tag{12.2}$$

with $a_{av,j}$ the average patch size, $a_{t,j}$ the total class area and n_j the number of patches of the class considered. Increasing degrees of fragmentation will lead to lower values of $a_{av,j}$.

12.2.2.2 Results and discussion

The analysis of the land cover dynamics between 1972 and 1986, and between 1986 and 2006, showed that land use had shifted towards agriculture and that natural habitats (forest, savannah) decreased considerably in area (Table 12.1).

Table 12.1 Land cover (%) of the Banikora region in 1972, 1986 and 2006. Total study area equal to 192 km^2

Land cover	1972	1986	2006
Forest	57.3	13.7	4.2
Savannah	34.4	39.8	16.2
Agriculture	8.1	45.1	75.4
Built-up	0.2	1.3	4.3

While forest and savannah dominated the landscape in 1972 (cumulative area 176 km^2), their area decreased to 103 km^2 and 39 km^2 in respectively 1986 and 2006. Agricultural land use increased in the same period from 16 km^2 (1972) over 87 km^2 (1986) to 145 km^2 (2006). A landscape dominated by a forest matrix evolved hence in a 34-year period into an agricultural landscape. These observations coincide with those of Arouna et al. (2002) in North Benin between 1975 and 1998, where forest galleries, open forest and savannah vegetations have been ousted by a mosaic of fallow lands and fields, characterized by a regression of 41% of the forest area. In the centre of Benin, Oloukoi et al. (2006) have also recorded a dominance (61.2%) of agricultural fields over forests. An increase of the importance of the built-up class was also noted, likely due to an increase of the population density.

The observed substitution of forest and savannah by agricultural fields and fallow lands is evidenced by the transition matrix between 1972 and 2006 (Table 12.2). It shows that only a small fraction of the original forest area (3.95%) was not subject to change, and that the forest was mainly transformed into agricultural land use (83.32%) over time. The savannah vegetation was more resistant to land use change (27.23%), nevertheless a large part is also used for agricultural production (65.08%). Agricultural land cover was the most stable patch type (76.85%), and its conversions to forest (2.95%) and savannah (12.10%) can be linked to shifting agriculture (Bogaert et al. 2008). Built-up was the second most stable class (47.46%). The conversion of built-up to savannah (32.43%) is explained by land abandonment and demographic shifts. The initial villages are abandoned for new ones more closely situated to more productive lands. The transition of built-up to agriculture can be explained by the recuperation of formerly inhabited zones by immigrants.

Table 12.2 Land cover dynamics in the Banikoara region between 1972 and 2006. All land cover fractions are expressed as a percentage of the 1972 class area. Total study area equal to 192 km^2.

		2006			
		Forest	Savannah	Agriculture	Built-up
1972	Forest	3.95	8.03	83.32	4.70
	Savannah	4.85	27.23	65.08	2.84
	Agriculture	2.95	12.10	76.85	8.10
	Built-up	0.80	32.43	19.31	47.46

The land cover dynamics described in Tables 12.1 and 12.2 have had profound impacts on the landscape configuration (Fig. 12.3).

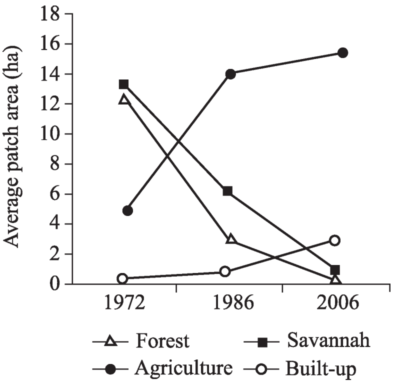

Fig. 12.3 Evolution of the average patch area for each land cover class in the Banikoara region between 1972 and 2006.

Increased levels of fragmentation were observed for the classes representing natural land covers, as quantified by the average patch size, which dropped considerably for the forest and savannah land cover classes, especially between 1972 and 1986, and which increased remarkably for the agricultural land cover in the same period. The built-up land cover class had the highest increase within the period between 1986 and 2006.

The aforementioned tendencies reflect an anthropisation of the Banikoara region between 1972 and 2006, reflected by a substitution of the forest landscape matrix by cotton fields, and by an increase of the population of about 46% between 1992 and 2003 (INSAE 2003). This type of agricultural development, where natural land covers are replaced by anthropogenic ones, leading to a more fragmented status of the natural habitats, was earlier confirmed in Benin by Codjia and Gnagna (1993), Tenté (2000) and Orékan (2007), and for the Democratic Republic of the Congo by Bamba et al. (2008) and Bogaert et al. (2008).

12.3 Empirical evidences of the impact of fragmentation on biodiversity

In this section, the effect of habitat fragmentation on biodiversity is shown by means of two case studies.

12.3.1 Fragmentation alters forest diversity in a forest-savannah transition zone (Tanda region, Ivory Coast)

12.3.1.1 Context, data set and methods

Many studies have reported rapid land cover changes in the tropics (Skole and Tucker 1993); landscape fragmentation as an expression of anthropogenic activities has been considered one of the dominant drivers of landscape dynamics (Bucini and Lambin 2002). Forests situated at forest-savannah contact zones are considered to be more vulnerable to this type of degradation because of the high frequency of edges and of the heterogeneity of the landscape matrix itself (Hennenberg et al. 2008). The forest-savannah contact zones situated in the eastern part of Ivory Coast are accepted as the physical expression of the expanding savannah vegetations nearby (Barima et al. 2009). For the Tanda region, characterized by such transition zones, the impact of fragmentation on forest species diversity was studied in order to quantify the consequences of anthropogenic landscape degradation.

The study area is situated in the mesophylic sector of the Guinean domain; its principal climax vegetation is the humid semi-deciduous dense forest. Three vegetation types dominate the land cover: secondary forest, wooded savannah and tree savannah (Guillaumet and Adjanohoun 1971). By means of Landsat

ETM+ imagery of 2002, forest landscapes were identified (Barima et al. 2009) and 50 sites of each 1 km^2 were defined based on a randomized stratification strategy. For every site, the number of forest patches (N) and the total forested area (A, expressed as the number of pixels) were determined using ArcGis 9.2. Patches were defined considering four neighbouring positions for every pixel. The degree of forest fragmentation (F, Monmonier 1974) was consequently calculated:

$$F = \frac{N-1}{A-1} \tag{12.3}$$

High values of the fragmentation index correspond to a high degree of fragmentation. A forest inventory was executed in one patch per site; species with DBH>10 cm were included. Species names were based on Lebrun and Stork (1991-1997). Species were classified as heliophylic or non-heliophylic according to Hawthorne (1996), Molino and Sabatier (2001) and Bakayoko (2005). The presence of heliophylic species is accepted as characteristic for open, disturbed vegetations. The diversity of every site has been quantified using the Shannon index (H', Magurran 2004), which integrates, in one single metric, both richness and evenness of the sample:

$$H' = -\sum_{i=1}^{S} p_i \log p_i \tag{12.4}$$

with S the number of species in the sample and p_i the proportional abundance of the i-th species. A high value of H' reflects the presence of many species and/or the absence of dominating species in the sample. Poor, dominated samples will be characterized by low values of H'. The relationship between forest fragmentation, the percentage of heliophylic species and forest diversity was investigated by linear regression.

12.3.1.2 Results and discussion

Forest diversity was found to be determined by forest fragmentation (Fig. 12.4, $R^2 = 0.41, p < 0.001$). Also the presence of heliophylic species was influenced by the patchiness of the landscape ($R^2 = 0.33, p < 0.001$). Forest diversity diminished when forest fragmentation increased, which suggests that communities in degraded landscapes contained less species, or that they were dominated by one or a few species, or a combination of both. At the same time, a shift towards communities with more heliophylic species was observed with increasing fragmentation.

Fragmentation creates more open forest canopies (Uhl et al. 1997), which favours the proliferation of these species developing in direct sunlight. This observation, which links fragmentation to canopy openness and hence to tree density, suggests a key role in degradation for selective forest logging. The higher numbers of large heliophylic species or pioneers in logged forest, shown here and elsewhere (Bischoff et al. 2005; Berry et al. 2008), suggest that these

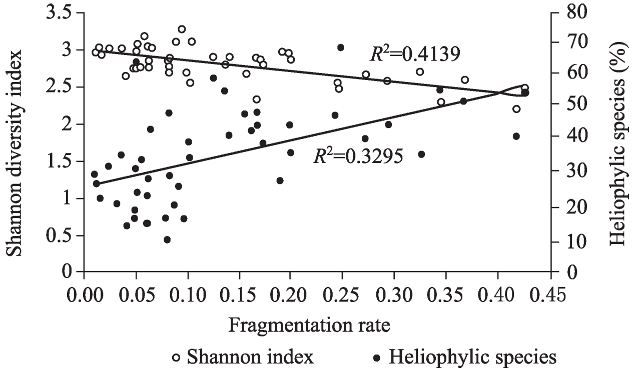

Fig. 12.4 The impact of anthropogenic forest fragmentation on forest diversity in the Tanda region (Ivory Coast), measured by the Shannon diversity index and by the presence of heliophylic species. Fragmentation was measured by the Monmonier fragmentation index (Monmonier 1974).

species were more likely to survive in logged forest or became established in greater numbers soon after disturbance (Whitmore 1984). The empirically shown impacts of anthropogenic patchiness on forest diversity and vegetation composition confirm earlier observations by Benítez-Malvido and Martínez-Ramos (2003) and by Berry et al. (2008).

12.3.2 Edge effects and rodent abundance in Kisangani (Democratic Republic of the Congo)

12.3.2.1 Context, data set and methods

Habitat fragmentation leads to an increased frequency of patch edges (Bogaert 2000). Edge effects are observed when two different land cover types are adjacent and when both types are sufficiently different in structure (Forman 1995; Farina 2000a). Due to the contact with a contrasting land cover, the peripherical zones of both land covers are altered with regard to their microclimates; as a consequence of the strong link between ecological conditions and biodiversity, differences in species composition are observed between the central unaltered parts of a patch and the area along its perimeter (Forman 1995; Kolasa and Zalewski 1995; Bogaert et al. in press). Measuring these edge effects is appealing to improve understanding of anthropogenic effects on landscapes (Bogaert et al. in press). Consequently, empirical data should be collected *in situ* to enable more realistic estimates of edge effects and their ecological consequences (Chen et al. 2008).

The influence of edge effects on rodent diversity was assessed by means of the presence of two rodent species, *Hybomys univittatus* and *Praomys* cf. *jack-*

soni, in the Masako Forest Reserve (MFR, 0°36′ N et 25°13′ E) in the Democratic Republic of the Congo (Iyongo Waya Mongo 2008). MFR, with an area equal to 2,105 ha, is situated at about 15 km of Kisangani and is characterized by an equatorial-continental climate type denoted as "Af" according to Koppen (Dudu 1991). MRF is mainly composed of primary forests of *Gilbertiodendron dewevrei* (Caesalpiniaceae), next to secondary forests, swamp forests and fallow lands (Makana 1986; Kahindo 1988; Mabay 1994). A heterogeneous (*sensu* mixed or transitional) edge zone of about 40 m situated in between a secondary forest and a fallow land has been selected to assess edge effects on the presence of the two aforementioned rodent species.

Four capture zones have been installed covering both adjacent land cover types and the transition zone in between; each capture zone was composed of four parallel transects of 350 m situated 10 m apart. The distance between two traps inside a transect was 7 m (50 traps per transect) and *Elaeis guineensis* pulp was used as bait. Individuals have been captured by means of classic "Lucifer" rat traps between November 2007 and January 2008, a period characterized by one rainy month followed by two dry months. Captures have been pooled for every habitat type involved and compared between the habitats by means of the Kruskal-Wallis and Mann-Whitney statistical tests. Species have been identified by phylogenetic sequence analysis (cytochrome of mitochondrial DNA) using a data base of DNA sequences (Terryn et al. 2007) at the Royal Belgian Institute of Natural Sciences (Vertebrate department).

12.3.2.2 Results and discussion

Three hundred and ninety-nine individuals have been captured of which 110 of *Hybomys univittatus* and 289 of *Praomys* cf. *jacksoni*. Comparison of the average number of individuals captured per trap and per habitat showed a significant effect of the habitat type for both *Hybomys univittatus* and *Praomys* cf. *jacksoni* (Fig.12.5), suggesting a causality between the ecological characteristics of each habitat and the abundance of the species. These results were confirmed by the Mann-Whitney test. For *Hybomys univittatus*, a significant difference between the abundance in the fallow land and the secondary forest was observed ($H = 13.40, p < 0.01; U = 17, p < 0.05$). Between the abundance recorded in the fallow land and in the edge zone ($H = 13.40, p > 0.05; U = 40, p > 0.05$), and between the abundance recorded in the secondary forest habitat and in the edge zone ($H = 13.40, p > 0.05; U = 91, p > 0.05$), no significant differences were detected. For *Praomys* cf. *jacksoni*, the abundance observed varied differently: while no significant difference was recorded for the comparison of the abundance between the fallow land and the secondary forest habitat ($H = 28.19, p > 0.05; U = 96, p > 0.05$), the abundance of the edge zone and the fallow land habitat ($H = 28.19, p < 0.001; U = 14, p < 0.001$) were found significantly different; the same conclusion was made for the comparison of the abundance between the edge zone and the forest habitat ($H = 28.19, p < 0.001; U = 4, p < 0.001$).

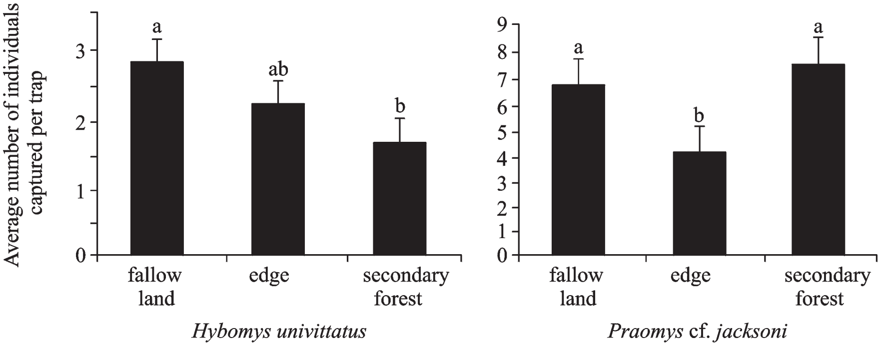

Fig. 12.5 Edge effect on rodent abundance in the Masako Forest Reserve (Democratic Republic of the Congo). Average number of individuals captured per trap in fallow land, edge and secondary forest habitat for *Hybomys univittatus* (left) and *Praomys* cf. *jacksoni* (right) are given together with their standard errors. Significant differences are indicated by different characters.

For both species the expression of the edge effect was clearly different, since the distribution of the number of individuals among the habitat types was not similar. For *Hybomys univittatus*, an intermediate abundance is observed for the edge zone, relative to the adjacent fallow and forest cover types, which corresponds to a classic edge effect of gradual change across a boundary between two different land cover types (Iyongo Waya Mongo 2008; Bogaert et al. in press). According to the capture data, *Hybomys univittatus* preferred the fallow habitat type to the secondary forest habitat, which should be interpreted with caution, since this observation is inconsistent with Dudu (1991), who signalled a higher presence of *Hybomys univittatus* in secondary forest habitats. This contradiction should be verified; it could, however, be explained as a seasonal variability of species abundance due to seasonal changes in precipitation influencing insect and fruit availability in the habitats concerned (Nicolas and Colyn 2003). *Praomys* cf. *jacksoni* seemed to avoid the edge zone, and to prefer either the forest habitat or the fallow habitat, which confirmed the observations of Dudu (1991), who noted likewise quasi equal abundance for this species in both fallow and secondary forest vegetations.

For both species, an undeniable edge effect has been observed, as shown by the significant differences in abundance between the three habitats considered, which corresponds to one type of edge effect as described by Murcia (1995), who also mentioned changes of the physical environment (e.g. increased temperatures in the edge zone) and changes of the interaction between species (e.g. altered predation patterns) as types of edge effects. Comparison of both species emphasizes that edge effects can take different forms in nature; the edge zone constitutes a distinct habitat, preferred or avoided by species, or simply considered as a transitional zone between more and less favourable habitat types. A classification of species according to their type of response to land cover transitions, as presented for rodents in Iyongo Waya Mongo (2008), is therefore indispensable for fully understanding the impact of fragmentation and the concomitant edge effects on diversity. Due to the direct link between land cover pattern, microclimate and diversity, edge effects can be considered as a typical example or application of the pattern/process paradigm, a central theme of landscape ecology, which links landscape pattern to its ecological consequences (Turner 1989; Coulson et al. 1999).

12.4 Implications for landscape management — conclusions

The four case studies previously discussed provide tangible examples of the main causes and ecological impacts of forest fragmentation. Landscape planners should consider this type of information in their landscape-scale design proposals (Brown et al. 2007; Corry et al. 2008).

To mitigate the negative effects of fragmentation on diversity and ecosystem function, landscape corridors could be created in order to compensate for lower diversity due to edge effects or small patches (Farina 2000b). An example is therefore presented in which anthropogenic, scattered landscape elements are spatially rearranged to create corridors between existing, valuable ecosystems.

12.4.1 Creation of a teak (*Tectona grandis* L. f.) corridor network in the Atlantic Department (Benin) to remediate forest isolation

12.4.1.1 Context, data set and methods

Vast areas of forest are destroyed every year in Benin as a consequence of agricultural development or timber extraction (FAO 2005). This deforestation has not spared the natural forests of the municipality of Zè, situated in the oriental part of the Atlantic Department where it has led to considerable patch isolation. Nevertheless, a fraction of the lost forest area has been compensated for by forest plantations, especially teak (*Tectona grandis* L. f.) (Ganglo et al. 1999). In the municipality of Zè, more than 618 patches of teak covering a cumulative area of about 1000 ha have been registered (Toyi 2007). These plantations are primarily considered as wood production units although an important ecological function could also be attributed to these landscape elements if their spatial pattern should be taken into account: a spatial aggregation of the areas of the teak plantations could establish planned continuous (*sensu* Hilty et al. 2006) corridors between the isolated natural forests. Landscape corridors constitute key elements for the conservation and restoration of biodiversity since they offer supplementary habitats and increase habitat connectivity (Paillat and Butet 1994; Hilty et al. 2006). Designing a network of connectivity across a landscape benefits directly humans, as well as biodiversity (Hilty et al. 2006). This consideration of a second function of teak plantations, next to purely wood production, corresponds to the notion of the multiple ecosystem services (Costanza et al. 1997); corridors can provide free ecosystem services (Hilty et al. 2006). In this contribution, different scenarios of corridor creation using teak plantations for the municipality of Zè are analysed in order to illustrate the concept and to evidence its potential for landscape planning based upon ecological and economical grounds.

Five natural forest patches have been chosen in the aforementioned municipality (Fig.12.6): Djigbé-Agué (6°52′48″ N, 2°23′6″ E; 18.57 ha), Djigbé-Agoundji (6°52′03″ N, 2°23′15″ E; 28.11 ha), Ouovinou (6°52′57″ N, 2°24′18″ E; 48.55 ha), Aglangouin (6°52′48″ N, 2°24′54″ E; 129.68 ha) and Sèdjè (6°47′42″ N, 2°24′00″ E; 329.57 ha). These forests are isolated and situated in a zone not appropriate for shifting agriculture; anthropogenic pressure on these forests is consequently negligible, which emphasizes their importance for diversity conservation. The maximum distance between the teak plantations and the forest patches to determine the plantations to be included in the study was set to 5 km. The dislocation of plantations with area superior to 20 ha was avoided. One hundred and fifteen patches of teak were considered in this analysis, with total area equal to 305 ha, which constitutes the upper limit of the total corridor area to be established. A corridor width of 100 m has been chosen.

Five scenarios are considered to define the corridor networks: (A) a minimum number of links between the forests, with minimal cumulative corridor distance; (B) a closed peripherical corridor loop in which every forest is linked to two other forests; (C) the same scenario as B completed with one extra link (the shortest); (D) a corridor network in which every forest is connected to every other forest and in which crossing points are not considered as network nodes; (E) a corridor network in which every forest is connected to every other forest and in which crossing points are considered as network nodes.

To quantify the proposed corridor network architecture, the gamma and alpha index are used (Forman and Godron 1986). The gamma index (γ), measuring connectivity, is the ratio of the number of links in a network (L) to the maximum possible number of links in that network which is determined by the number of network nodes (V) present, i.e.,

$$\gamma = \frac{L}{3(V-2)} \tag{12.5}$$

The gamma index varies from zero (none of the nodes is linked) to 1 (every node is linked to every other possible node). A second network index, the alpha index (α), is a measure of circuitry, the degree to which "circuits" that connect nodes in a network are present. The alpha index is the ratio

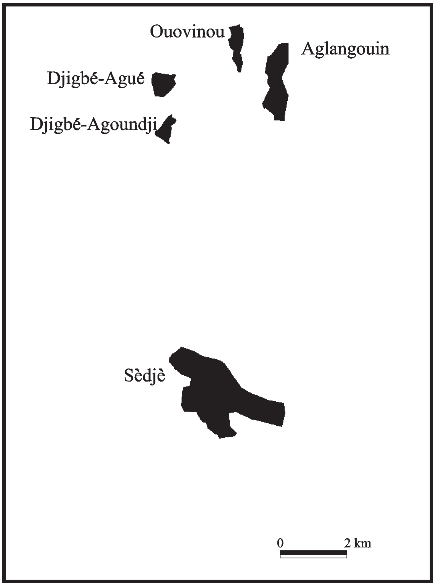

Fig. 12.6 Forest fragments selected in the municipality of Zè. Existing teak plantations could be used for creating ecological corridors in this landscape.

of the actual number of circuits in the network to the maximum number of possible circuits, and is calculated by

$$\alpha = \frac{L - V + 1}{2V - 5} \tag{12.6}$$

and α ranges from zero, for a circuit-less network, to 1 for a network with the maximum possible number of loops present. Together, connectivity and circuitry, indicate the degree of network complexity (Forman and Godron 1986).

12.4.1.2 Results and discussion

Fig.12.7 shows the five diagrams of the corridor networks proposed. In E, the crossing points of the corridors are considered as secondary nodes since, in practice, at these points animals can change of corridor. The secondary nodes are certainly not equivalent to the main nodes of the network, i.e. the forests in the municipality of Zè, representing a larger area and biodiversity. Table 12.3 shows the results of the network complexity analysis for the five scenarios proposed.

Three of the proposed networks (A, B and C) do not utilise all the resources available for corridor creation; scenario A is the most simple to realize, due to its short distance. Nevertheless, this network is not characterized by good connectivity and circuitry values, which undermine its effectiveness in conservation and to enhance interactions between individuals of the isolated forests.

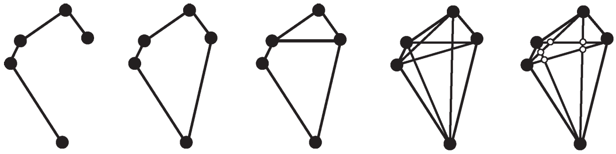

Fig. 12.7 Diagrammatic representation of the five corridor network scenarios considered for the municipality of Zè. Black filled circles represent the five forests to be connected by the network. Small open circles are secondary nodes situated at the crossing of corridors.

Table 12.3 Network complexity statistics of the five corridor network scenarios. A corridor width of 100 m is assumed. D is the total network length. R is the ratio of the network area to the cumulative area of teak plantations (305 ha). Network complexity is determined by the number of links (L) and the number of nodes (V). γ quantifies network connectivity; α quantifies network circuitry.

	Network scenario				
	A	B	C	D	E
D(km)	13.05	21.60	24.48	39.18	39.18
R	0.43	0.71	0.80	1.28	1.28
L	4	5	6	10	18
V	5	5	5	5	10
γ	0.44	0.56	0.67	1.00	0.75
α	0.00	0.20	0.40	1.00	0.60

Scenarios B and C are characterized by higher values for γ and α, which indicates that they should lead to better results with regard to conservation. Scenario C could be preferred over B because its connectivity is higher and more circuits are available for the species using the corridor network. Its relatively short length is expected to provide increased connectivity than longer corridors (Hilty et al. 2006), a characteristic not quantified by γ and α. Scenario D is to be preferred based upon γ and α, but cannot be realized *in situ*, since the resources needed exceed the total teak area available for spatial rearrangement by 28%. When the crossing points of the corridors are considered as secondary nodes, the connectivity and circuitry indices indicate lower values, due to a potential number of links that could theoretically still be created with these secondary nodes. It should be noted that corridor width, in this study set to 100 m, remains subject to debate (Hilty et al. 2006) and should be considered with regard to the species considered. Nevertheless, the chosen value lies inside the range described in other studies (Hilty et al. 2006).

Relating the composition and structure of landscapes to the ecosystems they provide is a challenge for landscape ecologists (Crow 2008). Connectivity is one of the landscape characteristics that can compensate for diversity loss due to edge effects, and that can show that a landscape contains a higher species number than predicted by island biogeography theory (Farina 2000b). Maintaining or restoring landscape connectivity is currently a central concern in ecology and land conservation planning (Saura and Torné 2009).

For the municipality of Zè, five corridor networks have been analyzed to link five existing forests. Three of the scenarios can be realized, of which one should be preferred based upon its architecture and length. This exercise underlines the potential of landscape planning in biodiversity management at the landscape scale. As an application of the "pattern/process paradigm", landscape configuration could be used to have a beneficial effect on landscape biodiversity, a concept which has led recently to the development of a software package (Conefor Sensinode 2.2) quantifying the importance of habitat patches for landscape connectivity (Saura and Torné 2009).

By rearranging the existing plantations, a network can be created that mitigates the negative effects of forest fragmentation such as population isolation and edge effects. In this way, timber production could contribute to a better functioning of the ecosystems by linking them; in this way, economical and ecological objectives are integrated. Nevertheless, this type of theoretical consideration should be validated by long-term experiments; the empirical understanding of corridor effects on community structure and diversity is still in its infancy (Haddad and Tewksbury 2006).

12.4.2 Summary and concluding remarks

In order to enable landscape managers to manage fragmented landscapes adequately, the causes and ecological consequences of habitat fragmentation have to be fully understood. Field data should guide landscape ecologists in this more comprehensive understanding and their interpretation should constitute a main occupation of landscape ecologists (Chen et al. 2008).

In this chapter, two case studies are presented illustrating the main drivers of fragmentation in the Democratic Republic of the Congo (Oriental Province) and in North Benin. Anthropogenic pressure and population density caused forest degradation leading to the dissipation of forest habitats. Two case studies quantifying the impacts of fragmentation on biodiversity are discussed; decreasing levels of forest diversity in fragmented forests were detected in the Tanda Region of Ivory Coast; edge effects on two rodent species were observed in Kisangani (Democratic Republic of the Congo). The fifth case study considered the possibility to remediate fragmented landscapes by a spatial planning of teak plantations in the Atlantic Department in Benin. All five studies were based on field data analysis and provide additional clues on the process of forest fragmentation, observed in different forms and associated with divergent consequences.

Although these studies give a limited perspective on the possible causes and consequences of fragmentation, they contribute to the ongoing debate on landscape management and conservation at multiple scales. A more comprehensive view on fragmentation as well as more efficient landscape management plans are needed, avoiding dispersion of valuable natural resources in the future and mitigating the impact of less favourable spatial patterns on diversity.

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Part IV
Practicing Sustainable Forest Landscape Management

Chapter 13 Application of Landscape and Habitat Suitability Models to Conservation: The Hoosier National Forest Land-management Plan

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Abstract

We demonstrate an approach to integrated land-management planning and quantify differences in vegetation and avian habitat conditions among 5 management alternatives as part of the Hoosier National Forest planning process. The alternatives differed in terms of the type, extent, magnitude, frequency, and location of management activities. We modeled ecological processes of disturbance (e.g. tree harvest, prescribed fire, wildfire, windthrow) and succession using LANDIS, a spatially explicit landscape decision-support model, and applied habitat suitability models for six species of birds to the output from that model. In this way, we linked avian habitat suitability models to spatially explicit vegetation change models that include ecological processes affecting vegetation composition, horizontal and vertical structure, and configuration. The detailed and synthetic nature of our approach provides a framework and structure that (1) is readily conveyed to multiple constituencies, (2) is based on explicitly stated assumptions and relationships, (3) provides a basis for testing, refinement, and extension to other forest commodities and amenities, and (4) provides a way to consider cumulative effects of multiple forest attributes at multiple spatial and temporal scales.

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Keywords
Decision support, dynamic landscape model, forest planning, habitat suitability models, LANDIS, multi-resource evaluation, *Scolopax minor*, *Dendroica cerulea*, *Bonasa umbellus*, *Hylocichla mustelina*, *Helmitheros vermivorus*, *Icteria virens*.

13.1 Introduction

A common goal in National Forest planning is to describe relationships of management actions, vegetation conditions, and wildlife habitat conditions for large landscapes. Inherent in most planning efforts are concepts of landscape ecology (e.g. ecological processes of disturbance and succession) as well as the implications of those processes on the composition, horizontal and vertical structure, and configuration of vegetation and wildlife habitat. Problem definition and priority setting are critical elements of planning, especially when multiple management objectives are desired, when competition or tradeoffs among management objectives exists, or when management objectives are unequally weighted (Lindenmayer et al. 2008). Because forest planning often involves many integrated objectives and multiple wildlife species, some modeling approaches (e.g. optimization models, Lu and Buongiorno 1993) may be difficult if not impossible to implement (Thompson and Millspaugh 2009). When multiple, integrated, or adaptive objectives exist, the conceptual model used to characterize and simulate landscape change should provide the spatial and temporal information needed for management decisions (Lindenmayer et al. 2008). Thus, for planning purposes an ideal modeling approach would consider broad-scale landscape dynamics while retaining the fine-scale resolution needed to quantify changes in wildlife habitat (Zollner et al. 2008; Noon et al. 2009).

Our objectives are to demonstrate an approach to integrated land-management planning and to quantify differences in vegetation and avian habitat conditions among management alternatives using the Hoosier National Forest planning process as both a vehicle and application of this approach. We build upon previous planning efforts for the Hoosier National Forest lands that included the evaluation of multiple management alternatives on vegetation conditions (Gustafson and Crow 1994) and salamander habitat (Gustafson et al. 2001). As in the previous planning efforts, the management alternatives differ in terms of the type, extent, magnitude, frequency, and location of management activities. We modeled ecological processes of disturbance and succession using a spatially explicit landscape decision-support model, and applied habitat suitability models for six species of birds to the output from that model. In this way, we linked avian habitat suitability models to vegetation change models that include ecological processes affecting vegetation composition, vertical and horizontal structure, and configuration of vegetation patches.

13.1.1 Overview of the Hoosier National Forest planning process

The Hoosier National Forest (HNF) is located in southern Indiana, USA and consists of four administrative units totaling approximately 261,000 ha. Only 31 percent (approximately 81,000 ha) of the land within the administrative unit boundaries is HNF, the remainder is privately owned (Fig. 13.1). This region was subject to intensive forest harvest from 1870 to 1910, shifting the tree species composition of a maple-beech and oak-hickory forest to a primarily oak-hickory forest. This was followed by a period of settlement and conversion to agricultural land uses that persisted into the early 1930s. At present, 96 percent of HNF lands are characterized as second-growth forest, with 75 percent of the total forest area older than 50 years of age (Woodall et al. 2007). The fragmented nature of the HNF, coupled with public opposition to tree harvest over the past several decades, has strongly influenced current land-management issues (Welch et al. 2001).

The HNF Planning Team, in conjunction with the public, identified watershed health, ecosystem sustainability, and recreation management as issues to address in the planning process. The primary means for maintaining watershed health and ecosystem sustainability on the HNF is vegetation management, typically through the application (or absence) of prescribed fire or tree harvest. Because vegetation management also affects habitat for bird species, there was strong interest in monitoring changes in habitat for a diverse suite of bird species.

The HNF Planning Team considered five forest management alternatives, each of which contained different tree harvest procedures (e.g. even-aged and uneven-aged techniques), amounts and locations of tree harvest and prescribed burning treatments, and types of recreation opportunities (Table 13.1). A detailed description of the forest management alternatives is provided in Rittenhouse (2008) and US Department of Agriculture Forest Service (2006). Alternative 1, referred to as the No Action alternative in the Draft Environmental Impact Statement (US Department of Agriculture Forest Service 2006), represented continuation of the forest management practices that were implemented with the 1985 Forest Plan as amended. For all management alternatives except Alternative 2, tree harvest and prescribed fire were used to maintain biological diversity and promote oak-hickory regeneration within specified management units. Alternative 2 emphasized natural processes and limited vegetation management. After reviewing the avian habitat suitability model output from initial model runs, the HNF Planning Team created a 5,260-ha focal area within the Tell City District (southern-most administrative unit) (Fig. 13.1). The majority of even-aged management was conducted within the focal area to provide habitat for bird species such as ruffed grouse

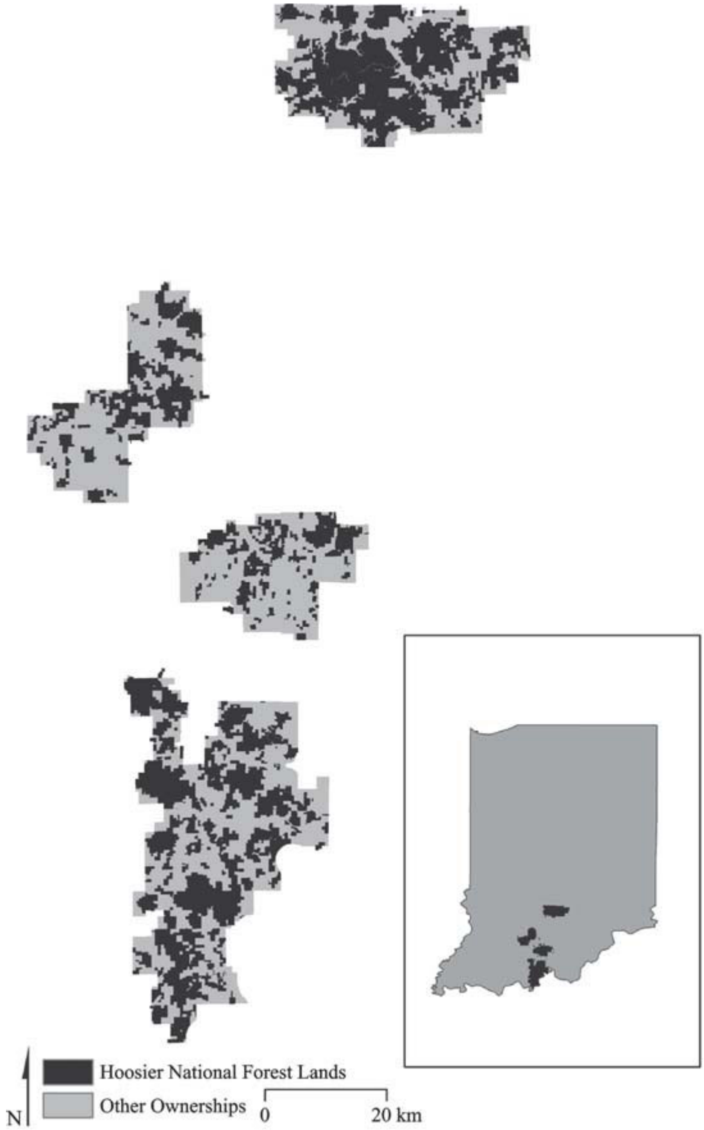


Fig. 13.1 The four Hoosier National Forest administrative units in southern Indiana. National Forest ownership is approximately 81,000 ha, or 31 percent of the total 261,000 ha within the administrative units. The majority of the remaining area within the administrative unit boundaries is privately owned.

and yellow-breasted chat that depend on early successional forest (see Table 13.2 for scientific names of bird species). Alternatives 3, 4 and 5 include the focal area.

Table 13.1 Approximate area in ha (percent) treated by management practices each decade for the 150-year planning horizon for 5 management alternatives on the Hoosier National Forest, Indiana. Alternative 5 differs from Alternative 1 only in concentrating most of the even-aged harvest in a 5,260 ha block designated for improved habitat for early successional bird species.

Altern-ative	Emphasis	Uneven-aged harvest	Even-aged harvest	Prescribed fire
1	Ecosystem sustainability, wilderness areas, and recreation areas	1,493 (1.8)	1,157 (1.4)	8,095 (10)
2	Natural processes and old growth	0	0	0
3	Diversity of forest age classes, increase recreational opportunities, and harvest focal area	1,643 (2.0)	2,294 (2.8)	20,235 (25)
4	Native hardwood restoration, early successional habitat, and harvest focal area	2,088 (2.6)	3,893 (4.8)	40,470 (50)
5	Alternative 1 with harvest focal area	1,493 (1.8)	1,157 (1.4)	8,095 (10)

13.2 Methods

We developed an approach to land-management planning on the Hoosier National Forest that contained desirable features from a large-scale, landscape perspective while retaining the fine-scale information useful for evaluating avian habitat suitability. The following sections describe modeling spatial and temporal trends of vegetation change and linking that change to avian habitat suitability.

13.2.1 Modeling vegetation change using LANDIS

We simulated spatial and temporal trends of vegetation change using LANDIS (version 3.6), a spatially explicit, landscape-scale, decision-support tool that models vegetation growth, succession, and response to disturbance by tree harvest, wind, and fire (He et al. 2003; He 2009). In LANDIS, a landscape is organized as a raster array of cells that represent sites in the landscape. Cell size in LANDIS is user-defined, and we used a 10m by 10m cell size (0.01 ha) because in this ecosystem it approximated the size of a canopy gap created by the death or harvest of a mature tree. Each cell contains a matrix of vegetation information such as the tree species (or species groups) present or absent in the cell and the 10-year age class of each species cohort.

We simulated four spatial processes (fire, windthrow, harvesting, seed dispersal) and four temporal processes (succession, regeneration, age-dependent mortality, sequential patterning of disturbance events) that affect the projected species composition and age structure of individual cells and, in the aggregate, of the landscape. To do this, we first calibrated the LANDIS regeneration and succession algorithms for 14 tree species or groups of similar species common to the HNF using Forest Inventory and Analysis (FIA) data for southern Indiana (see Rittenhouse 2008 for details): Eastern red cedar (J*uniperus virginiana* L.), pines (*Pinus echinata* Mill., *P. virginiana* Mill., and *P. strobus* L.), sugar maple (*Acer saccharum* Marsh.), red maple (*Acer rubrum* L), hickories (*Carya* spp.), American beech (*Fagus grandifolia* Ehrh.), ash (*Fraxinus americana* L. and *F. pennsylvanica* Marsh.), yellow poplar (*Liriodendron tulipifera* L.), black cherry (*Prunus serotina* Ehrh.), white oak (*Quercus alba* L.), chestnut oak (*Q. prinus* L.), red oaks (*Q. rubr*a L., *Q. velutina* Lam., and *Q. coccinea* Muenchh.), pin oaks (*Q. ellipsoidalis* E. J. Hill and *Q. imbricaria* Michx.), and elms (*Ulmus* spp.). We made small adjustments to the regeneration coefficients to make long-term shifts in species composition consistent with expected changes in species composition based on expert opinion from regional managers.

Next, we established initial vegetation conditions (tree age and species) for public and private lands within the HNF administrative unit boundary from FIA data, the HNF's inventory database, land-use and land-cover data, and Indiana GAP data (http://gapanalysis.nbii.gov). We estimated the expected number of trees by age class (seedling, age 1-10 years; sapling, age 11-40 years; pole, age 41-60 years; and mature, age $>$ 60 years) for each cell in a given stand. We used FIA data to develop observed species frequency distributions by forest cover type, age class, and ecological land type, and we assigned tree species to each cell in a specific stand by random draw from the appropriate frequency distribution.

We lacked spatially explicit maps of forest cover type, age class data, and ownership boundaries for forest stands on private lands within the HNF administrative units. Therefore, we utilized the digitized land use and land cover data created by Pangea Information Technologies (2003), the Indiana GAP data (http://gapanalysis.nbii.gov), and satellite data classified by the National Agricultural Statistics Service (2008) to map locations of nonforest, coniferous forest, upland deciduous forest, mixed forest, bottomland forest types and water for private lands. We assigned an age class and forest cover type based on the frequency distribution of forest age classes and forest cover types from FIA data for southern Indiana. We also created an artificial private land ownership boundary layer with ownership sizes approximating the size distribution of forested land parcels reported by Birch (1996). This layer was used during LANDIS simulations to identify management units (e.g. stands) for private lands where stand boundary maps were unavailable. We combined our derived maps of initial conditions for private lands with corresponding maps for the HNF and used them together as initial conditions for LANDIS scenario analyses for each of the four HNF administrative units (Fig. 13.1).

We modeled tree harvests to mimic the proposed harvest actions for each Forest Plan alternative (Table 13.1) (US Department of Agriculture Forest

Service 2006) using the methodology described by Gustafson et al. (2000). The HNF designated Management Areas that divide the forest into thematic zones based on suitable management activities (e.g. riparian buffers vs. wilderness vs. timber management vs. habitat for a designated bird species). We used the Harvest module for LANDIS (Gustafson et al. 2000), which allows tree harvest activity to vary within each management area, to model differences in management practices among management areas as specified in each Forest Plan alternative.

LANDIS output included maps of tree species composition and dominance, tree age classes, fire disturbance, wind disturbance, and tree harvest disturbance in 10-year increments for each cell in the landscape. We expected the forest plan alternatives would differentially affect the spatial and temporal distribution of forest conditions based on the differences among the alternatives in the type, frequency, and extent of disturbances. To capture those differences, we summarized forest and landscape attributes for spatially defined groups of cells at different spatial scales (e.g. administrative units, management areas, or the entire HNF). Attributes included tree age class distribution, tree species composition, contiguous core forest area and edge density.

13.2.2 Linking vegetation change to avian habitat suitability

We used Landscape HSImodels version 2.0 (Dijak et al. 2007) to evaluate breeding habitat suitability or year-long habitat suitability for 6 bird species selected by the HNF Planning Team (Table 13.2). Landscape HSImodels is a Microsoft Windows-based software program that uses suitability indices (SI) to assign habitat quality across large landscapes for individual species (Larson et al. 2003; Dijak et al. 2007; Dijak and Rittenhouse 2009). Habitat suitability is described by an empirical or assumed relationship between habitat quality and resource attributes on a relative scale that ranges from 0 (unsuitable habitat) to 1 (highly suitable habitat) (US Fish and Wildlife Service 1980, 1981). We developed the suitability indices with specific objectives in mind (Rittenhouse et al. 2007). First, the SIs addressed habitat requirements for reproduction or survival and they were supported by empirical data, published literature, or expert opinion. Second, all SIs were estimated from available GIS (geographic information system) layers of vegetation (and landscape) structure and composition. Third, all required GIS layers of vegetation information were derived from LANDIS projections. Thus, we could apply the habitat suitability index (HSI) models to modeled future vegetation conditions and compare landscapes in terms of future habitat conditions.

The avian habitat suitability models use LANDIS output as well as ecological land type and land-cover type information (Table 13.2). We used ecological land types (ELT) derived from 10m Digital Elevation Model (DEM) layers by Guafon Sho (Purdue University). The ELT coding followed Van Kley et al.

Table 13.2 Description of habitat suitability index models for bird species used to evaluate proposed management alternatives on the Hoosier National Forest, Indiana.

Species	Life requisite	Habitat requisite	Model parameters and implementation	HSI equation
American woodcock (*Scolopax minor*)	Nest sites, roost sites, and food.	Early- and mid-successional forest.	SI_1: Tree species SI_2: Tree age by ELT	$HSI = \sqrt[2]{\left(\max\left(\left(SI_1 \times SI_2\right), SI_4\right) \times SI_5\right)}$
	Display sites, roost sites, and food.	Open habitat.	SI_3: Land cover type SI_4: Land cover type	
		Interspersion of life requisites.	SI_5: Moving window analysis of SI_2 and SI_4	
Cerulean warbler (*Dendroica cerulean*)	Nest sites and food.	Mature deciduous forest. Large forest patches.	SI_1: Tree species SI_2: Tree age by ELT SI_3: Patch size algorithm	$HSI = \sqrt[3]{SI_1 \times SI_2 \times SI_3}$
Ruffed grouse (*Bonasa umbellus*)	Food.	Hard mast.	SI_1: Model of tree age, tree species, and land type	$HSI = \left(\sqrt{\max\left(SI_1, \sqrt{SI_2 \times SI_3}\right) \times SI_4}\right) \times SI_5$
	Nest sites, food, and cover.	Dense forest regen. Large habitat patches. Interspersion of life requisites. Large forested area.	SI_2: Tree age by ELT SI_3: Patch size algorithm SI_4: Moving window analysis of SI_1, SI_2, and SI_3 SI_5: Patch size algorithm	

Continued

Species	Life requisite	Habitat requisite	Model parameters and implementation	HSI equation
Wood thrush (*Hylocichla mustelina*)	Nest sites and food.	Deciduous forest.	SI_1: Tree species SI_2: Tree age by ELT	$HSI = SI_1 \times \left(\sqrt[3]{SI_2 \times SI_3 \times SI_4}\right)$
	Post-fledging habitat.	Large forest patch.	SI_3: Patch size algorithm	
		Early successional forest.	SI_4: Tree age by ELT	
		Interspersion of life requisites.	SI_5: Moving window analysis of SI_4	
Worm-eating warbler (*Helmitheros vermivorus*)	Nest sites and food.	Deciduous forest.	SI_1: Tree species SI_2: Tree age by ELT	$HSI = \left(\sqrt[3]{SI_1 \times SI_2 \times SI_3}\right) \times SI_4$
		Large forest patch.	SI_3: Patch size algorithm	
		Fire avoidance.	SI_4: Fire history	
Yellow-breasted chat (*Icteria virens*)	Nest sites and food.	Early successional forest and old fields.	SI_1: Tree age by ELT	$HSI = \left(\sqrt[2]{SI_1 \times SI_2}\right) \times SI_3$
		Large habitat patch.	SI_2: Patch size algorithm	
		Edge sensitivity.	SI_3: Moving window analysis of SI_2	

(1994) and grouped types by slope, aspect, and relative moisture. ELT classes generally correspond to north and east (cool and mesic) slopes, south and west (warm and dry) slopes, wide ridges or upland flats, narrow ridges, and mesic bottoms. We classified land-cover type using the HNF forest type codes (for public lands) and the land-use and land-cover data described above for private lands. We collapsed the HNF forest type map and the public land-cover map into 6 general land-cover types used in the HSI models: 1) forest, 2) croplands, 3) grasslands, 4) water, 5) urban areas, and 6) roads.

Rittenhouse et al. (2007) provided a thorough discussion of habitat variables used in the development of the habitat suitability models, including literature citations supporting suitability relationships of each species. The primary input data (i.e. resource attributes) for the SIs included raster maps of tree species, tree age, ecological land type, land-cover type, and fire history. Landscape HSImodels contains functions to compute patch size, edge effects, distance to resource, and composition of habitat. Thus, the suitability value of any given cell on the landscape considered attributes of that cell as well as the attributes of the surrounding cells in the landscape (Table 13.2). Landscape HSImodels computes a single Habitat Suitability Index value representing the overall habitat suitability for each species, for each cell.

We applied the species-specific habitat suitability models to raster maps from LANDIS output at four time periods for each management alternative: initial conditions, year 10, year 50, and year 150. We followed traditional habitat evaluation procedures and used the habitat unit as our metric for the amount of suitable habitat. We defined a habitat unit as the HSI value of an individual cell multiplied by the cell's area (0.01 ha). For each bird species we summarized HSI values for each 0.01 ha site across the entire HNF landscape and grouped habitat units by five HSI categories (0, 0.01-0.24, 0.25-0.49, 0.50-0.74, and 0.75-1.00). For convenience, we refer to habitat units with HSI values >0.01 as suitable habitat, and HSI values of 0.75-1.00 as high quality habitat. The HNF Planning Team assumed habitat suitability was synonymous with population viability; therefore we did not assess population viability (see Section 13.4.3 for discussion of this issue).

13.3 Results

We simulated changes in vegetation conditions and avian habitat suitability for five management alternatives. The following sections detail spatial and temporal changes in forest age class distribution, tree species composition, and avian habitat suitability. The primary emphasis for planning purposes was to summarize effects at short, intermediate, and long periods of plan implementation for the HNF. Therefore, we typically present results only for HNF ownership at simulation year 0, 10, 50, and 150 for each plan alternative.

13.3.1 Simulated changes in vegetation conditions

The five management alternatives differed in the type and frequency of disturbance due to tree harvest and prescribed fire, resulting in differences in the temporal and spatial distribution of forest by age class (Figs. 13.2, and 13.3), landscape attributes of contiguous core forest area (Fig. 13.4) and edge density (Fig. 13.5), and the temporal and spatial distribution of tree species composition (Figs. 13.6, and 13.7). The primary emphasis on planning purposes was to summarize effects at short, intermediate, and long periods of plan implementation for the HNF. Therefore, we typically present results only for HNF ownership at simulation year 0, 10, 50, and 150 for each plan alternative.

13.3.1.1 Spatial and temporal changes in forest age class distribution

The initial forest age class distribution was the same for all management alternatives. At year 0 of the simulation, less than 1 percent of the initial HNF landscape was classified in the seedling age class (1-10 years old), 18 percent was in the sapling age class (11-40 years old), 15 percent was in the pole age class (41-60 years old), and two thirds of the HNF was in the mature age class (>60 years old) (Fig. 13.2). The relative proportions of each age class shifted over time in response to disturbance by tree harvest, fire, and wind (Table 13.1, Fig. 13.2).

Three patterns stand out in the comparison of forest age class proportions over time for each alternative (Figs. 13.2, and 13.3). The first pattern, a "V" shape in the age class distribution, was partially an artifact of the way we developed the initial landscape conditions and the way LANDIS implemented age-dependent mortality, wind disturbance, and mortality due to epidemic Dutch elm disease in the first decades of the projection. The size of this effect was evident in Alternative 2, which showed a 5-8 percent increase in the seedling size class in the first decade (Fig. 13.2). The second factor contributing to the "V" shape was the implementation of harvest at the prescribed levels. The HNF is predominately old, relatively undisturbed, and undergoing transition from oak to maple. Thus, any harvest changes current and near future vegetation structure and composition. The seedling age class increased by 3-7 percent with the magnitude of the increase corresponding to the differences in amount of harvest among management alternatives. As a result of these events, in the first few decades there were rapid changes in the seedling age class that were carried forward into the older age classes in later decades. We expected this shift in age class distribution, just not as abruptly as the simulation suggests.

The second pattern occurred 90-100 years from plan implementation when age class distribution as a proportion of area equilibrated (Fig. 13.2). From years 100 to 150 of the simulation the proportion of the landscape in the 4 age classes remained stable within Alternatives 1, 3, 4, and 5 (the alternatives implementing tree harvest and prescribed fire). By year 150, the combined to-

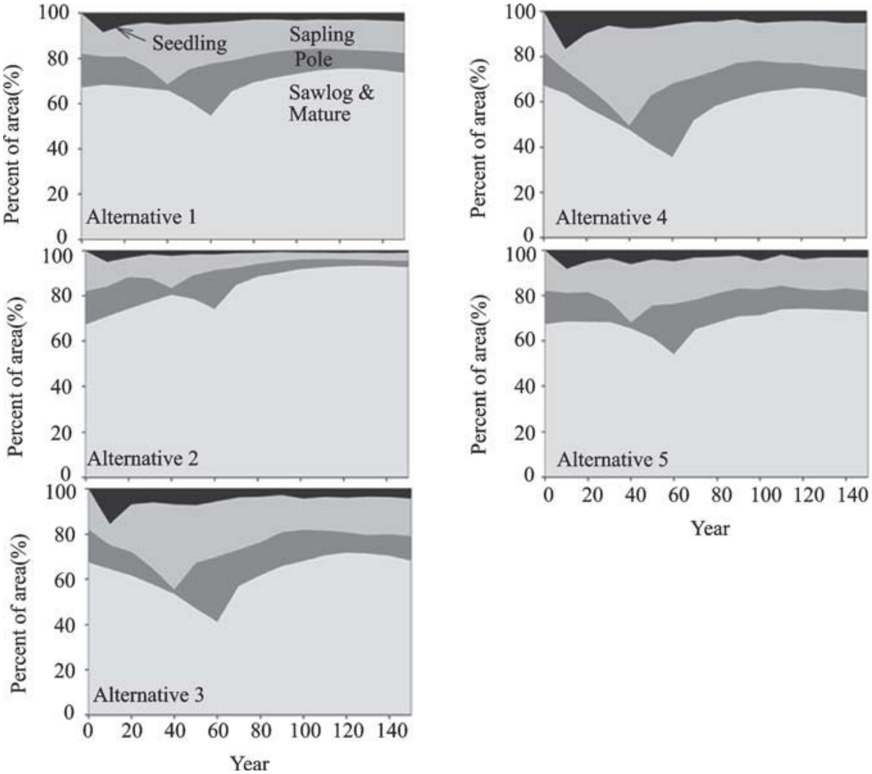


Fig. 13.2 Forest area by age class for 5 management alternatives on the Hoosier National Forest, Indiana. See Table 13.1 for details of the management practices associated with each alternative. Age classes are seedling (1-10 years old); sapling (11-40 years old); pole (41-60 years old); and mature (>60 years old).

tal of the seedling and sapling age classes as a proportion of the total area declined (relative to initial conditions at year 0) for Alternative 1 (1 percent decline), Alternative 2 (14 percent decline), and Alternative 5 (1 percent decline), whereas Alternative 3 (2 percent increase) and Alternative 4 (7 percent increase) increased the area in the seedling and sapling age classes compared to initial conditions.

The third pattern was evident in the spatial arrangement of forest age classes beginning in year 10 and continuing to year 150 of the simulation (Fig. 13.3). Even-aged harvest in Alternatives 1, 3, 4, and 5, produced even-aged patches of regeneration ranging in size from 2 ha to 16 ha. Uneven-aged harvest produced many small, similar age patches on the landscape (group selection) and stippled areas of intermixed age classes (single-tree selection). Alternative 2 resulted in a homogenous landscape dominated by the oldest age class, although scattered pockets of younger forest were maintained by a combination of fire disturbance, wind disturbance, and gap-scale replacement

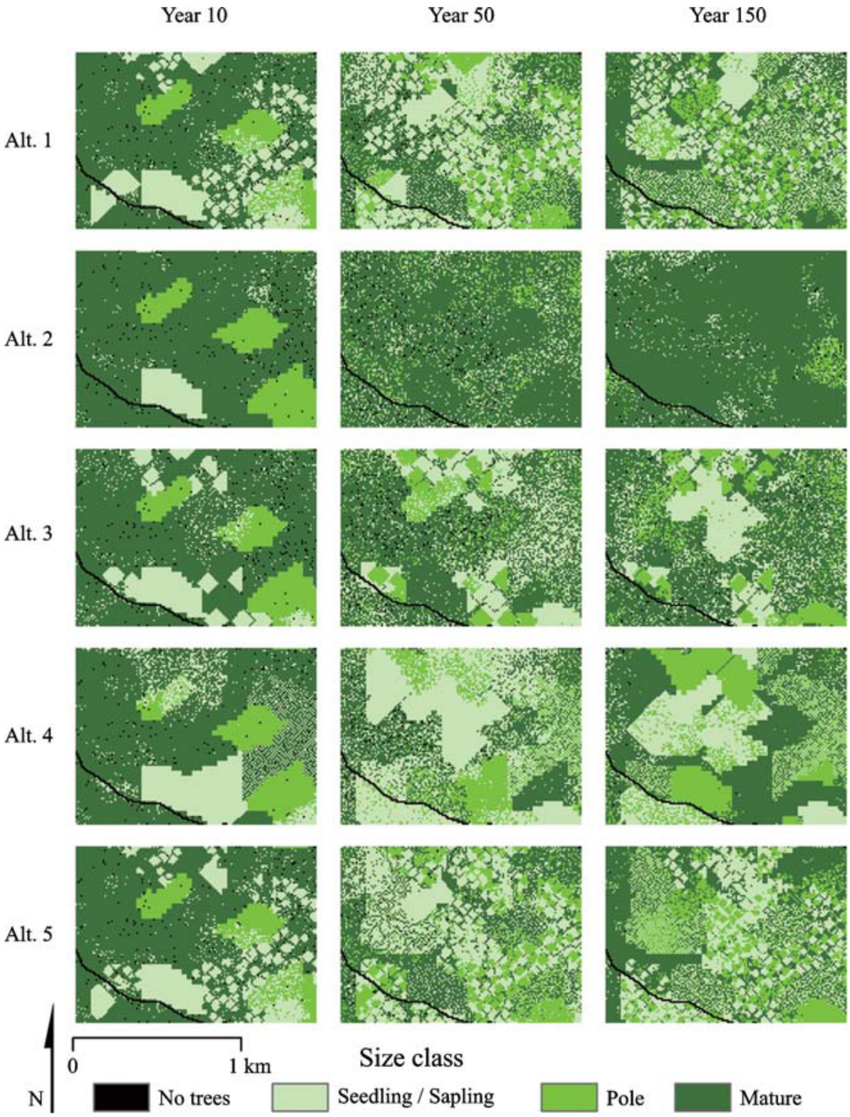


Fig. 13.3 Forest age class maps by management alternative at year 10, 50, and 150 of the plan horizon. The portion of the Hoosier National Forest displayed is approximately 150 ha. Age classes are seedling (1-10 years old); sapling (11-40 years old); pole (41-60 years old); and mature (>60 years old).

of senescent trees. Core area (Fig. 13.4) and edge density (Fig. 13.5) further document the spatial differences among alternatives in the effects of forest regeneration. When projected core and edge values equilibrated approximately 100 years into the projection, Alternative 2 created about three times as much core area and about half the edge density of the other alternatives. The other

4 alternatives were clustered in their estimated edge density and core area.

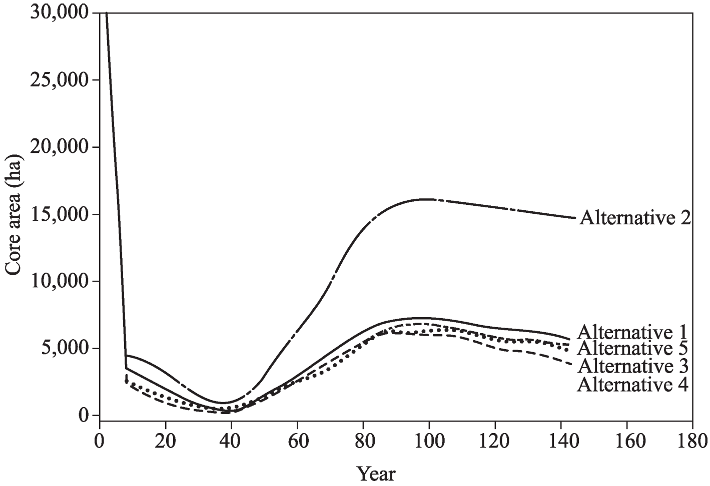


Fig. 13.4 Core area of forest in the pole and sawtimber age classes that was at least 60 m from an edge with a younger age class or nonforest on the Hoosier National Forest, Indiana. Pole and mature age classes correspond to forest ages of 41-60 years and >60 years, respectively. Computations were based on a 0.01 ha cell size, so any 0.01 ha or larger opening created by mortality or tree harvest was a breach in the core area. The minimum size opening that is ecologically relevant as a breach of core area can differ with avian habitat preferences and can be recomputed for other minimum opening sizes.

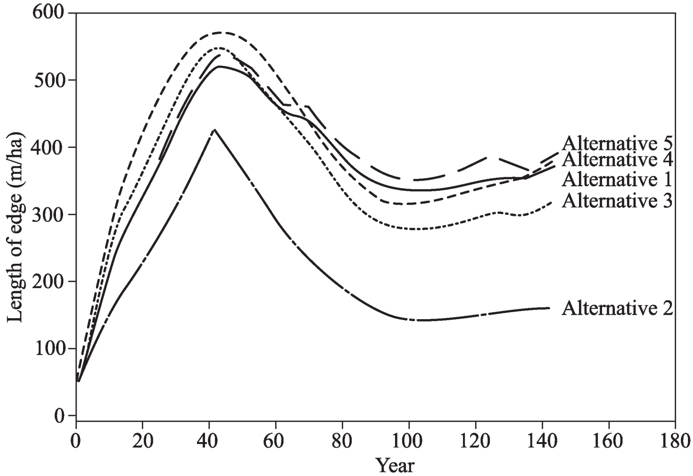


Fig. 13.5 Edge density (m per ha) between forest in the pole and older age classes (i.e. >40 years of age) with a younger forest and nonforest on the Hoosier National Forest, Indiana. Computations were based on 0.01 ha pixel size.

13.3.1.2 Spatial and temporal changes in tree species composition

The HNF planning team was particularly interested in the proportion of oaks relative to maples and other mesic species; therefore, we summarized temporal (Fig. 13.6) and spatial patterns (Fig. 13.7) in tree species composition for each alternative in terms of white oak, maple, and red oak groups. The initial tree species composition was the same for all management alternatives. At year 0, oaks were dominant (i.e., oldest tree per cell) on 42 percent of the HNF forested landscape (white and post oak comprised 19 percent; red oak group 18 percent; and chestnut oak 5 percent), followed by hickories (14 percent), and maples (12 percent). Each of the remaining species or species groups was dominant on less than 10 percent of the initial landscape.

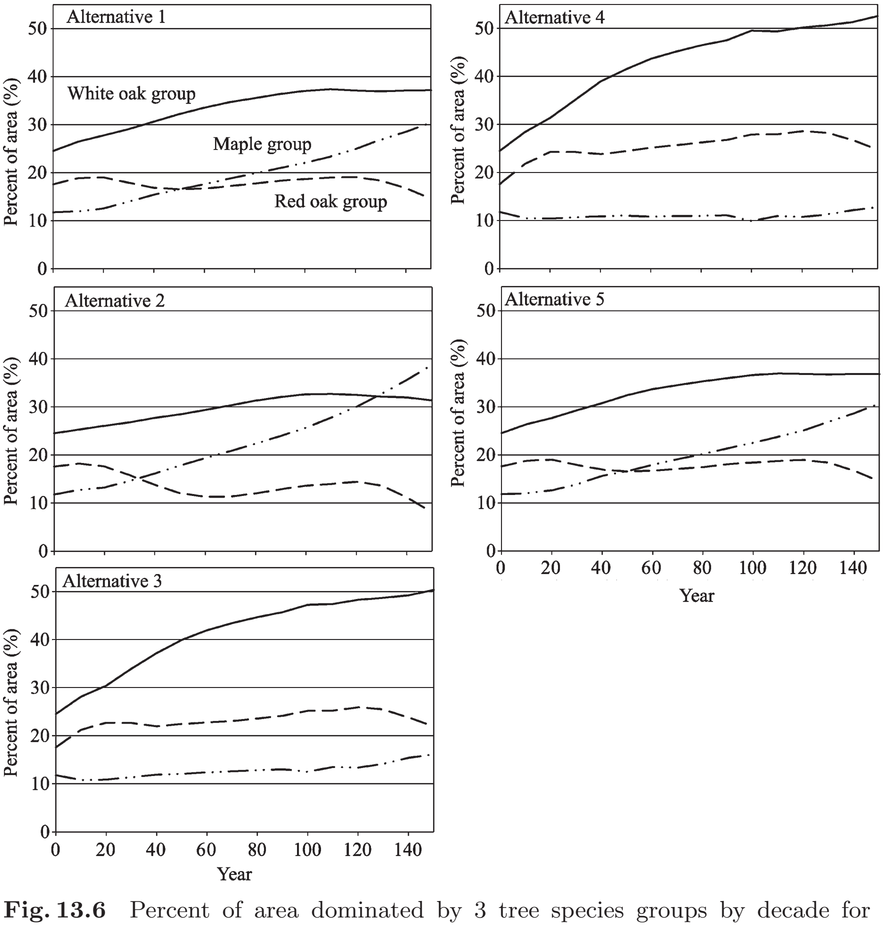


Fig. 13.6 Percent of area dominated by 3 tree species groups by decade for 5 management alternatives on the Hoosier National Forest, Indiana. Species groups were: red oaks (northern red, black and scarlet oaks), white oaks (white and chestnut oak), and maple (sugar and red maple).

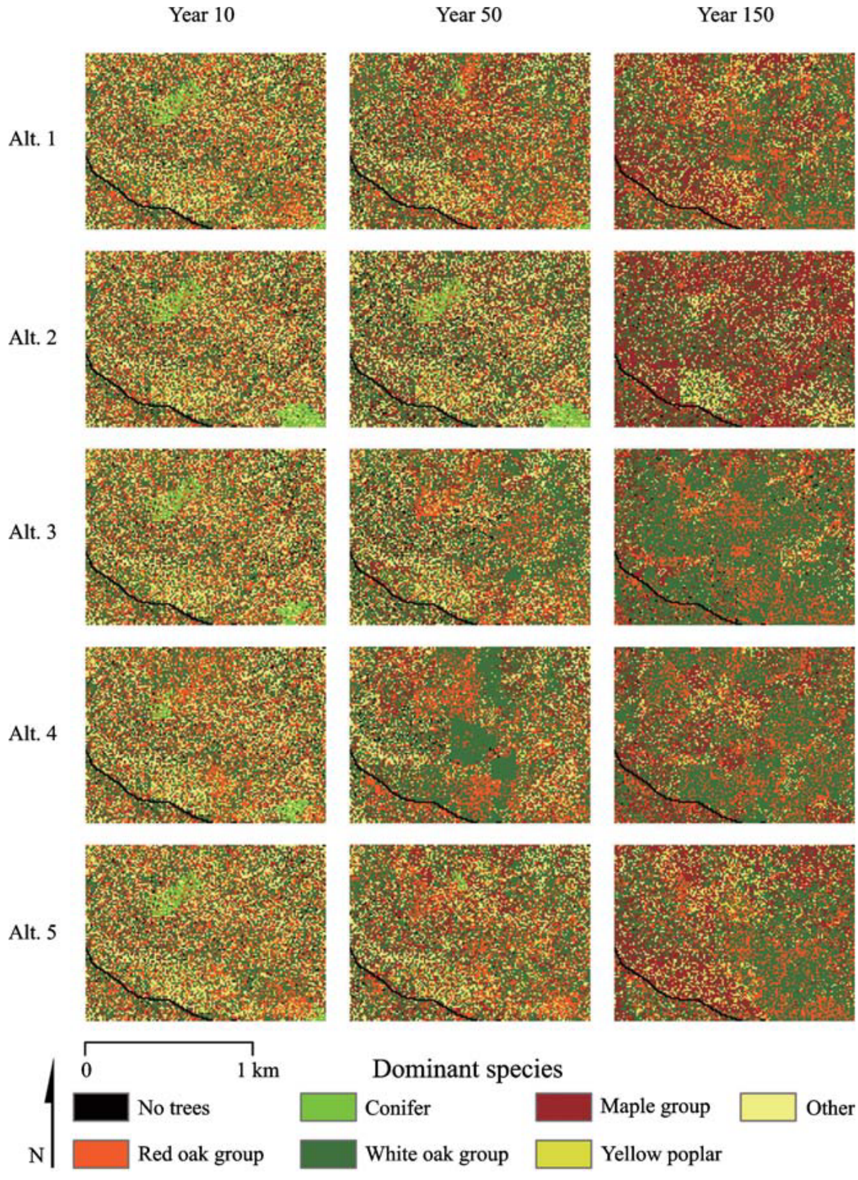


Fig. 13.7 Dominant tree species composition maps for 5 management alternatives at year 10, 50, and 150 for a 150-ha portion of the Hoosier National Forest, Indiana.

Over the 150-year simulation of vegetation change, Alternative 2 realized the greatest increase in maple dominance, from 12 to 39 percent of the forest in 150 years (Fig. 13.6). Under Alternative 4, the area of forest dominated by the maple group remained nearly constant over the 150-yr simulation while the area dominated by the red oak group increased to 25 percent and the area dominated by the white oak group increased to 52 percent (Fig. 13.6). Alternatives 3 and 4 reached the highest dominance by white oaks at 50 and 52 percent of forest area, respectively and were the only alternatives where the red oak group was dominant over a greater area than the maple group (Fig. 13.6). Alternative 5, which mirrored Alternative 1 with the exception of the added focal area to concentrate tree harvest activities, had the same species composition as Alternative 1 (Fig. 13.6).

As for forest age class, the spatial pattern in tree species composition varied by management alternative (Fig. 13.7). Even-aged and uneven-aged harvests produced patches of forest that were dominated by the red and white oak groups. By contrast, areas without harvest had higher dominance by maples.

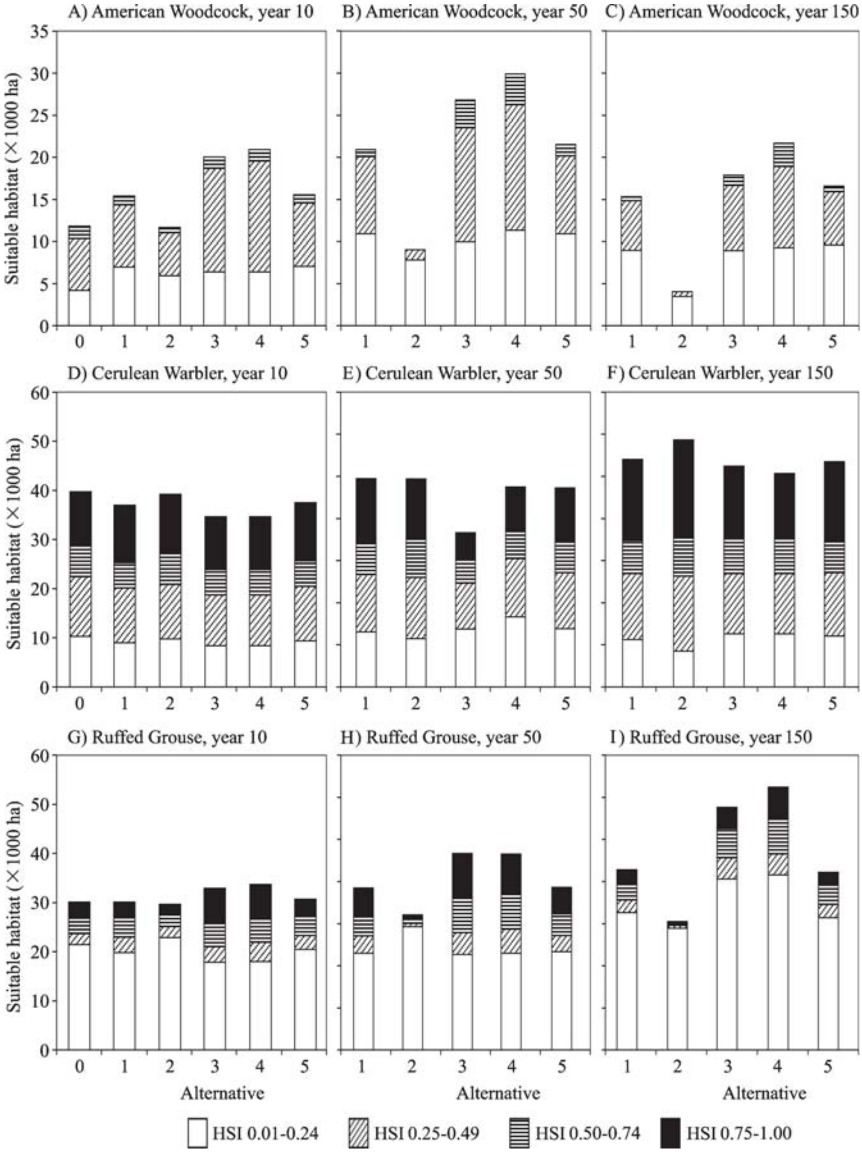
13.3.2 Changes in avian habitat suitability

13.3.2.1 American woodcock

The American woodcock is a ground-nesting, migratory species associated with early- to mid-successional, moist forested areas (Keppie and Whiting 1994). High quality American woodcock habitat for breeding occurs on mesic forest sites containing deciduous species 1-40 years old with interspersion of forest and open habitat. Alternative 4 had the highest tree harvest levels and highest prescribed fire levels among all alternatives. These levels of disturbance created early successional (regeneration) habitat used by woodcock for display and nesting, and the interspersion of young and old forest. Compared to Alternative 1, the amount of high quality woodcock habitat (HSI > 0.75) in Alternative 4 increased by 150 percent by year 10 and 10800 percent by year 150 (Fig. 13.8). Alternative 5, which added the focal area to Alternative 1, increased the amount of high quality habitat by 170 percent by year 50 and 830 percent by year 150. Under Alternative 2, the amount of high quality habitat increased by 30 percent by year 10, largely due to succession of open areas and gap-level dynamics associated with tree mortality from senescence, windthrow, or disease. However, the continued absence of tree harvest or prescribed fire agents led to the elimination of high quality habitat by year 50 (Fig. 13.8). When ranked by the total amount of suitable habitat, the rank of each alternative was constant over time (Fig. 13.8).

13.3.2.2 Cerulean warbler

The cerulean warbler is a neotropical migratory species that breeds in large tracts of mature and second-growth deciduous forests of eastern North America (Hamel 2000). High quality cerulean warbler habitat for breeding in the study region occurs in deciduous forest patches exceeding 100 years of age and 3,000 ha in size. Compared to Alternative 1 at year 10, the percent change in the amount of suitable habitat for cerulean warbler ranged from an 8 percent decrease in Alternative 3 to no difference in Alternative 5 (Fig. 13.8). The greatest separation of management alternatives occurred around year 50, with Alternative 3 producing a 53 percent decrease and Alternative 2 producing a 15 percent increase in the amount of high quality cerulean warbler habitat compared to Alternative 1 at year 50 (Fig. 13.8). It is unclear whether the percent change at year 50 was an artifact of the initial landscape conditions



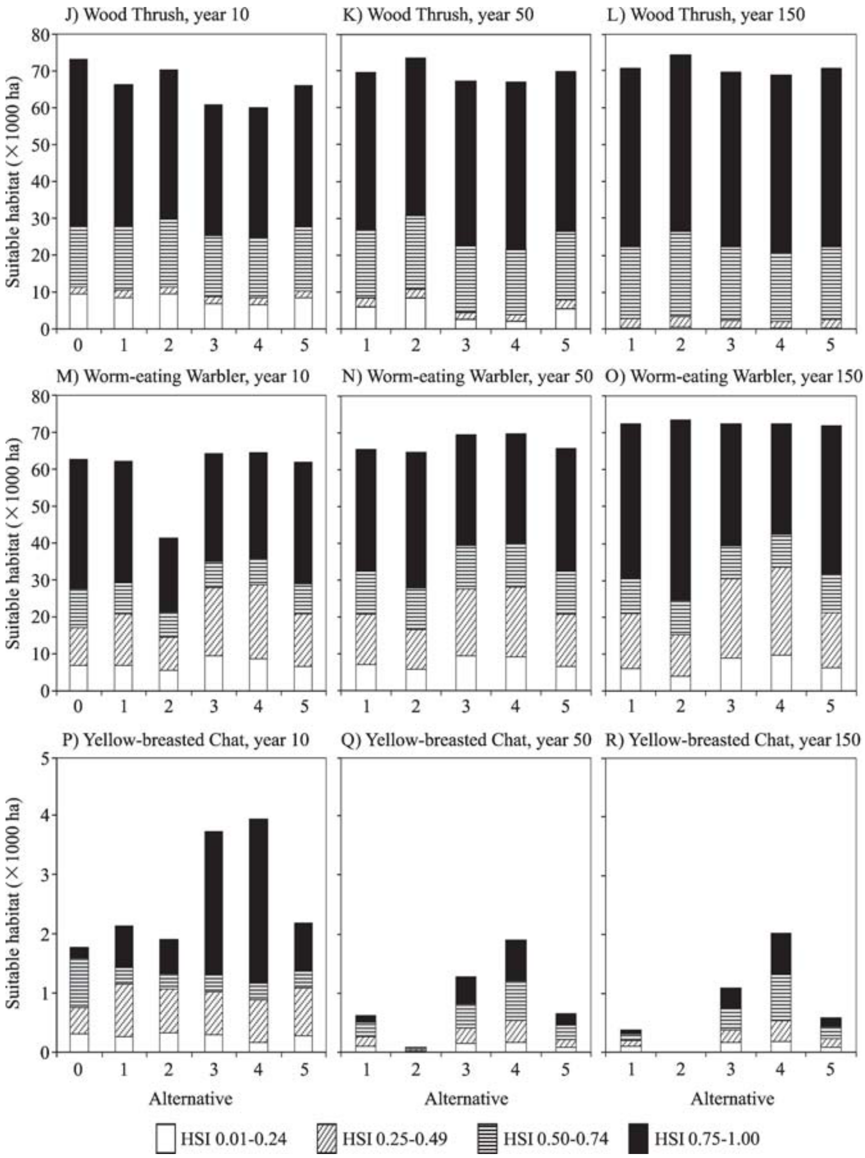


Fig. 13.8 Amount of suitable habitat (in ha) by alternative at year 10, 50 and 150 on the Hoosier National Forest, Indiana. Current conditions presented as Alternative 0 in year 10 column.

or a result of the tree harvest and prescribed fire levels. By year 150, all management alternatives had greater amounts of high quality habitat and greater total amount of suitable habitat than initial conditions. Alternative 2 produced 20 percent more high quality habitat than Alternative 1 (Fig. 13.8).

The relative rank of each alternative was not constant over time; Alternative 4 provided a greater amount of suitable habitat in year 50 than all other alternatives except Alternative 2. By year 150, though, Alternative 4 had the least amount of suitable habitat among all alternatives (Fig. 13.8).

13.3.2.3 Ruffed grouse

The ruffed grouse is a non-migratory game species associated with early successional forests in all parts of their range (Rusch et al. 2000). High quality ruffed grouse habitat occurs in forests with small patches of early successional forest surrounded by mast-producing trees. All four of the alternatives that implemented tree harvest had a greater amount of high quality ruffed grouse habitat than Alternative 2 (Fig. 13.8). Alternatives 3 and 4 consistently produced more high quality habitat than Alternatives 1 and 5 due to the higher tree harvest levels and increase in prescribed fire (Fig. 13.8). Alternative 5, which added the focal area to Alternative 1, increased the amount of high quality habitat 10 percent by year 150 (Fig. 13.8). However, the greatest increase in total amount of suitable habitat and high quality habitat was achieved through a combination of the focal area and higher tree harvest and prescribed fire levels; Alternatives 3 and 4 each increased the amount of high quality habitat by 140 percent from that under Alternative 1 by year 10 (Fig. 13.8). The large increase in high quality habitat was maintained for the plan duration such that by year 150, Alternative 3 had 60 percent more high quality habitat and Alternative 4 had 140 percent more high quality habitat than Alternative 1 at year 150 (Fig. 13.8). The relative rank of each alternative was constant over time (Fig. 13.8).

13.3.2.4 Wood thrush

The wood thrush is a neotropical migratory bird that nests in shrubs and small trees in deciduous and mixed-deciduous coniferous forests in eastern North America (Roth et al. 1996). High quality wood thrush habitat for breeding occurs in large forests with both early- and late-successional forest. The change in the amount of high quality wood thrush habitat compared to Alternative 1 was greatest for Alternative 3 (9 percent decrease) and Alternative 4 (8 percent decrease) at year 10 (Fig. 13.8). At year 50, Alternatives 3 and 4 had 10 percent more high quality habitat as Alternative 1. The change from a decrease to an increase in the amount of high quality habitat from year 0 to year 50 was an artifact of the HSI model for wood thrush and the 10-year time step of the simulation. Because the initial landscape conditions contained only dominant trees, where harvest was implemented in the first 10 years, all cells were assigned a tree age of 1-10 years. The wood thrush HSI model assigned SI $= 0$ for all cells with tree age <10 years. As a result, the alternatives that implemented the highest levels of tree harvest (Alternatives 3 and 4) had the largest decrease in the amount of high quality habitat. By year 50, cells subject to tree harvest in the previous time steps were 10-40 years old.

Of those, any cells subject to 20-40 years post-harvest were retained by the wood thrush HSI model as post-fledging habitat. Addition of the focal area in Alternative 5 produced less than 1 percent difference in the amount of high quality habitat for wood thrush compared to Alternative 1 (Fig. 13.8). The relative rank of each alternative was constant over time (Fig. 13.8).

13.3.2.5 Worm-eating warbler

The worm-eating warbler is a neotropical migratory bird that nests on the ground in large tracts of mature deciduous and mixed deciduous coniferous forests in eastern North America (Hanners and Patton 1998). High quality worm-eating warbler habitat for breeding occurs in moist ravines within large patches of unburned deciduous forest. Alternative 1 had the greatest amount of high quality habitat at year 10 (Fig. 13.8). Alternative 2 had only 61 percent as much high quality habitat as Alternative 1 at year 10 but provided 10 percent more high quality habitat than Alternative 1 at year 50 and 20 percent more high quality habitat at year 150 (Fig. 13.8). Alternatives 3 and 4, which had the highest levels of prescribed fire (25 and 50 percent, respectively), provided 10 to 20 percent less high quality habitat than Alternative 1 at each time step (Fig. 13.8). Addition of the focal area to Alternative 1 resulted in a 5 percent reduction in the amount of high quality habitat at year 150 (Fig. 13.8).

13.3.2.6 Yellow-breasted chat

The yellow-breasted chat is a disturbance-dependent shrubland bird that breeds in deciduous and coniferous forests in North America (Eckerle and Thompson 2001). High quality yellow-breasted chat habitat for breeding occurs in the interior of early successional forest patches exceeding 5 ha in size. Without tree harvest or prescribed fire, Alternative 2 contained only 82 percent as much high quality yellow-breasted chat habitat as Alternative 1 at year 10 and had no high quality habitat after year 50 (Fig. 13.8). Alternative 5 increased the amount of high quality habitat by 20 percent at year 10 of the simulation, 60 percent at year 50, and 160 percent at year 150 compared to Alternative 1 (Fig. 13.8). However, the greatest amounts of high quality habitat were produced under Alternatives 3 and 4, which had higher tree harvest and prescribed fire levels than Alternatives 1 and 5, in addition to the focal area. Alternative 4, which had the largest even-aged cut size (16 ha), the highest level of even-aged management (3 percent per decade) and the highest level of prescribed fire (50 percent per decade), produced 300 percent more high quality habitat than Alternative 1 at year 10, 480 percent more at year 50, and 1160 percent more high quality habitat at year 150 (Fig. 13.8). The relative rank of each alternative was constant over time (Fig. 13.8).

13.4 Discussion

Our approach to land-management planning on the Hoosier National Forest contains desirable features from a large-scale, landscape perspective while retaining the fine-scale information useful for evaluating avian habitat suitability. We simulated spatially explicit changes in vegetation structure, composition, and configuration due to anthropogenic and natural agents of disturbance and succession. Our comprehensive treatment of these processes advances previous Hoosier National Forest planning efforts (Gustafson and Crow 1994; Gustafson et al. 2001) by utilizing spatially explicit vegetation and avian habitat suitability models. By retaining the spatial context, we revealed important differences among alternatives in terms of the cumulative effects of management actions. First, tree harvest and prescribed fire influenced not only the species composition of vegetation communities, but also the species composition of avian communities. This result is not surprising in general, but it is unique to examine these spatially specific interactions over large landscapes and long time periods for multiple management alternatives. The scenarios indicate that in the absence of tree harvest and prescribed fire, the HNF will likely be dominated by sugar maple within 125 years, and yellow-breasted chat and ruffed grouse may face extirpation within 50 years. Second, the spatial context of tree harvest affected habitat suitability for early successional bird species. By concentrating even-aged timber harvest within a focal area, a given level of tree harvest provided more suitable habitat for yellow-breasted chat and ruffed grouse than applying the same tree harvest level across the entire HNF, without appreciably affecting habitat suitability for the late-successional bird species. Thus, linking vegetation simulation and avian habitat models provided a straightforward, intuitive, and scientific basis to support subsequent management decisions.

Our approach provided a comprehensive yet readily communicable perspective of landscape change. One of the goals of the HNF planning team was to engage the public and instill ownership of the HNF plan. The vegetation and avian habitat suitability maps were important tools for visualizing changes in landscape configuration, such as the spatial patterns that emerged over time from the different tree harvest techniques, despite similar composition with respect to tree age classes (Figs. 13.2, and 13.3). The maps also facilitated discussion of the HNF management goals and the methodology for achieving those goals, including the type and location of tree harvest activities. For example, public responses to proposed management actions on the Hoosier National Forest typically identified tree harvest as a controversial activity. The type and intensity of tree harvest affected forest species composition (Fig. 13.6) and ultimately affected avian habitat suitability through impacts on forest structure and mast production by the red oak and white oak groups. Thus, when selecting among management alternatives it is important to clearly understand the simultaneous tradeoffs, the potential conflicts, and the potential synergies among avian habitat quality for multiple species, levels of tree harvest, and by extension the availability of products, services and amenities that improve people's lives.

13.4.1 Tradeoffs among management alternatives

We tracked tree age class, tree species, core area, edge density, and avian habitat suitability information for five different management alternatives over a 150-yr planning horizon. No single management alternative maximized vegetation and habitat conditions for all features and species of interest. Rather, tradeoffs existed among all management alternatives. The sharpest tradeoffs occurred among Alternative 2, which contained no tree harvest or prescribed fire, and the remaining alternatives in terms of early vs. late-successional forest conditions and species composition. The range of alternatives considered was consistent with contemporary public land management polices, but narrow compared to the extent and severity of anthropogenic disturbances that affected this landscape over the previous 150 years. Modeling generalized management scenarios that incorporate higher levels of disturbance via harvest or prescribed burning is an approach that can be used to gain insights into how higher levels of disturbance are likely to affect vegetation structure, vegetation species composition, and avian habitat suitability without modeling new alternatives across the entire HNF (Shifley et al. 2006).

The LANDIS projections of dominant forest vegetation (Figs. 13.6, and 13.7) illustrate four important points with respect to management decisions. First, white oak will increase in area of dominance under all alternatives. White oaks are generally longer lived and marginally more shade tolerant than species in the red oak group. Over the next century, white oaks currently in the forest canopy are expected to survive in greater proportion than the red oaks. Second, the proportions of red oak species and maples are affected by the intensity of forest disturbance via harvest and fire. Red oaks are favored more than white oaks and much more than maples in the face of intense and/or repeated disturbances such as harvest or fire. This dynamic is visible in the pattern of tree species composition change over time (Figs. 13.6, and 13.7). The relative proportion of red oak to maple increases over time in response to increasing levels of disturbance. Third, in the absence of anthropogenic disturbance, the HNF will be dominated by late-successional vegetation conditions. Finally, the alternatives differed greatly in terms of the area subject to even-aged versus uneven-aged harvest techniques. Importantly, the relatively large increase in early successional vegetation due to even-aged management under Alternative 4 did not correspond to a large reduction in the amount of suitable habitat for late-successional bird species. Several recent studies support our simulation results that sustainable levels of harvest based on single tree selection, group selection or clearcutting improve habitat conditions for early successional bird species with minimal impacts on late-successional birds (Annand and Thompson 1997; Robinson and Robinson 1999; Gram et al. 2003), provided the spatial distribution of cuts maintains core areas of mature forest (Wallendorf et al. 2007).

Alternatives 3, 4, and 5 modeled the 5,260-ha focal area designed to consolidate the location of even-aged regeneration harvests for the benefit of bird species that depend on early successional forest habitat. The effect of the focal area was most apparent when comparing Alternative 1 with Alternative 5, which had the same tree harvest and prescribed fire levels, but Alternative 5 contained the focal area. Alternative 5 increased the effective size of early successional forest patches within the focal area and provided a greater amount of suitable habitat for ruffed grouse and yellow-breasted chat than Alternative 1. The focal area also increased interspersion of early successional forest patches with mature, mast-producing forest, and this improved the habitat suitability for ruffed grouse. Besides those avian benefits, the focal area reduced the amount of tree harvest occurring elsewhere on the forest. This would generally benefit the aesthetic qualities of vegetation outside the focal area, but may simultaneously reduce habitat suitability for ruffed grouse and yellow-breasted chat outside the focal area.

13.4.2 Interactions between public and private lands at landscape scales

Our approach to land-management planning was designed to take advantage of LANDIS's ability to simulate changes in forest vegetation over time under different management scenarios, and to produce GIS layers of outputs (e.g. tree age, tree species, wind damage, and fire history) over time. We used those GIS layers as inputs for the avian HSI models. This approach worked well within the predominantly forested landscape of the HNF; however, it had less value when applied to the non-forested parts of the HNF and surrounding private lands. Time since disturbance and type of disturbance (e.g. grazing, haying, and prescribed fire) are important factors in determining what bird species will be present within grasslands (Walk and Warner 2000). The 10-yr time step we used limited our ability to model succession within grassland vegetation and associated changes in habitat suitability. However, newer versions of the LANDIS software permit modeling and analyzing vegetation change using annual time steps (He et al. 2005).

An important consideration of our approach was the treatment of private lands adjacent to the HNF. Private lands cannot be relied upon to meet policy requirements for species viability on National Forests, but they may play a vital role in the conservation and management of habitat for many avian species particularly when public lands are embedded within a predominantly private land matrix. Private lands provide adjacent habitat that can complement or detract from habitat quality on public lands.

We used different tree harvest scenarios on private lands (e.g. high grading and selective harvest) other than on the HNF. We made four simplifying assumptions about private land management. First, we assumed private land-management trends were static over time; we did not increase or decrease the area of private lands subject to tree harvest per decade. However, tree harvest constraints on public lands may increase tree harvest on private lands (Haynes 2002). Second, we assumed the amount and location of public and private lands would remain constant over the analysis period. Third, we assumed that land use (forest, agriculture, developed) would also remain constant, even though conversion to residential development is likely to increase in some regions of the United States (Brown et al. 2005; Pocewicz et al. 2008). Fourth, we assumed private land parcel size would be stable over time. However, the average size of private forest land parcels is decreasing over time (Mehmood and Zhang 2001). If these trends extend to private lands adjacent to the HNF, then habitat suitability for some bird species could decline across the entire landscape over time despite management efforts on the HNF per se. Coordination of site-specific management efforts among private and public ownerships is certainly desirable and may be necessary to achieve regional avian habitat and conservation goals (Thompson and DeGraaf 2001).

13.4.3 Habitat suitability as a proxy for viability

Throughout the forest planning process the HNF planning team assumed that changes in habitat suitability were synonymous with numerical changes in avian populations. Rittenhouse et al. (2010) validated the wood thrush and yellow-breasted chat HSI models using 10-year territory density and nest success data from the Missouri Ozark Forest Ecosystem Project (Shifley and Kabrick 2002). They found support for HSI models as predictors of demographic response to vegetation change, but the strength of support varied by demographic response (e.g. territory density, nest success) and species. Other modeling approaches link population viability modeling to LANDIS using a habitat model as an intermediate step between vegetation simulation and viability analysis (Akcakaya et al. 2004; Larson et al. 2004). These modeling approaches may be considered an advancement over HSI models because of the link to population viability. Yet, at a minimum, population viability analysis requires estimates of adult survival and fecundity. Unfortunately, demographic data are lacking for many avian species despite being the critical link needed to translate population goals into habitat objectives. Further, when multiple species are included in the planning process it is convenient to have one metric for comparison among species. Thus while HSI models may not represent a demographic response for all species, they remain a common and convenient basis for evaluating wildlife habitat for many species.

13.5 Recommendations for future planning efforts

The detailed and synthetic nature of our approach provides a framework and structure that (1) is readily conveyed to multiple constituencies, (2) is based on explicitly stated assumptions and relationships, (3) provides a basis for testing, refinement, and extension to other forest commodities and amenities, and (4) provides a way to consider cumulative effects of multiple forest attributes at multiple spatial and temporal scales. We offer the following recommendations and observations to help guide the future application of landscape and wildlife habitat models to conservation planning:

1) Define explict, detailed management objectives. They are the focal points for model development, verification and validation, application, and comparison of outcomes.
2) Carefully consider tradeoffs between geographic extent, resolution of the modeling approach, and study objectives. While coarse resolutions will reduce processing time of large landscapes, they may lack detail necessary to assess impacts of forest management on avian species of concern. Select a resolution that is small enough to simulate disturbance, succession, and avian habitat at a scale relevant to the objectives. We chose to model ecological processes and habitat suitability at a resolution of 0.01 ha to account for disturbances as small as a tree fall gap. This resulted in hundreds of hours of computer processing to estimate habit suitability. However, the burden associated with modeling of avian habitat suitability for multiple species may be alleviated by additional computing capacity or modifications to the algorithms.
3) Model outputs, and even the modeling process, can be valuable tools for fostering communication and discussion with stakeholders. Understanding the relative differences among management alternatives and uncertainty associated with the modeling process is critical for making informed management decisions. Presentation of future vegetation conditions and avian habitat suitability as interactive maps provides scientific information in a format amenable to comprehension by the diverse stakeholders involved in the planning process.
4) Develop methods for evaluating and comparing multidimensional outcomes of management alternatives. We produced a variety of tabular and graphical output to allow comparison of management alternatives. The planning team recommended an alternative based on a review of these materials and stakeholder input. Alternatively, mathematical models could be developed to guide the selection of a preferred management alternative based on quantifiable objectives, provided such objectives can be articulated. Planning teams often default to choosing from a few alternatives by consensus because of the difficulty of quantifying objectives and developing optimization approaches when many resources and bird species are being considered.

5) We believe planning, model development, and application will be most effective when considered in an adaptive management framework. Ongoing monitoring of forest response during implementation of the chosen Forest Plan can provide valuable feedback on model performance and other assumptions made in the planning process, especially when models have not been previously validated. Forest plan implementation executed in specific project areas ranging from a few hundred to a few thousand hectares in extent can utilize many of the same modeling tools, although at finer resolutions and/or with greater site-specificity. The associated forest inventory and monitoring can provide a means to test and improve forecasting capabilities specific to a geographic region.

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Chapter 14 Agriculture Abandonment, Land-use Change and Fire Hazard in Mountain Landscapes in Northeastern Portugal

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Abstract

In this study we analysed changes in pattern and process in a Mediterranean mountain landscape, the França parish, Portugal, over a 47-yr period of time. We quantified changes in composition and configuration based on land use/land cover data obtained from 1958, 1968, 1980, 1993, and 2005 aerial photography. For the same period, we studied changes in fire, a relevant ecological disturbance in the region sensitive to change in land use/land cover pattern in the same period, through fire behaviour modeling and simulation with FlamMap and FARSITE. The results showed that the study landscape went through relevant modifications over the past 47 years. Agriculture land was replaced by shrublands and forests decreasing progressively from 22% of the landscape in 1958 to less than 5% in 2005. Shrublands were the dominant land use in all the dates. Structurally, there was an increase in patch size for the most combustible land classes (shrublands and forests) as well as in connectivity for the same classes. Fire simulations indicated that as landscape structure changed over time there were also changes in fire behaviour parameters: increase in fire intensity in the landscape and average burned area. There was, however, a decrease in average burned area for 4-hr simulated fires from 1993 to 2005 and a decrease in the rate of growth of the highest fireline intensity class (EXTREME). This study showed that changes caused by human abandonment affected the structure of a mountain area in the northeast of Portugal which enhanced landscape

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conditions for the occurrence of larger and more intense fire events over time.

Keywords
Abandonment, fire modeling and simulation, FlamMap, FARSITE, Parque Natural de Montesinho, Portugal.

14.1 Introduction

Abandonment has become a major process of landscape change in many human-dominated landscapes (MacDonald et al. 2000; Alados et al. 2004; Shoyama 2008). Abandonment is restructuring landscapes across the globe affecting dominant large scale processes [e.g. Burgi and Turner (2002) for North America; Otto et al. (2007) for Macaronesia; Fu et al. (2006) and Shoyama (2008) for Asia; and Hoffman and Rohde (2007) for Africa] as well as the services these landscapes provide in extensive areas (Pereira et al. 2005).

Currently, in Europe, people and activities tend to be concentrated in the most accessible and productive locations with the correspondent marginalization of large abandoned areas of extensive land use (Antrop and Van Eetvelde 2008). This trend is also evident in Mediterranean landscapes, that is, agriculture intensification and abandonment act simultaneously (Aranzabal et al. 2008). Abandonment started mainly after the agriculture industrialization phase following the WWII supported by advancement in mechanization, chemical fertilization, and irrigation (Zomeni et al. 2008). It was also favoured by socioeconomic, technological, natural, and cultural driving forces of change (Burgi et al. 2004). It didn't, however, take place uniformly throughout Europe.

Mountain areas were some of the first to undergo agriculture abandonment since there agriculture is considerably less competitive than in other areas due to biophysical, cultural, economical and social constraints that limit productivity and capacity for adaptation (MacDonald et al. 2000; Lasanta et al. 2006). The agriculture intensification period and the "post-productivist" era that followed in northern and central areas in Europe were often absent in mountain areas which went directly from subsistence or extensive land use systems to abandonment (MacDonald et al. 2000; Mottet et al. 2006; Zomeni et al. 2008). Some mountain areas, however, showed a tendency to maintain ownership structure and production systems and to invert abandonment in recent years (Lasanta et al. 2006; Mottet et al. 2006).

Agriculture abandonment causes recognizable effects on landscape composition and configuration. As expected, agriculture land is replaced by shrub and forest land uses (Lasanta et al. 2006). These changes can occur at relatively high rates, particularly in marginal areas, where natural vegetation becomes rapidly dominant (Poyatos et al. 2003). Many times abandoned agriculture areas are replaced by less intensive land uses, often perennial crops such as Christmas trees or woody biomass production systems for energy (Antrop and Van Eetvelde 2008).

Landscapes in the Mediterranean region of Europe have experienced variable change due to abandoment. In “dehesa” systems (*Quercus ilex* or *Q. suber* savannah-like systems) in the south of Spain, abandonment is in general a relatively minor cause of change (5% in area from 1956 to 1998) (Garcia del Barrio et al. 2004) in spite of higher local rates of change (Plieninger 2006). In mountainous areas of the Mediterranean region, agriculture has dropped in a more pronounced way. For example, in the mountainous areas of the northwest of Portugal, agriculture decreased by near 29% in area from 1958 to 1995, mostly replaced by tall shrublands and forest plantations (Moreira et al. 2001b). In the Mediterranean coast of Spain, agriculture decreased by 23% in area from 1956 to 1993 (Lloret et al. 2002). In Greece, values up to 92% of agriculture land abandonment were observed between 1945 and 1995 (Zomeni et al. 2008). In the Spanish Central Pyrenees, 78.9% of the area cultivated in 1940 was abandoned by 1981 and replaced by homogeneous shrublands (Lasanta et al. 2006).

Changes in landscape configuration as a consequence of land abandonment have also been evaluated in Mediterranean mountain areas. The trend is towards decreasing landscape heterogeneity (Lloret et al. 2002; Lasanta-Martinez et al. 2005; Moreira et al. 2008; Zomeni et al. 2008) although not all the results from available pattern analysis studies showed strong evidence of that occurrence (Romero-Calcerrada and Perry 2004). Change in heterogeneity is dependent on the initial heterogeneity of the landscape of interest. Since many mountain landscapes facing abandonment are complex landscapes integrating heterogeneous biophysical and human influences, abandonment results in loss of landscape heterogeneity. In simpler landscapes, often strongly human-dominated, abandonment can actually increase heterogeneity (Garcia del Barrio et al. 2004; Aranzabal et al. 2008). Heterogeneity depends also on the components selected to analyse it (Li and Reynolds 1995) and on the indicators selected to express it, issues that have not been properly addressed in landscape change studies.

Effects of abandonment on landscape function are seldom evaluated in the literature, particularly in Mediterranean mountain landscapes, although they are major concerns in the scientific and political communities (Kleijn and Sutherland 2003; Moreira and Russo 2007). Landscape change driven by abandonment has impacts on composition and on distribution patterns of birds (Farina 1997; Preiss et al. 1997; Moreira et al. 2001a; Sirami et al. 2008) and other vertebrates (Moreira and Russo 2007). Agriculture fields in mountain areas are particularly rich in plant and animal species. Abandonment of these areas will cause the replacement of open areas specialist species by shrubland or scrubland and forest species. Community composition will ultimately depend on the vegetation dynamics and the role of fire in landscape fire pattern dynamics (Moreira and Russo 2007). Plant richness, however, can increase in agriculture systems after abandonment (Bernáldez 1991; Bonet and Pausas 2004). Domestic species can also be affected by cropland abandonment in mountain areas due to the resultant reduction in grazing resources (Lasanta et al. 2006). Land abandonment affects other landscape functions, namely hydrological processes (Carvalho et al. 2002), carbon sequestration (Bolliger et al. 2008) and disturbance.

Fire is part of landscape and community dynamics in all terrestrial ecosystems including the Mediterranean regions (Pausas and Vallejo 1999; de la Cueva and Martin 2008). Landscape change driven by abandonment in mountain areas in the Mediterranean region of Europe is likely to change conditions affecting fire behaviour and regime. Fire intensity and extent are expected to increase in abandoned landscapes as a result of landscape homogenization and fuel load accumulation (Lloret et al. 2002; Lasanta-Martinez et al. 2005). Frequent or intensive fires can also favour fire-prone vegetation types in the landscape (Romero-Calcerrada and Perry 2004).

Being a subject of growing interest for scientists and practitioners, the effects of changes caused by abandonment in mountainous Mediterranean landscapes on fire behaviour and regime have seldom been formally addressed and evaluated in the literature, with a few exceptions. Moreira et al. (2001) found an increasing number of fires along with landscape structure change in nothwestern Portugal. Landscape homogeneity indices (Homogeneity and Angular Second Moment) were positively correlated to fire occurrence in Valencia, Spain (Vega-Garcia and Chuvieco 2006). Lloret et al. (2002) also found a similar trend based upon statistical tests with landscape pattern and fire data in Catalonia, Spain.

Contemporary land abandonment in Portugal follows the great emigration movements of the 20th century. Depopulation in rural areas resulted mostly from massive emigration movements to Brazil (around 1.5 million from 1880 to 1960, mainly in the 1900-1930 period), to Europe (e.g. one million to France from 1958 to 1975) and, until 1974, to the former Portuguese colonies (more than 250, 000 from 1943 to 1974). These are significant figures considering that population in Portugal in the beginning of the 20th century was only 5 million. Most of the emigration movements came from the countryside, namely from the northern interior mountainous regions, in spite of particular geographic migration patterns (e.g. Madeira and Azores Islands to Canada and California, USA). Emigration was caused principally by the lack of economic development of the country and by the natural attraction of the New World (Brazil, Africa, Canada, and the USA) and the post-war-developed Europe. Internal processes, such as the afforestation of communal lands campaigns launched in 1938 in the north and centre of Portugal and the unsustainable rural development policies starting in 1929, rushed these movements. Depopulation of mountainous areas persists today with abandonment as its most visible consequence (Moreira et al. 2001b; Moreira et al. 2008). Changes at the landscape level caused by abandonment in Portugal are expected to be similar to those described above for other Mediterranean mountain areas in Europe (Moreira et al. 2001b; Coelho-Silva et al. 2006; Moreira et al. 2008).

14.1.1 Case study — the França parish

In this work we analyzed changes in landscape composition and configuration from 1958 to 2005 in the França parish, northeastern Portugal, an area subjected to strong depopulation since the 1950's. Considering the actual changes that took place during this period, we analyzed the effects of change in landscape structure on fire behaviour through fire modeling and simulation. This is an effective way to test the hypothesis that "change at the landscape level driven by land abandonment leads to increasing fire hazard [a fuel complex defined by volume, type condition, arrangement, and location, that determines the degree of ease of ignition and of resistance to control (NWCG 2008)]" since it physically simulates the process of fire spread under changing conditions and it is able to avoid the unpredictable effects of human behaviour on fire ignition. In spite of the relevance of the topic for scientists and managers, the study of the relationship between landscape change and fire behaviour has not received much attention from researchers in Portugal and other Mediterranean countries that are simultaneously affected by severe fires and landscape change.

14.2 Methodology

The França parish is located in the Bragança Municipality, northeastern Portugal. This is a 5, 700-ha area mostly on a granite plateau above 1, 000m (maximum elevation 1, 481m). The area also comprises part of a larger flat valley of metamorphic geology. Average annual precipitation is above 1, 200mm reaching a maximum value of 1, 600mm at the highest elevations. Mean annual temperature is 8℃ or less.

The França parish comprises 3 villages—França, Portelo and Montesinho, whose population totalled 275 inhabitants in 2001 (4.9 inabitants/km^2). Only 101 of these were active population (Fig. 14.1). Within the study period, population decreased from 834 inhabitants in 1960 to 275 in 2001 (Fig. 14.1). Agriculture (including forestry and animal husbandry) is the most important activity in the parish (40% of the active population in 2001). Currently, tourism is the second major activity providing jobs for 18% of the active population.

The França parish has a very high conservation value. It is both a Special Area of Conservation, according to the EC Habitats Directive (92/43/EEC),

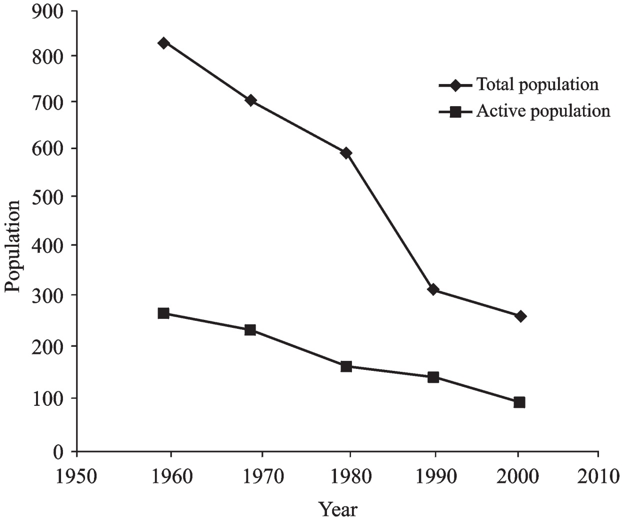

Fig. 14.1 Population in the França parish during the study period. Active population refers to people within certain age limits that are potentially repeated or actually involved in an economical activity. Source of data: Instituto Nacional de Estatística, Lisboa.

and a Special Protection Area, according to the EC Bird Directive (79/409/EEC). It is, therefore, part of the Natura 2000 Network, the European Union conservation network [Montesinho/Nogueira Site (PTCON0002)]. The significance of the area for conservation was first recognized in 1979 when it was included in the Montesinho Natural Park, a protected area established by the Portuguese government and named after the mountain where it is located and the highest elevation village it comprises. Local fauna includes Endangered and Vulnerable species such as the Pyrenean desman (*Galemys pyrenaicus*), the Iberian wolf (*Canis lupus signatus*), or the golden eagle (*Aquila chrysaetus*), among many others. Flora is noteworthy mainly for finicolous species such as *Ranunculus abnormis*, *Rumex longifolius*, *Viola parvula*, and *Viola bubanii* that find in the França parish the southwesternmost limits of their distribution. One species, *Rubus genevieri*, is practically endemic from this area.

We used digitized aerial photographs of 1958 (1:26, 000), 1968 (1:28, 000), 1978 (1:30, 000) and 1993 (1:22, 000) (provided by the Instituto Geográfico do Exército, Portugal) and a orthophotomap made from 2005 digital photography provided by the Câmara Municipal de Bragança (1 m resolution). The 1958, 1968, 1978 and 1993 images were orthocorrected based on 40 ground points, on average, for each photo. Photointerpretation was based on the COS'2005 land use/land cover system of the Instituto Geográfico Português (Caetano et al. 2006), a 3-level hierarchical system. Each land unit above 0.25 ha was classified according to the most detailed level of the COS'2005 system in a GIS. Field confirmations were conducted for the most recent coverage, when necessary.

Landscape transition probability matrices were calculated between sequence dates based upon cross tabulation of land use transitions between dates. Landscape structure for each of the dates above was quantified based upon landscape metrics calculated for level 1 land use/land cover classes using FRAGSTATS software (McGarigal and Marks 1995).

We used surface fire modeling and simulation to analyse fire behavior in the França parish over the 1958-2005 study period. A fuel modeling approach allows the relationship between landscape change and fire hazard to be addressed in a functional way. It also allows the definition of parameters for expected fire regimes with changing biophysical characteristics of the area independent of unpredictable weather conditions and of social factors that affect fire (ignition frequency and location) in regions where fire has usually a human origin. Burned area and fireline intensity were the parameters chosen for fire characterization.

We used two complementary models: 1) FlamMap 3.0 (Finney 2006) to map and analyse spatial variability of fire behaviour simultaneously for the entire study area in each of the dates under consideration; 2) FARSITE (Finney 1998), a deterministic fire growth simulation model, to simulate fire behaviour in a spatially explicit way. Although part of the same model family and sharing basic cartographic information (the Landscape File LCP), FlamMap and FARSITE offer complementary information concerning fire behaviour and hazard of the area under consideration. FlamMap calculates fire behaviour parameters (e.g. fireline intensity, flame length, or rate of spread) independently for each cell of a grid coverage of the area of interest based upon constant weather and wind parameters (Finney 2006). FARSITE, on the other hand, simulates spatially the spread of fire from known ignition points over time, considering variations of weather and wind parameters (Finney 1998). FARSITE has been used to analyse the effects of management alternatives on fire spread (LaCroix et al. 2006; Ryu et al. 2007).

The 5 land use/land cover vector coverages were converted into fuel raster files considering the correspondences in Table 14.1. Seven out of the 13 standard NFFL (Northern Forest Fire Laboratory) fuel models (Anderson 1982) used in FlamMap/FARSITE softwares were considered for the study area (Table 14.1). NFFL fuel models have been used in the Iberian Peninsula for fire prevention planning and research purposes (ICONA 1990; Loureiro et al. 2002; Loureiro et al. 2006; DGRF 2006). Canopy cover files were created with parameters in Table 14.2. Terrain data is a 10m resolution DEM created for this work based upon existing altimetric cartography (10m isolines) at the 1:25, 000 scale.

For FlamMap simulations, fuel moisture conditions were fixed for all surface fuel models and landscapes independent of topography or canopy cover as: 1h=4%; 10h=5%; 100h=6%; Live Woody=70%; Live Herbaceous=70%.

These values correspond to extreme summer weather conditions. Wind speed was 25 km/h at 6 m elevation. We assumed an uphill wind direction for fire behaviour characteristics calculation, which is appropriate for fire simulations according to diurnal topographic winds. From the FlamMap simulations outputs we selected the variable fireline intensity (kW/m) in the direction of the maximum rate of spread to express potential fire hazard in the landscape. The output was later reclassified according to Fire Danger Classes (Alexander and Lanoville 1989) (Table 14.3) and analysed in terms of spatial pattern based on landscape metrics.

Table 14.1 Fuel models classification, distribution and correspondence with Land Use/Land Cover classes in the França parish for the 1958-2005 period. Correspondence between LULC classes and the fuel models was done according to DGRF (2006).

Fuel Group	Fuel Model	Area (%)					LULC class in the study area
		1958	1968	1978	1993	2005	
	1	33.1	31.7	26.3	20.9	16.8	Annual Dry Crop
Grass	2	10.3	3.3	3.9	8.7	10.2	Maritime Pine New Plantation; Natural Herbaceous Pastureland
Shrub	4	10.6	21.9	28.2	37.2	38.4	Dense Cytisus and Genista Shrubland
	5	5.2	5.6	7.9	7.4	4.4	Dense Cistus Srubland; Dense Erica, Ulex, and Mixed Erica-Ulex Shrubland; Maritime Pine Forest; Softwood Forest; Hardwood-Softwood Mixed Forest; Low Density Cistus Shrubland
	6	32.6	28.7	24.3	15.2	19.4	Low Density Cytisus and Genista Shrubland; Low Density Erica, Ulex, and Mixed Erica-Ulex Shrubland
Timber	8	3.4	3.6	4.3	5.2	5.6	Sweet Chestnut Orchard; Holm Oak-Other Hardwood Mixed Forest; Riparian Forest
	9	1.4	1.5	1.6	2.2	2.2	Oak Forest; Oak-Other Hardwoods Mixed Forest
No fuel	99	3.4	3.6	3.6	3.2	2.9	Urban Areas

In FARSITE we simulated fire spread with each of the 5 fuel maps using a set of 16 random locations as ignition points. Simulations were done with one ignition point at a time. Weather conditions were also fixed for the 5 dates corresponding to extreme summer temperature and air moisture. Wind speed was 25 km/h at 6 m elevation and wind direction was 120°. All the simulations were run from 2 to 6pm having, therefore, the same duration: 4h. Time step was 30 minutes. Although in several cases fire spread outside the parish area, we considered only the burned areas within the França parish. The results of the simulations for the Burned Area (ha) variable were averaged for each year.

Table 14.2 Canopy cover values used for different forest types in fire behaviour simulations.

Forest Type	Canopy Cover (%)
Riparian	80
Other Softwoods	80
Holm Oak	80
Oak-Chestnut	70
Holm Oak-Other Hardwood	70
Mixed Hardwood-Softwood	60
Maritime Pine-Other Softwoods	60
Sweet Chestnut Orchards	50
Poplar	50
New Maritime Pine Plantations	30
New Hardwood Plantations	30

Table 14.3 Fire danger classes.

Fire Danger Classes	Fireline Intensity (kW/m)	Difficulty of suppression
Low	<500	Possibility of direct attack on the head or flanks of the fire with hand tools.
Moderate	500-2, 000	Water use of burnout operations are necessary. Ground suppression is effective.
High	2, 000-10, 000	Aerial means are necessary for direct attack on the head fire. Spotting is expected
Extreme	>10, 000	Extreme fire behaviour. Direct attack on the head fire is ineffective.

Adapted from Alexander and Lanoville (1989)

14.3 Results

There were perceptible alterations over the 1958-2005 study period. For the major land use classes (level 1), agriculture presented the most significant change dropping from 1,174 ha (22% of the parish area) in 1958 to 260 ha (5%) in 2005 (Fig. 14.2, Table 14.4). This represents a 77% loss in agriculture area for this period. The agriculture abandonment rate is very regular after 1968, approximately 22.7 ha/yr. Overall rate of abandonment is 19.46 ha/yr. In the same period, forests increased from 741 ha (14% of the area) to 1, 118 ha (21%). Shrubland is the dominant land cover in any of the dates (Table 14.4) representing 47% of the parish area in 1958 and 52.5% in 2005. Semi-natural pasturelands also increased considerably representing, however, just a residual percentage of the area (Table 14.4). Most of the changes described above resulted from abandonment of agriculture fields that became dominated by shrubs in a short period of time. New forest plantations were established mostly in shrublands with the exception of the 1993-2005 period when agriculture areas were used in afforestation (Table 14.5). Some of the shrublands were, however, previously agriculture fields. The transitions between shrubs

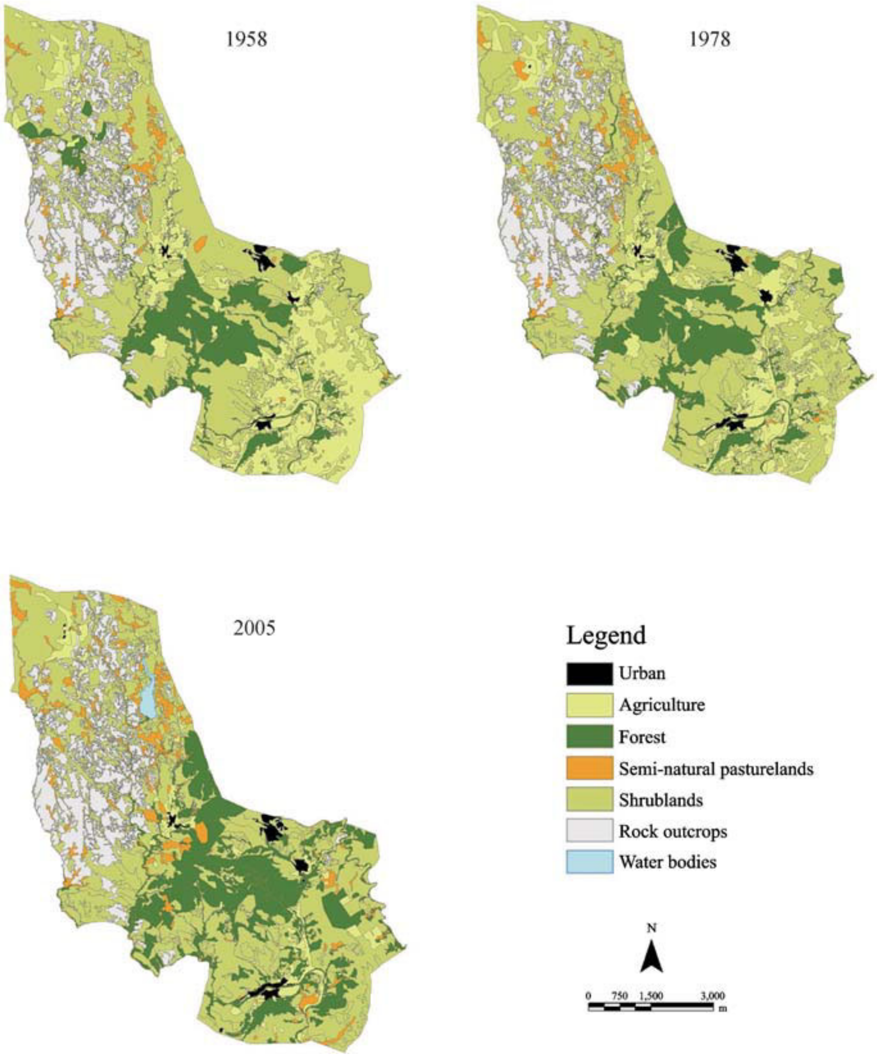

Fig. 14.2 Land Use/Land Cover level 1 classes in the França parish in 1958, 1978, and 2005.

and forests are irregular due to the frequent occurrence of wildfires, that set forest stands into shrublands, and to afforestation in shrublands (Table 14.5).

Table 14.4 Variation in percentage of Land Use/Land Cover (level 1) occupancy in the França parish from 1958 to 2005.

Land use/land Cover	1958	1968	1978	1993	2005
Shrublands	47.4	47.9	52.4	51.8	52.5
Agriculture	21.8	20.4	14.9	9.2	4.8
Forests	13.8	15.0	14.7	18.7	20.8
Rock Outcrops	14.2	14.2	14.3	14.3	14.4
Semi-natural Pasturelands	2.0	1.8	2.7	4.4	5.8
Urban	0.7	0.8	0.9	1.0	1.2
Water Bodies	0.0	0.0	0.0	0.6	0.6

Table 14.5 Transition probabilities of major Land Use/Land Cover classes for the entire period of analysis.

	Urban	Agriculture	Forest	Sem-Nat. Past.	Shrublands	Rock Outcr.	Water Bodies
1958-1968							
Urban	1	0	0	0	0	0	
Agriculture	0.0028	0.8429	0.0136	0.0030	0.1378	0	
Forest	0.0003	0.0042	0.9363	0.0017	0.0573	0.0002	
Semi- Nat. Past.	0	0.0906	0.0437	0.6608	0.2050	0	
Shrublands	0.0005	0.0383	0.0369	0.0085	0.9158	0	
Rock Outcr.	0	0.0000	0	0.0001	0.0002	0.9997	
1968-1978							
Urban	0.9890	0.0034	0	0	0.0077	0	
Agriculture	0.0036	0.6576	0.0191	0.0159	0.3033	0.0005	
Forest	0.0005	0.0159	0.7786	0.0314	0.1709	0.0027	
Semi- Nat. Past.	0	0.0233	0.0073	0.7059	0.2634	0.0002	
Shrublands	0.0018	0.0251	0.0535	0.0114	0.9048	0.0034	
Rock Outcr.	0	0.0000	0.0001	0.0002	0.0091	0.9906	
1978-1993							
Urban	0.9614	0.0004	0.0100	0.0014	0.0269	0	0
Agriculture	0.0075	0.5555	0.0492	0.1005	0.2857	0.0005	0.0010
Forest	0.0007	0.0089	0.8376	0.0034	0.1411	0	0.0085
Semi- Nat. Past.	0	0.0070	0.0020	0.8291	0.1578	0.0035	0.0007
Shrublands	0.0004	0.0145	0.1076	0.0105	0.8579	0.0012	0.0080
Rock Outcr.	0	0.0001	0.0017	0.0001	0.0037	0.9945	0
1993-2005							
Urban	0.9995	0.0005	0	0	0	0	0
Agriculture	0.0010	0.4462	0.1923	0.1492	0.2114	0	0
Forest	0.0001	0.0070	0.7457	0.0069	0.2390	0.0013	0
Semi- Nat. Past.	0.0124	0.0005	0.0245	0.8869	0.0758	0	0
Shrublands	0.0009	0.0116	0.0958	0.0083	0.8825	0.0010	0

Continued

	Urban	Agriculture	Forest	Sem-Nat. Past.	Shrublands	Rock Outcr.	Water Bodies
Rock Outcr.	0	0.0000	0	0	0.0005	0.9995	0
Water Bodies	0	0	0	0	0.0003	0	0.9997
1958-2005							
Urban	0.9560	0.0003	0.0038	0.0019	0.0381	0	0
Agriculture	0.0121	0.2007	0.1842	0.1100	0.4915	0.0008	0.0007
Forest	0.0016	0.0042	0.6997	0.0175	0.2727	0.0031	0.0012
Semi- Nat. Past.	0.0042	0.0081	0.0664	0.7615	0.1548	0.0042	0.0008
Shrublands	0.0043	0.0081	0.1470	0.0306	0.7934	0.0056	0.0111
Rock Outcr.	0	0.0002	0.0001	0.0003	0.0128	0.9867	0

Landscape configuration in the França parish also suffered modifications over the nearly 50-yr period. Landscape metrics considering major land use classes indicated that there was a decrease in heterogeneity at the landscape level. Diversity and Evenness as measured by the Shannon's and Simpson's indices decreased slightly during that time interval (Table 14.6) in spite of the new land cover established in the 1990's (water bodies) and part of the 1993 and 2005 data only. There were also decreases in the Number of Patches until 1978 and in the Largest Patch Index [percentage of the landscape occupied by the largest patch of all the land classes (Table 14.7)]. Initial and final values for Edge Density and Landscape Shape Index were practically the same although the indices suffered a strong decline from 1968 to 1993 when conversion of agriculture into shrublands was stronger.

Table 14.6 Variation in diversity and evenness landscape metrics in the França parish from 1958 to 2005.

Year	SHDI	SIDI	SHEI	SIEI
1958	1.3509	0.6880	0.7540	0.8256
1968	1.3477	0.6864	0.7522	0.8237
1978	1.3256	0.6607	0.7398	0.7929
1993	1.3662	0.6660	0.7021	0.7770
2005	1.3364	0.6551	0.6868	0.7643

SHDI-Shannon's Diversity Index; SIDI-Simpson's Diversity Index; SHEI-Shannon's Evenness Index; SIEI-Simpson's Evenness Index.

Considering the major Land Uses/Land Covers individually, agriculture decreased slightly in number of patches and strongly in mean patch area, Largest Patch Index and Edge Density (Fig. 14.3). Forests and shrublands decreased in Number of Patches and increased strongly in Mean Patch Area. The Largest Patch Index increased for forests and decreased for shrublands. Edge Density for these classes increased slightly during the 1958-2005 period. In summary, forests and shrublands became more aggregated in the landscape therefore creating fewer but larger landscape units. Agriculture became extremely fragmented and located in the vicinity of urban areas or in very high fertility sites.

Table 14.7 Variation in configuration landscape metrics in the França parish from 1958 to 2005.

Year	NP	LPI	ED	LSI	ENN_MN
1958	836	23.88	166.12	30.43	86.03
1968	906	14.64	170.27	31.19	146.30
1978	764	19.14	168.33	30.83	265.09
1993	766	14.93	164.49	30.13	209.36
2005	751	15.71	166.58	30.51	162.19

NP: Number of Patches(#);LPI: Largest Patch Index (%); ED: Edge Density (m/ha); LSI: Landscape Shape Index; ENN_MN: Mean Nearest Neighbour (m)

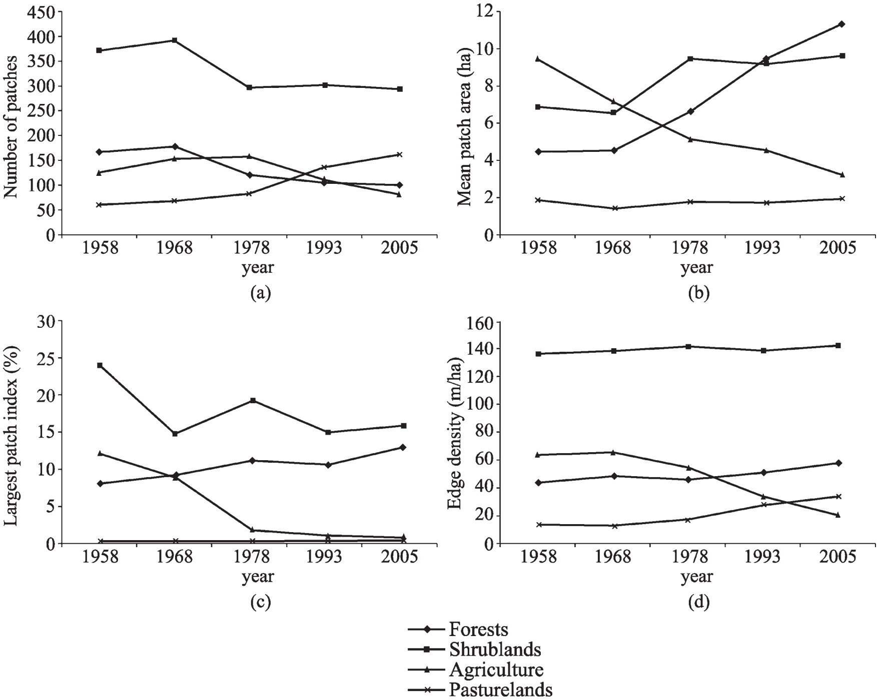

Fig. 14.3 Variation in selected landscape metrics for major land use/land cover classes (level 1) in the França parish from 1958 to 2005. Number of patches (a); Mean patch area (b); Largest patch index (c); Edge density (d).

FlamMap simulations showed a general increase in fireline intensity in the França parish during the 1958-2005 period (Fig. 14.4). These changes are due to a substantial increase in area of the maximum danger class (EXTREME) which goes from 10% of the parish area in 1958 to almost 40% in 2005 (Fig. 14.5). EXTREME class replaces HIGH and MODERATE classes over time. There was also an increase in average size area for the EXTREME class.

Although it was one of the classes with lowest number of patches, EXTREME increased slightly in this parameter. HIGH class increased considerably in Number of Patches (Fig. 14.5). Changes were more pronounced from 1958 to 1968. There was a reduction in the rate of increase of EXTREME area and

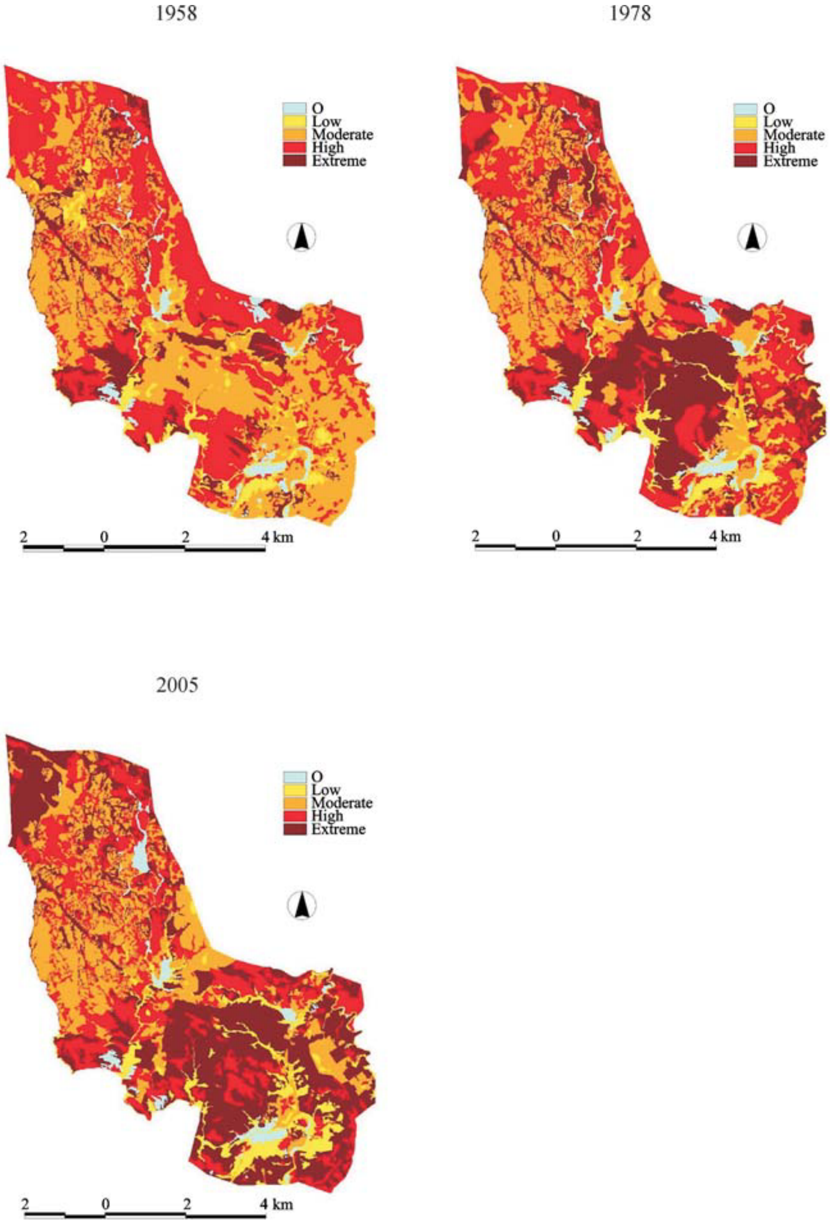

Fig. 14.4 FlamMap simulations outputs for Fireline Intesity (kW/m) for the França parish in 1958, 1978, and 2005.

Mean Patch Area from 1993 to 2005.

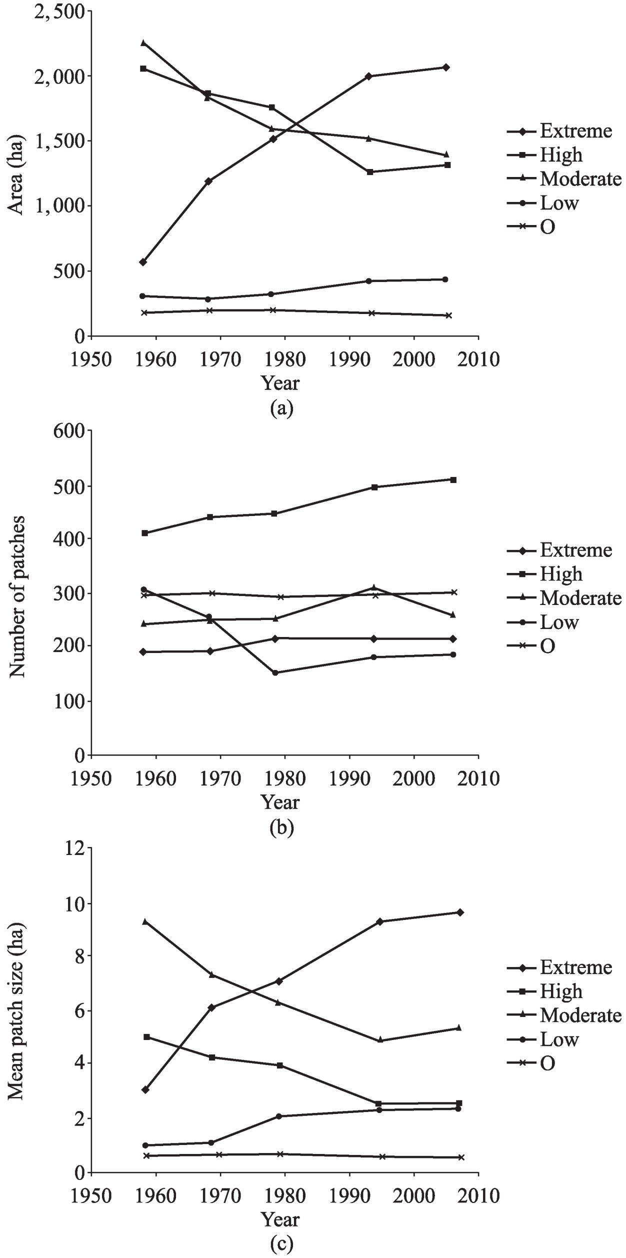

Fig. 14.5 Pattern measures for Fire Danger Classes: (a) Class Area; (b) Number of Patches; (c) Mean Patch Size.

According to the FARSITE simulations, average burned area per 4-h fire event increased, in general, over the 1958-2005 period of analysis (Fig. 14.6). This variation was faster for the 1958-1968 interval. For the rest of the period of analysis growth in burned area was small and there was a reduction from 1993 to 2005 in burned area for fires after 3 hours of simulation only (Fig. 14.6). Increases in burned area over time are more evident for the initial hours of fire spread where more recent dates showed a tendency to present larger burned areas (Fig. 14.7). Differences in burned areas among dates were not statistically significant (F test).

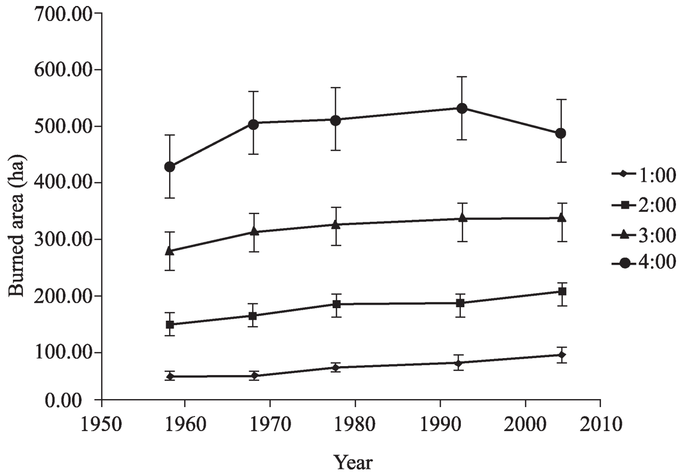

Fig. 14.6 Simulated average burned area over 4 hours in the França parish in the 1958-2005 period.

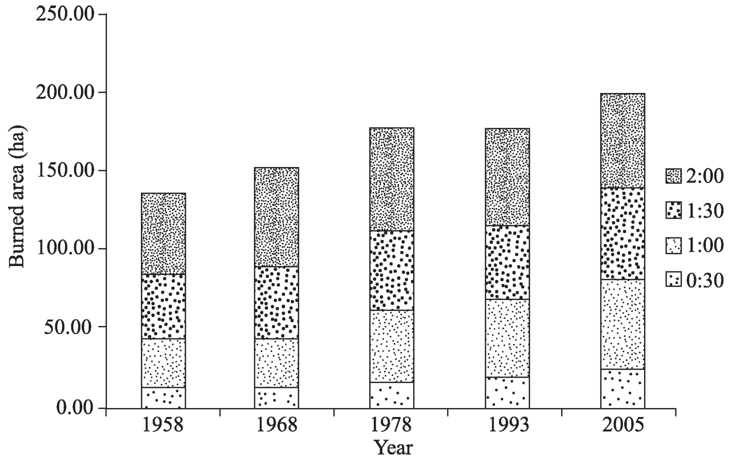

Fig. 14.7 Average contribution of each time step for the burned area over the initial 2 hours of fire in the França parish in the 1958-2005 period.

14.4 Discussion

Changes described for the França parish are similar to changes found in Mediterranean (and other) mountainous landscapes (Lasanta et al. 2006; Zomeni et al. 2008). Agriculture land abandonment is the most noteworthy process of change resulting in an increasing proportion of shrublands and forests, fire-prone land covers, in the landscape. The abandonment rate here seems relatively high compared with non-mountain regions (Garcia del Barrio et al. 2004). Causes for abandonment are mainly depopulation that occurred in the area (Fig. 14.1). Changes in landscape structure are also analogous to those detected elsewhere: increasing homogeneity in the landscape (Lasanta-Martinez et al. 2005).

Observed changes in composition and configuration correspond to changes expected to favour higher intensity and larger size of fires in the landscape (Vega-Garcia and Chuvieco 2006). Simulated fire behaviour in the França parish pointed out increasing conditions for the occurrence of high intensity fires over the 1958-2005 period. Not only there was an increase in the total area of the EXTREME fire danger class, replacing HIGH and MODERATE classes, but there was an increase in the size of the EXTREME class areas. Also, the simulated burned areas increased over the same period of time, particularly in the beginning of fire spread (<3h). In spite of the necessary cautions in the interpretation of these results (potential photointerpretation errors for past coverages, relatively small size of the study area, fire and vegetation growth cycles not matching sampling dates) this study showed that changes in landscape structure have favoured conditions for the occurrence of larger and more severe fires over time.

The relatively small rate of increase in average burned area, its decrease (4-hour fire events) from 1993 to 2005, and the decrease in growth rate for total area and mean patch area of the EXTREME fire danger class for the same period of time, are possibly related with growing fire-proneness in the landscape over time. Fire statistics (Table 14.8) suggest the occurrence of a very great number of fires and a corresponding large burned area in the França parish in the last 30 years. These recent reductions in rates of change raise the question whether the landscape will reach in the near future an “equilibrium” state between fuel characteristics (load and configuration) and fire occurrence. The results of this study indicate that in spite of frequent recent wildfires (Table 14.8), fire hazard increased the França parish over time. Additionally, the study area presents some potential for further transitions towards more combustible cover types, fuel accumulation and landscape homogenization and, therefore, higher fire hazard in the landscape. This increases the possibility of occurrence of large catastrophic wildfires with potentially severe impacts on wildlife, soil and water conservation (Carvalho et al. 2002; Moreira and Russo 2007).

Another question raised by the results of this work is relative to fire suppression, a current practice in this and other regions of the country. Will the maintenance of fire suppression strategies in the França parish in a context of agriculture abandonment, fuel accumulation, and increasing connectivity of flammable land uses benefit the landscape and the ecosystem services it provides? Possibly, under such a scenario, the fire-proneness of the landscape will farther increase thus making the occurrence of catastrophic fire even more likely.

Table 14.8 Wildfires in the França parish from 1975 to 2005.

Year	Number	Area (ha)	Area (%)	Year	Number	Area (ha)	Area (%)
1975	2	1, 447.1	25.3	1991	3	46.1	0.8
1976	4	27.0	0.5	1992	0.0	0.0	0.0
1977	2	333.6	5.8	1993	0.0	0.0	0.0
1978	10	1, 637.5	28.6	1994	8	302.6	5.3
1979	2	164.6	2.9	1995	6	111.6	1.9
1980	3	295.2	5.2	1996	14	555.0	9.7
1981	2	156.0	2.7	1997	3	36.8	0.6
1982	1	40.3	0.7	1998	6	203.8	3.6
1983	3	154.5	2.7	1999	10	139.4	2.4
1984	18	522.2	9.1	2000	8	843.4	14.7
1985	13	829.0	14.5	2001	8	176.4	3.1
1986	9	215.1	3.8	2002	2	18.2	0.3
1987	2	141.9	2.5	2003	0.0	0.0	0.0
1988	1	33.5	0.6	2004	2	20.5	0.4
1989	16	262.4	4.6	2005	1	10.8	0.2
1990	8	133.3	2.3				

Source: Parque Natural de Montesinho

14.5 Implications for management

Landscape approaches are necessary to address complex spatial processes, such as fire, in complex systems, such as Mediterranean mountain landscapes. Landscape ecology offers conceptual and practical tools for analysing, modeling, and managing spatial processes and their interaction with pattern (Turner 1989). Fire is a key process in the França parish driving the landscape dynamics and, on the other hand, depending on its landscape structure.

Considering the tendency observed in this study for increasing fire hazard over time, active landscape management in the França parish is required to prevent extreme fire regimes dominated by catastrophic events. Management priorities should include 1) decreasing fire hazard at the patch level, 2) increasing landscape heterogeneity, and 3) decreasing connectivity of the most flammable land covers in the landscape.

Vegetation patch level measures such as land use replacement or fuel reduction (e.g. mechanical or prescribed burning treatments) should be implemented to reduce fuel accumulation, which will contribute to increasing landscape heterogeneity. Heterogeneity can also be increased by reducing the overall area of the highest fire danger classes and the size of their land units. Maintenance of agriculture areas can contribute to fire hazard reduction at the patch level, simultaneously increasing heterogeneity and decreasing connectivity in critical areas at the landscape level.

While fire suppression strategies are important to avoiding loss of lives and property, they should be reconsidered in the context of ongoing landscape change. The occurrence of smaller and cooler fires during periods of non extreme weather conditions and in areas dominated by continuous shrublands would reduce fire hazard and connectivity, creating heterogeneity in these homogeneous areas.

Besides the identification of landscape structure and fire hazard trends in the França parish, this study can contribute significantly to the management of the area at the landscape level. First, the results obtained here allow the identification of the most critical areas where management is more urgent. These are corresponding to EXTREME and HIGH fire danger classes identified through FlamMap simulations (Fig. 14.4). The priorities mentioned above should, therefore, be directed mainly to these areas. Second, the methodology applied in this work proved useful to evaluate management measures and to provide baseline conditions for system analysis. The use of both FlamMap and FARSITE allows the simulation of fire behaviour in the study area according to management scenarios prepared to include the measures and priorities described above. This methodology makes it possible to define best scenarios for the reduction of fire hazard at the landscape level. Fire behaviour models have been used with this purpose in Portugal (Loureiro et al. 2002; Loureiro et al. 2006) and in the USA (LaCroix et al. 2006; Ryu et al. 2007). In this context, the results of our study can be used as baseline conditions with which alternative management scenarios could be compared.

14.6 Conclusion

During the 1958-2005 period, the França parish went through important modifications. There was a strong decrease in agriculture area and a regular increase in forests and shrublands. The resulting landscape is less diverse and more homogeneous. Fire simulations indicated that changes in landscape structure might affect fire behaviour parameters: increase in fire intensity and average burned area. Our main conclusion is that changes caused by human abandonment affected the structure of a mountain area in the northeast of Portugal which enhanced landscape conditions for the occurrence of larger and more intense fire events over time.

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Chapter 15 Overview of Biodiversity Loss in South America: A Landscape Perspective for Sustainable Forest Management and Conservation in Temperate Forests

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Abstract

South American forests contain a large fraction of the world's biodiversity, but it is obvious that if current trends of deforestation continue unchanged over the next decades, native forests will decline to an unacceptable levels. A landscape perspective to sustainable forest management and conservation provides a holistic framework to build up future research and tools towards an adaptive forest management approach to preserve forest biodiversity value while promoting the sustainable use of these forests. In this chapter we stress the importance of a landscape ecology perspective towards managing forests. We focus mainly on the temperate forests of Argentina and Chile, but within a broader framework of other forests in the South American region. An overview of threats to native forests is presented, and then new perspectives of conservation and management alternatives are analyzed. Our aim is to provide specific examples where a landscape ecology holistic approach contributes to integrating biodiversity value with the need for forestry activities, and provide insights into forest conservation and management initiatives, comparing traditional and timber-oriented management with new emerging approaches, including contributions of a landscape ecology perspective.

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Keywords
Native forests, ecosystem services, climate change, biodiversity conservation, adaptive forest management, restoration, human induced pressures.

15.1 Introduction

Biological diversity encompasses the variety of existing life forms, the ecological roles they perform and the genetic diversity they contain (FAO 1989). In forests, biological diversity allows ecological communities to adapt continuously to dynamically evolving environmental conditions, to maintain the potential for tree breeding and improvement (to meet human needs for goods and services and changing end-users requirements), and to support their ecosystem functions. Forest systems are impacted by multiple uses and influenced by global drivers. The multiple-use character of forests means that many different and sometimes conflicting goals exist regarding their management. Wide-ranging effects on the condition of forest ecosystems and their services are to be expected based on factors such as global market changes, increasing per capita income, demographic change, changes in consumption patterns, urbanization, globalization of the economy and new technology. Climate change will influence forest ecosystems around the world and the chain of causality is currently difficult to understand. Projected changes in the climatic system will affect natural and social systems globally, increasing their vulnerability and affecting their ability to supply goods and services to meet an ever increasing demand. For forestry and other natural resources management, the major challenges are in developing best practices for adaptive measures to maintain ecosystem resilience, and to reduce their vulnerability under various climate change scenarios.

While timber production often dominates the way in which forests were managed in the 20th century, new pressures in the 21st century drive a more balanced approach, calling for delivery of multiple goods and services. The process towards sustainable forest management is now considered consistent with the conservation of biological diversity. The values derived from biological diversity are associated with different scales that require different assessment methodologies. These methods included ecosystems, landscapes, species, populations, individuals and genes. Varying and complex interactions exist among all these levels.

We stress within this chapter the importance of a landscape ecology perspective towards managing forests. Forest management at landscape level is a complex practice of understanding the critical patterns of the landscape and their reciprocal interrelationship through the processes. Managing forests at a landscape level implies focusing on mosaics of patches and long-term changes in these mosaics to integrate ecological values (e.g. the maintenance of forest ecosystem health and biodiversity conservation) with economic and social purposes (e.g. timber and recreation). Within this framework in this chapter, we address the importance of native forests and particularly, the value of forest biodiversity for the temperate forests of the South America region. We provide insights into forest conservation and management initiatives with particular examples from Argentina and Chile, which present a commonality in the need for sustainable forest landscape management. Our aim is to provide key examples and synthesis where a landscape ecology holistic approach contributes to integrate biodiversity value at the same time as the needs for forestry activities.

15.1.1 Forests and drivers of change

Our environment is continuously undergoing change caused by a combination of social, economic and natural processes which operate at all scales from the local to the global. The Convention on Biological Diversity agreed in 1992, and more recently in the UN Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), put in evidence a growing international awareness in the importance of maintaining healthy ecosystems to preserve life as we know it today. The problems associated with managing changes are primarily associated with how to detect the patterns of changes and link them to processes such as forest fragmentation due to human economical activities and their pressures. The fact that vegetation is changing at a variety of spatial and temporal scales makes it essential that we take into account variability at one scale when trying to interpret it at another (Hobbs 1990). Within this context, continual alteration of the Earth's vegetation cover is perhaps the most ecologically significant human impact on the global environment, with particularly serious implications for habitat loss and the maintenance of biodiversity (Vitousek 1994).

Superimposed on natural patterns of vegetation changes, are pressures induced by human impacts. Therefore the knowledge of anthropogenic imprints in the landscape is crucial to understand the alterations in critical ecological processes. Integrating human activity into climate system models is one of the great challenges within Earth system models. Particularly, changes in land use and in vegetation cover through deforestation or reforestation, invasive species, and changes attributable to the manipulation of the natural fire regimes are among the key topics where integrative, landscape ecology approaches can make a difference in our ability to understand and manage forested landscapes. In many areas of the world, as in South America, the natural vegetation is being significantly impacted and fragmented by clearance for agriculture, cattle and harvesting, and is being markedly changed by those traditional management practices (Tucker et al. 1986; Woodwell et al. 1986; Saunders et al. 1987; Hall et al. 2002; Gea et al. 2004; Martinez Pastur et al. 2009).

Net forest area continued to decline in Central and South America (FAO 2007). The leading cause of deforestation is the conversion of forestland to agriculture. Within the region, the largest area loss is in South America, while the largest percentage loss of forest area is in Central America (FAO 2006a). The continuing high rates of conversion from forests to other land uses in many countries within the region is a matter of great concern to decision makers. A more recent human pressure that has an important impact on biodiversity loss is the increase in forest commercial plantations of non-native species throughout the region. In Chile, the National Decree Law 701-1974 has been the forest policy initiative that subsidised the development of commercial tree plantations, converting the industry as a leading export sector in this country. Afforestation and non-native species introduction is also encouraged by REED (Rural Energy Enterprise Development program) projects, with funding from the United Nations Foundation; this program regrettably is also a cause of native forests loss in the search for wood energy.

15.1.2 Forests and ecosystem services

Humans use ecosystems to obtain a wide range of goods and services, such as clean water, air, food, wood and fuel, in so doing they modify them (Lambin et al. 2001). If we focus on forests alone, the latest Forest Resource Assessment (FAO 2006b) indicated that the functions of forests globally included production (34.1%), protection of soil and water (9.3%), conservation of biodiversity (11.3%), social services (3.7%), multiple purposes (33.8%) and no or unknown function (7.8%). In other words, one third of the World's forests are used primarily for production of wood and non-wood forest products and more than 300 million ha of forests are designated for soil and water conservation. Ecosystem goods and services and their continued delivery are essential to our economic prosperity and well-being. Modification of ecosystems to enhance one service generally has come at a cost to other services. For instance human intervention has increased food and timber production although this has resulted in changes in other services such as water regulation and recreation activities. Since changes in the quantity or quality of various types of natural resources and ecosystem services have strong impact on human welfare and the competitiveness of an economy, comprehensive methods to measure and value biodiversity and ecosystem services are needed (Juutinen et al. 2008; Kallio et al. 2008).

An important reason for degradation of biodiversity and ecosystem services is that benefits of the ecosystem goods and services are not fully captured in the commercial markets or adequately quantified in terms comparable with economic services and manufactured capital. Therefore they are often ignored or given too little weight in policy making (Costanza et al. 1997). Decisions on the use of natural resources should be based on a comparison of the expected monetary value of the harvested products and the values associated with the ecosystem goods and services foregone as a result of harvesting in managed forests (Kallio et al. 2008). Forests in particular provide timber through well established markets, but the un-marketed benefits of forests include recreational activities, forest carbon sequestration, maintenance of biodiversity, climate regulation, protection against natural hazards and water quality. Some preliminary studies in Chile demonstrate that this approach is becoming an important tool for decision making regarding costs to conserve forest resources (Núñez et al. 2006; Lara et al. 2009).

Un-marketed benefits of forests are directly linked to biodiversity value and forest habitat quality, studies in biodiversity loss due to forest management in South America showed significant impacts on the overall biodiversity loss (Deferrari et al. 2001; Spagarino et al. 2001; Martínez Pastur et al. 2002; Ducid et al. 2005). However, when forest diversity assemblages were analyzed at the landscape level, only some insects groups (e.g. coleopterons and dipterans) were greatly affected (Lencinas et al. 2007, 2008). It was demonstrated that the application of adaptive forest management strategies is the key to integrate economic needs, societal values and biodiversity conservation priorities (Martínez Pastur et al. 2007, 2009; Lencinas et al. 2007, 2009).

15.1.3 An overview of biodiversity loss in South America

By any standard of measure, the Latin American and the Caribbean (LAC) region is the repository of some of the World's richest biodiversity, containing 40% of Earth's plant and animal species (Global Environment Outlook 2000). The region boasts important forest resources accounting for 22% of the world's forest area (FAO 2007). Nine of the 25 most biodiverse countries are located in the LAC region (Caldecott et al. 1994). Particularly, South America has one of the largest expanses of primary forest in the world, representing 45% of the total area of primary forest reported in the World (FAO 2007). These forests are also very diverse. For example, the forests of the northern Andes (Peru, Ecuador and Colombia) rank among Earth's most biologically rich ones, while, further south, Chile and Argentina share one of the largest single blocks of remaining temperate forest in the world (Dinerstein 1995; Myers et al. 2000). Although South America still maintains vast areas of intact tropical and temperate forest, the region's biodiversity is facing significant and growing threats, including increased rates of deforestation. The last Forest Resource Assessment (FAO 2006b) shows that South America suffered the largest net loss of forests from 2000 to 2005, where about 4.3 million ha yr^{-1} and carbon in forest biomass decreased from almost 100 Gt annually in 1990 to 90 Gt in 2005. Of the ten countries in the world with the largest annual net loss in forest area during 2000-2005, two are in South America: Brazil has the highest net change in forest area in the world during that period with a decrease of

>0.5% of its total forest area per year and Venezuela is the second in the list (FAO 2007).

15.2 The biological importance of the native temperate forests of South America

Chile and Argentina together harbour the largest temperate rainforest area in South America, and more than half of the temperate forests in the Southern Hemisphere (Donoso 1993). These forests are classified as temperate forests or temperate-rainforests because of their geographical location outside the tropics, and because they experience high rainfall and low temperatures in winter. Similar forests are found in Oceania (Australia, New Zealand and New Guinea) and the Pacific northwest in North America. Forests in South America are important as they store vast quantities of carbon that contribute to global climate regulation, flood control, water purification and soil nutrient cycling, as well as provide habitat for a high diversity of species that contribute to the genetic material for valuable new products and a foundation for the resilience of natural systems. As an example, the Chilean flora is estimated to consist of about 5.1 thousand species, more than 180 families and one thousand genera. More than 50% of the species are thought to be endemic (Marticorena 1990). Within Chile more than 60% of the total flora and the endemic species are concentrated in central Chile from about 29.0° to 43.5° S. Central Chile has been identified as one of the World's 25 biodiversity hotspots, which are areas that contain at least 1,500 endemic species of vascular plants (>0.5% of the World's total) and have lost at least 70% of the original habitat (Myers et al. 2000). In a recent review of the World's hotspots, the central Chile area was expanded and re-designated as the "Chilean Winter Rainfall —Valdivian Forests Biodiversity Hotspot". The native vegetation of this area is estimated to have declined from almost 400 thousand km^2 to less than 120 thousand km^2 (Myers et al. 2000).

The temperate rainforests of southern Chile and adjacent Argentinean Andes are the largest in South America and represent almost one third of the world's few remaining large tracts of relatively undisturbed temperate forests (WRI 2003). These rainforests contain unique species such as the monkey-puzzle (*Araucaria araucana*), which can live as long as 1,500 years, and alerce (*Fitzroya cupressoides*), one of the largest trees found in the Southern Hemisphere. Alerce has the second longest lifespan in the world, with some trees living more than 3,620 years (Lara and Villalba 1993). Owing to their special biodiversity assemblages, these forests provide important ecosystem services that are the basis for a range of economic activities, including water production (quantity and quality), aquaculture and sport fishing, and ecotourism (Lara et al. 2009).

15.3 Threats to native forests

The leading cause of deforestation in South America is the conversion of forestland to intensive agriculture and cattle grazing. As was mentioned also plantations are among the growing pressures, increasing at a rate of 1.6 % per year in Central and South America (FAO 2006b). Consequently, native forests are suffering from intensive pressures that are threatening biodiversity and persistence of the eco-regions in the short and long term. As a result of human-induced pressures on these native forests, habitat quality and biodiversity are being degraded or fragmented, and large areas of forest are being lost. In all, the region's biodiversity is facing significant and growing threats. Many forests have already passed a threshold beyond which recovery is impossible (Newton 2007). The present situation is rather distressing, and many people are calling for the protection of remaining forest areas. Although the rate of deforestation in South America is high, vast areas of intact tropical and temperate forest remain, and it is critical that conservation measures are targeted to such areas.

15.3.1 Changes and pressures: An example from southern Chile

Miles et al. (2007) provided an overview of the threats to the temperate forests of southern Chile, based on a survey of expert opinion. Principal threats currently include land cover change, browsing by livestock, logging/fuelwood extraction, habitat fragmentation, pollution, loss of keystone species, fires, and invasive species. In coming decades, other threats are expected to become increasingly important, including climate change and development of infrastructure (such as roads, pipelines and dams). The relative importance of these different threats varies between different parts of the region, but many areas are being subjected to multiple threats simultaneously. Another key issue is that many threats interact. For example, in southern Chile, Echevería et al. (2007) documented positive feedback between the effects of habitat fragmentation, intensity of browsing by livestock and harvesting of trees for timber. As forest fragments decline in area, they become more accessible to both people and livestock, progressively eliminating old-growth forest areas from the landscape. In another example, an increase in the frequency of fires has been a major factor in the decline of the native forests in Chile, with an average of 13.6 thousand ha yr^{-1} of native forests destroyed by fires over the past two decades (Lara et al. 2002, 2006). In the summer of 2001-2002, more than 10 thousand ha of *Araucaria araucana* forests were burnt in areas protected by the State (Echeverria 2002). These threats have produced a landscape in which native forests have become increasingly reduced in extent and fragmented (Echeverria et al. 2006). Fragmentation has a range of effects, including an increased susceptibility to fire and invasion by invasive species, reduced pollination and restricted seed dispersal (Forman and Gordon 1986). All of these can lead to an increased risk of extinction of some threatened species (Bustamante and Castor 1998; Bennett 2003; Bustamante et al. 2003; Echeverria et al. 2007). Fragmentation is also one of the greatest threats to Chile's native fauna, particularly for the mammals and birds that need large areas of intact forest to survive (Cornelius et al. 2000; Vergara and Simonetti 2004). Fragmentation also affects the ability of native forests and native species to respond to changes associated with global warming and climate change. In Chile, it is predicted that climate change should have its greatest impact on the forests of central and southern regions, especially at their northern boundaries with other ecosystem types (IPCC 2007). The distribution of the different types of native forest is strongly related to temperature, rainfall, evapotranspiration rates, soil types and hydrology. Changes in these factors could make some parts of the areas currently occupied by native species unsuitable.

Originally, the historical temperate forest cover, in Chile, particularly the Valdivia Rainforest Ecoregion (36° to 48° S) (Fig. 15.1), is estimated to have covered up to 18.4 million ha (Lara et al. 1999). According to the last vegetation mapping and assessment of the current forest area (CONAF et al. 1999), the native forests now cover only 13.4 million ha, a decline of more than 40% (Table 15.1). It is also estimated that more than 84% of the remaining forests are concentrated between 40° and 56° S. In southern Chile, between 35° and 38° S, key areas of rich floristic diversity have experienced

Table 15.1 Areas of native forests in the Valdivia Ecoregion* in Chile (between 18° and 56° S).

Forest types	Original data year 1550 (ha)	National Assessment 1997 (ha)	Remaining portion (%)
Fitzroya cupressoides	615,100	280,364	46
Pilgerodendron uviferum	1,035,509	557,812	54
Araucaria araucana	504,332	264,109	52
Austrocedrus chilensis	102,375	40,637	40
Mediterranean *Nothofagus*	983,143	314,075	32
Nothofagus betuloides	796,311	814,828	102
Nothofagus rainforests	4,513,083	1,509,949	33
Dryland forests	1,370,561	39,924	3
Broadleaved evergreen	5,453,022	4,201,796	77
Nothofagus pumilio	2,860,106	2,141,806	75
Chilean Palm	2,541	0	0
Nothofagus antartica	185,389	167,335	90
Total	**18,421,473**	**10,332,545**	**56**

**An ecoregion is a geographically distinct assemblage of natural communities that share a large majority of species, dynamics and environmental conditions (CONAF et al. 1999).*

a particularly severe decline in diversity along with a decrease of continuous forest patches. Currently there are no intact forest patches greater than 5,000 ha. This decline has been particularly severe in the Coastal Cordillera (WRI 2003).

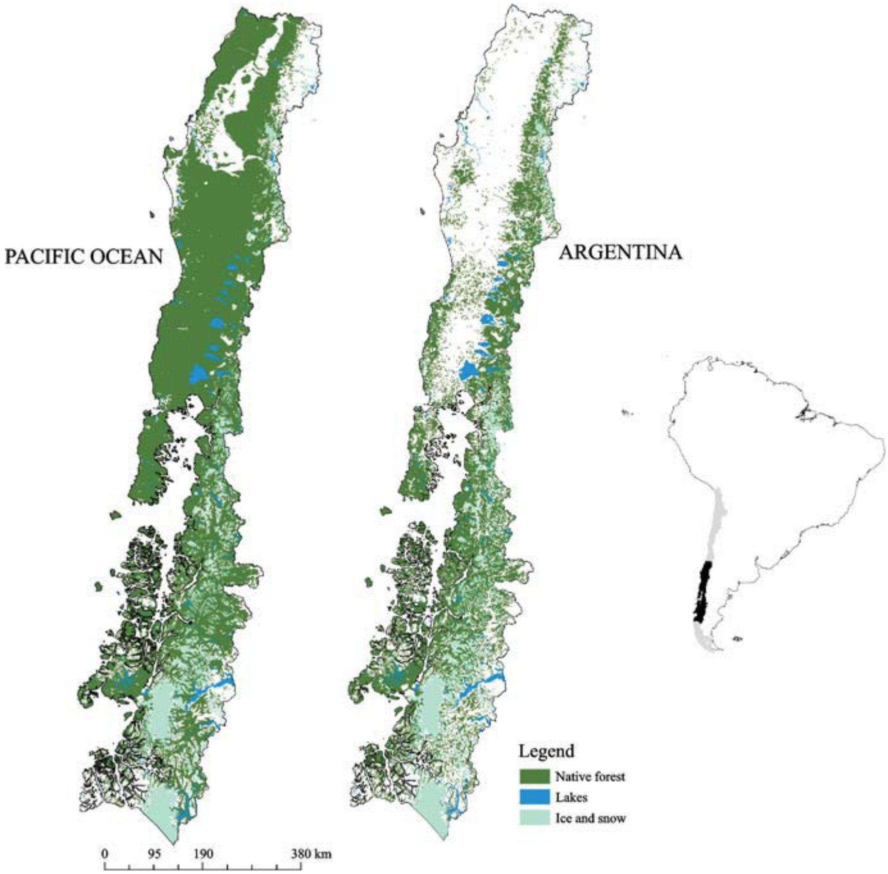

Fig. 15.1 Historical (1550-to the left) and current (1997-to the right) native forest cover, in the Valdivia Rainforest Ecoregion in southern Chile, between 35° and 38° S.

15.3.2 Changes and pressures: Examples from Argentina

In the case of Argentina, pressures on native forests are associated with the expansion of the agricultural frontier. Mainland Argentina covers a total area of 278.0 million ha with 41.5% in forested regions. Forest reserves protect several ecosystems (6.7 million ha) representing 5.8% of the total area. However, these reserves are not equally distributed varying from 0.1% in the central-northern region to 34.6% in the Patagonian forests (Table 15.2). Argentina has 28.9 million ha of native forests (Dirección de Bosques 2004; UMSEF 2007) (Table 15.2, Fig. 15.2 and 15.3). Similar trends as seen in southern Chile have been documented in Argentina, where native forests have decreased from 37.5 million ha in 1930 to 28.9 million ha in 2007 (UMSEF 2007). Deforestation and forest degradation are associated with a number of threats, which again vary in importance in different forest regions (Table 15.2). In order of importance, principal threats include cattle production, agriculture, settlement, firewood extraction, exotic forest plantation, harvesting and fires. One classic example of rapid conversion of land cover/land use with important negative impacts on the local population and biodiversity can be found in northern Argentina, where environmental conditions have encouraged the development of extensive industrial agribusiness (e.g. soybean, sugar cane, cotton and cattle

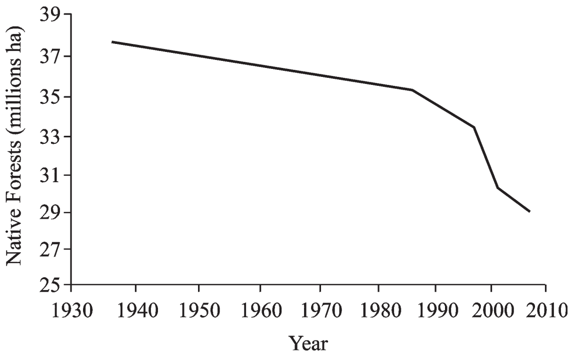

Fig. 15.2 Native forest rate loss in Argentina (1930-2007) (Dirección de Bosques 2004; Montenegro et al. 2005; UMSEF 2007).

Table 15.2 Main threats to native forests by region* in Argentina (0 = no pressure, 1 to 5 = pressure levels).

Forest Regions	Agriculture	Exotic plantation	Cattle production	Harvesting	Firewood extraction	Settlement	Fires
Selva Misionera	3	5	2	3	2	3	1
Selva Tucumano Boliviana	4	2	3	3	2	3	1
Parque Chaqueño	5	1	5	2	4	4	2
Espinal	5	3	5	1	5	5	4
Bosque Andino Patagónico	0	0	3	2	1	2	3

*For regions location see Fig. 15.3.

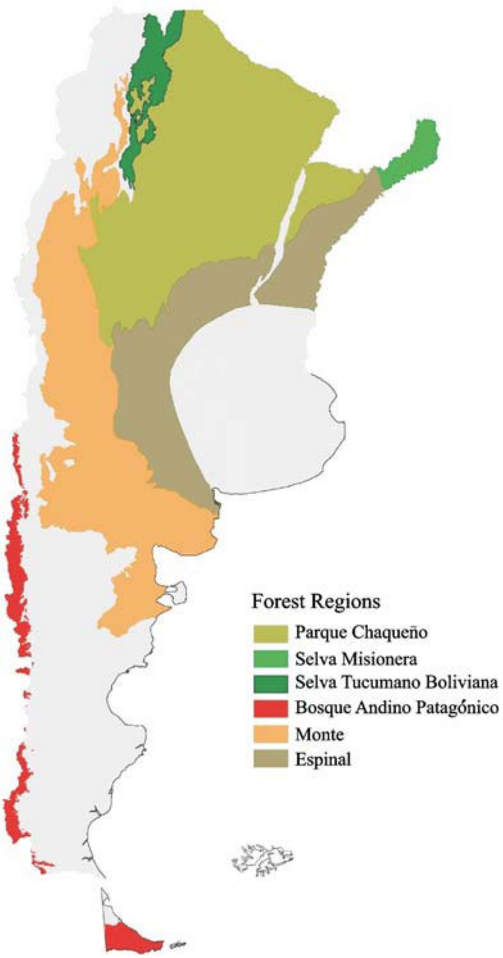

Fig. 15.3 Geographical location of native forest regions in Argentina (based on Dirección de Bosques 2004).

production). Furthermore, the development of agribusiness displaces local people who settle in forested areas, increasing the impacts through fuel-wood extraction and grazing livestock (cattle and goats) for family consumption.

In the south, in Patagonia, principal threats are due to human economical activities, such as cattle production, which impacts natural processes (e.g. forest succession and natural regeneration) and increases risks of forest fires. Many forest fires are intentionally set to increase the area of pastures or to allow the extraction of fuel-wood in places where live wood extraction is forbidden by law (e.g. Nahuel Huapi National Park).

Native forest harvesting in Argentina is mainly carried out in compliance with national and provincial law, but the lack of long-term forest policy leads to an economical and ecological degradation of the native forests. The continuous pressures of clear cuts and wood extraction without silvicultural planning result in unsustainable forest management.

15.4 Forest management and conservation strategies: A response to native forests' threats

In addition to the government system of national parks and reserves in Chile, there are numerous reserves that are not included in the State protection system, ranging in size from several hectares to several hundred thousand hectares. Many of these reserves are owned by local or indigenous communities, small collectives of private individuals, wealthy individuals or Chilean NGOs such as CODEFF (Chile's Committee for the Defence of Flora and Fauna, an affiliate of Friends of the Earth). Most of these reserves have been linked to form the Network of Protected Areas in Chile [Red de Areas Protegidas Privadas (RAPP)]. In 2005, the RAPP network included 133 reserves covering a total of 386.5 thousand ha (CONAMA 2005). In some cases private reserves have been established to assure connectivity (e.g. corridors and stepping stones) between the existing National Parks and the Reserve network. More than 80% of protected land is located either in the Andes or in the southern regions.

The increasing number of reserves could be an important milestone for the conservation of some threatened habitats and species. For instance, the Pumalín Park, a privately owned land located in southern Chile, is focused on the sustainable use linked to conservation actions in collaboration with local communities, containing significant areas of old-growth forests of *Fitzroya cupressoides*. This park is an example of a private initiative to protect an area that was declared a Nature Sanctuary on August 19 of 2005 by the Chilean government, granting it additional environmental and undeveloped protection. The Conservation Land Trust (a U.S. environmental foundation) donated an important part of these protected lands to Fundación Pumalín (a Chilean foundation) for their administration and continual development as a type of National Park with public access to a privately held reserve. However, many other reserves are relatively small and have only been recently established so that their longevity and the success of their management are not assured. On the other hand, between 35° and 39° S, many of the remaining native forests with a great biodiversity value are owned either by forestry companies or by small landowners. In the coastal Cordillera of these regions, the remnant forests on land owned by forestry companies tend to consist of scattered fragments either along watercourses or on unusable land and are often surrounded by plantations of *Pinus radiata*, *Eucalyptus globulus* or by agricultural lands. These areas are vulnerable to the effects of human disturbances, especially fuel wood extraction. However, a limited amount of restoration work is being undertaken on company lands. These companies are also developing codes of practice and management to obtain forest certification. This is a welcome development but the long-term commitment of these companies has yet to be tested and unless these areas are expanded, the biological viability of the smaller fragments remains uncertain.

In Argentina, forest reserves represent 5.8% of the total protected area, which encompasses 6.7 million ha of protected forest ecosystems (Table 15.3). However, these reserves are not equally distributed, varying from 0.1% in the Espinal region (northern Argentina) to 34.6% in the Patagonian forests (southern Argentina) (Table 15.3). These regions contain 28.9 million ha of native forests (Dirección de Bosques 2004; UMSEF 2007) (Table 15. 3).

Table 15.3 Forest reserves and native forest area by main forest Regions in Argentina (Dirección de Bosques 2004).

Forest Regions	Area	Reserves		Native Forests*	
	millions ha	millions ha	%	millions ha	%
Selva Misionera	3.01	0.49	16.1	1.45	48.3
Selva Tucumano Boliviana	5.48	1.50	27.3	3.73	68.0
Parque Chaqueño	67.50	2.46	3.6	23.37	34.6
Espinal	33.00	0.04	0.1	2.66	8.1
Bosque Andino Patagónico	6.45	2.23	34.6	1.99	30.8

15.4.1 Spatial conservation and prioritization approaches

Spatial conservation prioritization approaches, suitable for planning the expansion and connectivity of reserve networks constitutes one of the most successful conservation strategies to increase forest-protected areas (Margules and Pressey 2000; Pressey et al. 2007; Luque 2000; Luque and Vainikainen 2008). In 2005, globally, more than 400 million ha of forests, or 11% of the total forest area were designated for the conservation of biological diversity as the primary function. The area of forests devoted to conservation of biodiversity has increased by at least 96 million ha, or 32% since 1990. The latest trends show that in the last 15 years South America has one of the highest increases in conservation area in the World, from 70 million ha in 1990 to 92 million ha in 2005. However, declaring an area under a protection status is not enough. In order to preserve the future of forests, they need to be effectively managed to conserve the values for which they were created. Economic resources and technical capacity are limited and policy implementation is weak to implement a coherent plan for forest protection. Moreover, declaring forest-protected areas is not a viable option in most of the regions, as most of the land in the region is privately owned, and there is a need for income. Therefore, a viable conservation strategy in the region has to create corridors between forest-protected areas of different status. Viable areas for species movement, flow of materials and genetic exchange have been created with this approach (Pacha et al. 2007; Armenteras et al. 2003; Bennett 2003).

Clearly, the expansion of the protected areas should be based on all available information and expert knowledge. In addition to this, it is desirable to employ quantitative decision support tools to aid the decision making. Such tools in the form of decision-theoretic and optimization techniques have been developed in conservation biology, under the rubric of systematic conservation planning and spatial conservation prioritization (Margules and Pressey 2000; Cabeza and Moilanen 2001; Sarkar et al. 2006; Pressey et al. 2007). Previously, a variety of conservation approaches and solution methods have been applied in the context of forest conservation, such as species richness extrapolation (O'Dea et al. 2006), species compositional similarity (Steinitz et al. 2005), gap analysis (Montigny and MacLean 2005), multiple use management planning (Baskent et al. 2008), simple heuristic algorithms (Virolainen et al. 2001; Heikkinen 2002), genetic algorithms (Hölzkamper et al. 2006), simulated annealing techniques (Boyland et al. 2004; Rayfield et al. 2008) and linear programming optimization (Ricker et al. 2007). More recent works have provided a novel spatial conservation prioritization approach suitable for planning the expansion of conservation area networks. This approach is based on high-resolution GIS data covering the planning area. The relevant planning criteria depend on spatial information related to forest quality, connectivity of forest types, and proximity to existing conservation areas (Luque and Vainikainen 2008). Forest inventory data and remote sensing techniques at the regional and/or national level are used for the purpose of constructing a biodiversity quality index, which together with a cost-effect analysis provides an overall indicator suitable for protection of forest biodiversity (Juutinen et al. 2008). Kallio et al. (2008) incorporated similar indices from the same data into a spatial partial equilibrium model simulating the forest sector for optimal regional allocation of forest conservation sites.

Another option, apart from the design of protected areas and corridors, is to promote sustainable forest consumption. This is the approach of the Forest Stewardship Council (FSC), a nongovernmental, non-profit organization that promotes the responsible management of the World's forests. Established in 1993 as a response to concerns over global deforestation, FSC is widely regarded as one of the most important initiatives of the last decade to promote responsible forest management worldwide. FSC is the fastest growing forest certification system in the world (FAO 2007). Products carrying the FSC label are independently certified to assure consumers coming from forests that are managed to meet the social, economic and ecological needs of present and future generations. More than 100 million ha forest worldwide were certified to FSC standards in April 2008, distributed over 79 countries (http://www.fsc.org/facts-figures.html).

15.5 Management solutions: Modeling dynamics of forest ecosystems

Sustainability of forest ecosystems affected by the use of forest-based resources requires an understanding of the links and balance between productivity, natural forest dynamics, soil processes and their interaction with natural and anthropogenic disturbances. During the recent three decades intensive studies have been done to develop forest ecosystem models (Levine et al. 1993; Tiktak and Van Grinsven 1995; Ågren and Bosatta 1996; Friend et al. 1997; Morris et al. 1997; Mäkipää et al. 1998; Chertov et al. 2003). Modeling has been used to analyse the impacts of different harvesting systems, natural forest disturbances, forest dynamics, climate change and carbon balance. Forest ecosystem models can effectively extend the classical approach where growth functions and tables are used for the prediction of the forest growth and soil nutrition in the changing environment under new silvicultural regimes. The level of the basic forest unit (e.g. stand as inventory compartment) can now be modelled well in relation to the problems of the stand's productivity in different climatic and site conditions. Moreover, there are combined models which are able to describe the biological turnover of the elements (e.g. carbon and nitrogen) in the soil-vegetation system (Chertov et al. 2001; Komarov et al. 2003). Forest ecosystem models with a multi-scale approach are needed to meet demands from policy makers and managers to predict the impacts of different scenarios of use and management of forest resources.

Forest management practices, site preparation and fertilisation are known to deteriorate surface and ground water quality due to increased leaching of nutrients and export of suspended solids. Even though the export of nutrients from forested areas is far less than that from agricultural lands, because forest management is implemented in such large areas, the total stress on water bodies can be significant. The nutrient export from forested catchments can be effectively decreased by water protection, e.g. by leaving untreated buffer zones between water body and the treated forest area (Ahtiainen and Huttunen 1999; Jacks and Norrström 2004), as recommended by present guidelines for forest management. This, however, excludes the buffer zone areas from the practical forestry and causes losses of trade incomes from the timber located in the buffer zone. On the other hand, the buffer zones can provide important associated services like game, berries, habitat for species that are stressed by the management and improve the water quality exported from the catchments areas. So far, there are few studies concerning the costs and benefits of the buffer zones.

15.5.1 Landscape ecology as a tool for forest management and conservation

To achieve sustainable forest management, tools for assessing the forest system as a whole are needed. Both the protection of the remaining native forest and a sustainable management of forest and forestry operations for the whole landscape are needed. In recent years, several studies demonstrate that management of forest ecosystems should not exclusively occur at a single scale (e.g. at stand level) (Spies et al. 2002) or based on a disciplinary research framework (Wu 2006). On the contrary, the hierarchical and pluralistic framework of landscape ecology (Forman and Gordon 1986; Naveh and Lieberman 1994) may substantially facilitate the management and conservation of native forests. In Argentina and Chile, only a few examples exist of managing forest resources under a multiscale and interdisciplinary approach in order to maintain and restore the goods and services produced (Meynard et al. 2007; Lara et al. 2009) and to conserve biodiversity (Geisse and Nelson 2005; Hechenleitner et al. 2005).

A landscape perspective is needed whenever landscape spatial patterns can be expected to have a significant effect on forest health and sustainability (Fahrig 2005). Forests in Argentina and Chile have been severely affected by progressive fragmentation and forest loss in the last decades (Aizen and Feinsinger 1994; Echeverria et al. 2006, 2008). Under a landscape ecology perspective, it has been observed that changes in patch spatial attributes by fragmentation are associated with changes in forest structure and composition at the stand level. Particularly, changes were recorded in basal area and canopy cover (Echeverria et al. 2007) and species composition (Altamirano et al. 2007; Echeverria et al. 2007). Observed changes in canopy cover as a result of human disturbances (Echeverria et al. 2007) produce variations in growth and regeneration in uneven-aged forest in Chile (Donoso 2005). These changes have relevant implications for forest management (Donoso and Nyland 2005). Silviculture measures and conservation actions should not ignore the spatial patterns imposed at the landscape level. There is little doubt that landscape ecology is making a significant contribution to forest management and biodiversity conservation.

15.5.2 Managing strategies

Natural forests around the world have been mainly managed by the following economic criteria (McComb et al. 1993; McClellan et al. 2000). In the Northern Hemisphere, most of the natural forests were transformed by silviculture into single-species stands with a regulated age structure (Oliver and Larson 1996). Forests of Argentina and Chile follow this trend (Martínez Pastur et al. 2000), supported by prevailing economic interests regarding forest management decisions (Gea et al. 2004). Most of the silvicultural proposals recommend transforming the uneven-aged original structure to an even-aged managed forest via natural regeneration of the harvested stands through their own seed production (Schmidt and Urzúa 1982). The *Nothofagus* forests of Argentina and Chile have been traditionally managed through high grading cuttings or clear-cuts, and recently by shelterwood cuts (Schmidt and Urzúa 1982; Martínez Pastur et al. 2000; Gea et al. 2004; Rosenfeld et al. 2006), which significantly affects the original diversity (fungi, plants, birds, insects and mammals) (Deferrari et al. 2001; Spagarino et al. 2001; Martínez Pastur et al 2002; Ducid et al. 2005). Recently, ecological and social criteria have been elevated over economic criteria (DeBell and Curtis 1993; Mitchell and Beese 2002). For these reasons, new silvicultural methods were proposed. These new methods were designed to conserve some of the original heterogeneity of the natural old-growth forests. One of them proposes to leave 30% of the timber forests (stands with up to 40 m^3 ha^{-1} of saw-timber logs) as aggregated retention and 10-15% basal area as dispersed retention (for details see Martínez Pastur et al. 2007, 2009), which is expected to conserve the original biodiversity affected by forest management (Vergara and Schlatter 2006; Lencinas et al. 2007, 2009) (Fig. 15.4). Aggregated retention was defined as one circular patch of 60 m diameter per ha of original forest, while dispersed retention was composed of remnant trees homogeneously distributed between the aggregates. The implementation of this method was feasible at large ecological scale in Tierra del Fuego (Argentina). In this southernmost forest, the yield loss and costs increase due to the retention overstory and was compensated by the decrease in harvesting costs (Martínez Pastur et al. 2007). Furthermore, a short-term analysis showed that biodiversity ecological cycles were improved with this new method when compared to shelterwood cuts (Martínez Pastur et al. 2007; Lencinas et al. 2007, 2009). At a large scale, this method proved to be economically feasible across a gradient of site quality, producing stability in the remnant overstory and successful regeneration (Martínez Pastur et al. 2007, 2009) patterns.

Regeneration systems that include different kinds and types of retention were proposed to combine timber production interests and the consideration of other forest values (DeBell and Curtis 1993). The regeneration method with aggregated and dispersed retention maintains the same yield rates as the first cut of the shelterwood system. Contrary to shelterwood cuts, this method reduces both harvesting costs (Martínez Pastur et al. 2007) and biodiversity loss (Lencinas et al. 2007). The main disadvantage found was the loss of timber in the retained trees, caused by collateral damage while felling neighboring trees and blow-down after harvesting (Hickey et al. 2001). Nevertheless, this system helps to maintain ecosystem health, resilience, and productivity, as well as compositional, structural, and functional diversity of the managed forests (McClellan et al. 2000). Also, they produce a sustainable supply of timber while imposing a set of complex biologically and socially acceptable management objectives, combining economic and conservation purposes (Martínez Pastur et al. 2007).

Fig. 15.4 Variable retention silvicultural system with aggregated and dispersed retention in *Nothofagus pumilio* forests in Tierra del Fuego (Argentina).

15.5.3 Management alternatives for native forests: A solution for forest sustainability?

Within a multi-scale and a more mechanistic framework from traditional sylviculture, an integrative landscape-driven research program should be envisioned to relate ecosystem processes, global changes including climate changes and socio-economic processes across different governance levels. The development of adaptive forest management strategies under climate change is a key challenge for sustainable resource management worldwide. As climate changes, societal demands for goods and services from forests are also changing. Increasing demand intensifies the competition for resources between forest industry, the energy sector and nature conservation/other protective functions and services (including biodiversity, protection from natural hazards, landscape aesthetics, recreation and tourism). A main goal for a future sustainable forest management should be focused on the development and evaluation of different strategies that can adapt forest management practices to multiple objectives under changing environmental conditions with a particular focus on the landscape level effect.

Conservation strategies of forested landscapes must consider forest habitat quality and biodiversity value (Luque et al. 2004; Luque and Vainikainen 2008). Forest management modifies biodiversity, with the subsequent species loss (Wigley and Roberts 1997; Deferrari et al. 2001; Jalonen and Vanha-Majamaa 2001; Spagarino et al. 2001; Martínez Pastur et al. 2002). These losses are associated with changes in forest structure, microclimatic conditions and/or nutrient cycles (Reader and Bricker 1992; Lewis and Whitfield 1999; Caldentey et al. 2001). However, most of the studies only analyze biodiversity loss in timber-quality forests (Thomas et al. 1999; Quinby 2000), without considering interactions with associated sites within the same landscape (Hutchinson et al. 1999; Rosso et al. 2000; Peh et al. 2006). Usually, forested landscapes are mosaics of different site types, where timber-quality forests rarely constitute large, continuous blocks. Natural timber-quality forests mainly occupy the best quality sites, which in most cases are also the ones with high yield marketable products. On the other hand, associated non-timber-quality stands include sites that are not harvested because of being not profitable, have legal restrictions, or have special protective functions (Lencinas et al. 2008). For example, in the central zone of Tierra del Fuego (Argentina), only 64% of the landscape forest area corresponded to timber-quality stands of *Nothofagus pumilio* characterized by large trees, with a closed canopy and high tree volume. The rest was conserved as associate non-productive environments, which act as biodiversity protection areas: *Nothofagus antarctica* forests represented 11% of the landscape, border forests (2%), streamside forests (8%), forested wetlands (2%), and open places (13%) conformed by grasslands and peatlands (Lencinas et al. 2008).

15.6 Conclusions

Protected areas are just one part of the solution towards the maintenance of forest biodiversity in the region. In order to preserve the future of forest protected areas, proper management and resources are needed to conserve the values for which they were created. Nevertheless, increasing demands for forest supplies and energy will continue to set up pressures on valuable forests systems. Only with an adequate sustainable management can forest biodiversity be preserved; managing for biodiversity, water quality or natural disturbance requires a regional or landscape perspective. In addition, managers must also begin to anticipate how activities in one area might affect the physical and biotic properties of adjoining areas. The challenge is then to improve forest management and productivity while keeping the strength on biodiversity conservation measures.

Understanding the interrelation between ecosystems and landscapes level mechanism is critical. An integrative landscape-driven research should be envisioned to relate ecosystem processes, global changes including climate changes and socio-economic processes across different governance levels. Within this integrative framework, predicting biodiversity change involves understanding not only ecology and evolution, but also the complex changes in human societies and economies. One of the most important challenges for future forest research will be to integrate research across different scales, including spatial and temporal scales within an interdisciplinary and multidisciplinary framework. The success of forest management activities are grounded in the emulation of natural disturbance patterns. Maintaining or creating particular landscape characteristics increases the likelihood that all the biological diversity associated with the landscape will be perpetuated. Taking a landscape-level approach means that planning and resource decision-making are undertaken in the context of the entire landscape, as opposed to planning for discrete parcels of land. By planning and managing at a landscape level and considering spatial and temporal aspects, both resource protection and sustainable use can be better accommodated without undue conflict or displacement.

There is a need to develop management and planning options both for landscapes that are already significantly altered, in need of either improved management or restoration and for forest landscapes, which are still relatively altered, but which are under increasing human pressures. The ability to provide such options depends on an understanding of landscapes processes and the ability to use this understanding to develop strategies, which are effective in dealing with the biophysical problems while at the same time being socially and economically acceptable. National and international support for regeneration and restoration activities is needed.

Baseline data and continuous high quality data bases are needed to plan and monitor forest management. In this chapter, most of the national and regional data we presented are from the 90's and other statistics for the region are based on FAO (2007). Subsequently, long-term data are needed to develop appropriate management options and plan for changes within climate change scenarios that will affect these native forests. Good forest inventory data at National level are proven to be very expensive and difficult to keep within political instability and short-term planning. Free and open access to biodiversity data is today a reality (http://www.gbif.org), but much work needs to be done to fulfil the data portal with good data quality for countries where they are most needed. Particularly data for forest monitoring at the right spatial and temporal scales are lacking. Integration of methods and a more intensive use of remote sensing as Lidar sensors and geo-statistics are needed. In summary, more support needs to be given to enhance collaboration and maintain long-term databases. It is essential to link good quality data with a sound institutional framework to ensure continuity and long-term collaboration. In that sense, funding to create continuous high quality data should be a worldwide effort.

In the same way, as the lack of long-term spatial ecological data, the region is lacking academic programs in conservation (Mendez et al. 2007). In order to meet the forest management challenges that lie ahead, capacity-building opportunities on landscape ecology and conservation need to be implemented that encompasses different levels, audiences and contexts (Bonine et al. 2003; Brooks et al. 2006).

Focus on concrete measures in relation to policy implications and problems of implementation are another big challenge. Forest legislation in Argentina and Chile is quite advanced, but problems remain on the implementation. In the first place, an international code of ethics for logging companies operating around the world is needed. Investment in research from logging companies is also needed in the regions where they exploit most of the resources. IUFRO has an important role to play in the future supporting a framework for an international legislation in relation to timber extraction and forest management.

Despite all the progress achieved, integrative research is lacking, innovative questions are evasive or difficult to get funded. We need to reach a better understanding of the interwoven landscape mosaics to elucidate complexity and scale interdependencies mechanisms within the forest system. Landscape ecology provides an interdisciplinary approach to actually bridge the many gaps we face today to work towards the new challenges and endeavours of human social and ecological processes. This holistic approach of forest landscapes management can help to build up future research and tools towards an adaptive forest management approach to preserve forest biodiversity value while promoting the sustainable use of forests.

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Chapter 16 Conservation of Biodiversity in Managed Forests: Developing an Adaptive Decision Support System

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Abstract

Forest ecosystems provide several goods and services, but strategies for the conservation of biodiversity are missing in traditional forest management schemes. In this paper we develope a decision support system to optimize the conservation of biodiversity in managed forests, taking Dadia National Park as a case study area, a local Mediterranean hotspot of biodiversity in northeastern Greece. Using environmental niche factor analysis, we produced a series of spatially explicit habitat suitability models for vascular plants, amphibians, small birds and raptors and an overall model for total biodiversity. Further, we produced maps related to timber production and investigated potential conflicts between conservation of biodiversity and wood production. A decision support system based on a conflict assessment was created using three management scenarios. It enables the establishment of integrated management strategies and the assessment of their effects on biodiversity and timber production. Habitat suitability models for selected groups of organisms were found very effective to investigate the impact of the management on forests and wildlife. Further evaluation of key indicator taxa on these models could improve decision support systems and the sustainable management of forests.

Keywords

Forest ecology, sustainable use, timber extraction, habitat suitability,

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raptors, birds of prey, amphibians, vascular plants, Dadia National Park, Greece.

16.1 Introduction

The increasing exploitation of forests is one of the main reasons of human-induced loss of biodiversity (Lindenmayer et al. 2002; Foley et al. 2005). Although the socio-economic value of biodiversity was underestimated until recently (Costanza et al. 1997; Farber et al. 2002), its maintenance has become a commonly accepted goal of sustainable forestry (United Nations 1992; Kohm and Franklin 1997). The concept of ecosystem services provides a tool for communicating the importance of intact ecosystems for human well-being and a framework for the evaluation of multiple functions of landscapes and forests (Costanza et al. 1997; De Groot et al. 2002; Millennium Ecosystem Assessment 2005; Boyd and Banzhaf 2007; Steffan-Dewenter et al. 2007). In forest ecology, a major challenge is finding trade-offs between timber production and conservation of biodiversity (Johns 1997; Putz et al. 2001; Foley et al. 2005; Burke et al. 2008).

Forestry practices can enhance or reduce habitat for particular wildlife species by altering structural features at the stand scale (Burke et al. 2008; Rendón-Carmona et al. 2009). Forest management that enhances the heterogeneity of forests has in general a positive impact on the local biodiversity (Loehle et al. 2005; Gil-Tena et al. 2007; Torras et al. 2008; Kati et al. 2010; Poirazidis et al. 2010a; Schindler et at. 2010), but forest management guidelines for the maintenance of biodiversity are mainly valid for site specific conditions and can be rarely used as general directions (Loehle et al. 2005). As it is impossible to measure and monitor the effects of various management practices on the entire ecosystem, indicators are used as surrogates for biodiversity (Lindenmayer et al. 2000). Taxon-based proxies include flagship, umbrella and indicator species (Caro et al. 2004; Roberge and Angelstam 2004; Hess et al. 2006; Cabeza at el. 2008), while structure-based ones deal mainly with stand complexity, connectivity and heterogeneity (Lindenmayer et al. 2000; Schindler et al. 2008). Many researchers have explored the use of particular taxa, especially vascular plants, arthropods and birds, as surrogates for biodiversity, but a general pattern has not yet emerged (Kati et al. 2004b; Sauberer et al. 2004; Sergio et al. 2005; Billeter et al. 2008; Cabeza et al. 2008; Zografou et al. 2009). The importance of including several guilds of taxa to represent adequately overall biodiversity is currently stressed by several authors (Angelstam et al. 2004; Edenius and Milusinski 2006; Loehle et al. 2006).

In this study, we developed a decision support system with the ultimate goal of providing management guidelines and optimal solutions for the conservation of biodiversity in managed forests. We considered Dadia National Park, a Mediterranean forest mosaic in north-eastern Greece, as a case study. Using available data sets from systematic scientific research in the area, a series of habitat suitability models for groups of indicator species and for overall biodiversity was produced to discover potential conflicts between biodiversity and timber production. Additionally, the effectiveness of different management scenarios was assessed.

16.2 Methods

The following method section contains information about the study area, the species data, and the applied statistical analyses. It further deals with the methods of producing maps of habitat suitability, timber standing volume, and forest management categories.

16.2.1 Study area

This research was conducted within Dadia National Park (hereafter called Dadia NP), a sub-mountainous area with a diverse landscape mosaic, dominated by extensive pine (*Pinus brutia, P. nigra*) and oak (*Quercus frainetto, Q. cerris, Q. pubescens*) forest, but containing also a variety of other habitats such as pastures, cultivated land, torrents and stony hills (Schindler et al. 2008; Poirazidis et al. 2010a). Dadia NP covers 43,000 ha in the prefecture of Evros, northeastern Greece (Fig. 16.1), and was designed to protect the diverse community of birds of prey, including the last breeding colony of the Eurasian black vulture (*Aegypius monachus*) in the Balkan peninsula (Poirazidis et al. 2004, 2010b; Skartsi et al. 2008). Almost 45% of the National Park is managed mainly for timber production (Zone B1), while it has been recognized during the last years that this specific zone is of great value for many species (Grill and Cleary 2003; Kati et al. 2004a, b, c, 2007; Korakis et al. 2006; Poirazidis et al. 2010a,b).

16.2.2 Species data

We used five datasets of indicator species groups as surrogates for the total biodiversity in Dadia NP, systematically surveyed using appropriate sampling techniques per group. Those comprised woody plants, non-woody vascular plants, amphibians, small birds and birds of prey (Kati and Sekercioglu 2006; Korakis et al. 2006; Poirazidis et al. 2009; Kret, Poirazidis, Kati, unpublished data). For each sampling plot (the number of plots was ranging from 34 to 63 depending on the indicator species group) all present species were evaluated.

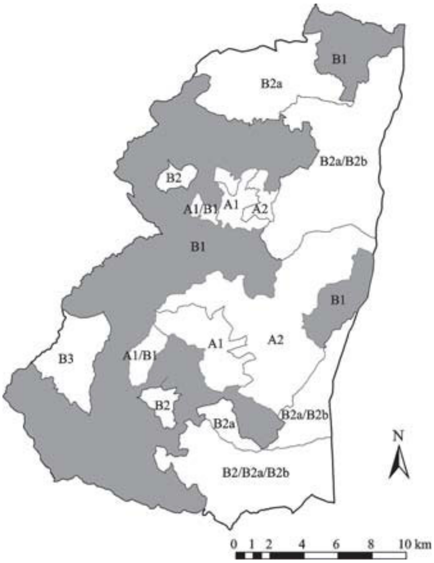

Fig. 16.1 Location and zoning of Dadia National Park, the case study area in north-eastern Greece. Zone B1 (highlighted in grey) represents the forest management area that was investigated in this study. A1, A2: strictly protected areas, B2: agroforestry area, B3: grazing land, A1/B1: forest management area that changed recently to strictly protected area.

The survey for vascular plants was based on fieldwork during the years 1999 and 2000, and the 62 sampling plots had been chosen in accordance to the survey for the Nature 2000 Network (Korakis et al. 2006). The sampling scheme for the amphibians was based on the breeding phenology of the species occurring in eastern Greece (Arnold 1978; Helmer and Scholte 1985), and each pond of the study area was visited once per month from February to July during the year 2007. The presence of amphibians was detected through a combination of visual encounter, aural and dip net surveys, during the diurnal transects in the banks of the ponds (Kret, Poirazidis, Kati, unpublished data). We excluded finally the species *Triturus cristatus* as its presence was verified at two sites, only. Similarly, a subset of the existing database for small birds (Kati and Sekercioglu 2006) was used for analysis. As the conservation value was one of the factors under evaluation, we included in our analysis only bird species that are “Species of European Conservation Concern” (SPEC; BirdLife International 2004). These included species with an unfavorable conservation status, concentrated in Europe (SPEC 2) or not (SPEC 3), as well as species with favorable conservation status, but concentrated in Europe (SPEC 4). Finally, for the small birds, the two species *Dendrocopos syriacus* and *D. medius* were used as a combined dataset due to limited detections of *D. medius*. The survey of birds of prey was based on a systematic monitoring of raptor territories that was conducted from 2001 through 2005 (Poirazidis et al. 2009, 2010b), and we pooled the data of all five years and plotted the centers of the yearly territories. The Black stork (*Ciconia nigra*), a species of conservation priority in the area (Tsachalidis and Poirazidis 2006), was included in the raptor dataset. A subset of the breeding raptor species was used in this study, and the criterion for selection was the relatively high abundance in order to produce stable habitat suitability models.

16.2.3 Habitat suitability maps and statistical analysis

Habitat suitability maps (HSM) have broad applicability within conservation biology and are of special interest to predict the distributions of wildlife species for geographical areas that have not been extensively surveyed. The methods for modeling habitat suitability can be classified into two groups: those requiring presence-only data and those requiring presence-absence data (Guisan and Zimmerman 2000). Here we prepare HSM using Ecological Niche Factor Analysis (ENFA) provided by the software BIOMAPPER (Hirzel et al. 2002). ENFA is a multivariate approach developed to predict habitat suitability based on the likelihood of occurrence of the species when absence data for the species are not available (Hirzel et al. 2002). Without absence data some limitations on the accuracy of the habitat suitability maps are possible (Hirzel and Le Lay 2008), and we reclassified the predictions into four robust levels (=bins) of suitability to settle this problem (Hirzel et al. 2006). The suitability is based on functions that define the marginality of the species, i.e. how the species mean differs from the mean of the entire area, and the specialization of the species, i.e. the ratio of the overall variance to the species variance. Marginality lies between 0 and 1, with larger values indicating that the focal species has habitat requirements that differ from the average available conditions. A high specialization value indicates that the focal species has a particular requirement for certain habitat characteristics and occupies a narrow range of variables compared to the overall range of variables within the study area (Hirzel et al. 2002).

We used 23 environmental variables, classified into four groups to derive potentially relevant predictors for species habitat selection (Table 16.1). This database contained maps stored in both a vectorial and a raster format. All species and habitat information was rasterized into a 50×50 m grid cell maps. Topographical data were directly obtained as quantitative variables. Variables quantifying land cover, landscape and potential sources of disturbance were transformed into frequency and distance variables. The forest cover categories were reclassified into pure broadleaves, mixed pine-oak and pure pine forest, but only the first two were used for the models, as the information from the third was redundant. As ENFA does not work with multinomial data, these qualitative maps were converted into several Boolean maps (i.e. one for each variable). Frequency describes the proportion of cells from a given category within a circle around the focal cell and it was derived using a circular moving window. We varied the radius of the moving window to test the performance of three different scales (200 m, 500 m and 1,000 m), but finally only the scale of 1,000 m was used as it performed better than the others. The topographical descriptors were averaged by means of a similar radius circular moving window. Spatial data analysis was conducted using ArcMap 9.0 and the Spatial Analyst extension.

Correlations between all variables of the initial pool of predictors (Table 16.1) were calculated prior to the ENFA. When two or more predictors had a correlation coefficient greater than 0.7, only the most proximal was kept (Austin 2002). Topographic and frequency environmental layers were normalized using the "box–cox" algorithm (Sokal and Rohlf 1981) and distance variables by the "square root" algorithm. There are different algorithms available in BIOMAPPER to build habitat suitability maps by ENFA (Hirzel et al. 2002) and following Hirzel and Arlettaz (2003) we used the geometric mean

Table 16.1 Environmental variables used in ENFA as predictors to define the species' ecological niche.

Environmental predictors	Scales (m)
Topography	**-**
1. Altitude	200, 500, 1000
2. 1 SD of altitude	200, 500, 1000
3. Slope	200, 500, 1000
4. Northness aspect	200, 500, 1000
Landscape/Forest attributes	**-**
5. Relative richness index	200, 500, 1000
6. Fragmentation index	200, 500, 1000
7. Frequency of broadleaves	200, 500, 1000
8. Frequency of mixed forest (Pine-Oak)	200, 500, 1000
Other ecological metrics	**-**
9. Frequency of openings	200, 500, 1000
10. Frequency of agricultural lands	200, 500, 1000
11. Frequency of permanent water	200, 500, 1000
12. Frequency of rocky area	200, 500, 1000
13. Distance to openings	-
14. Distance to agricultural lands	-
15. Distance to main river	-
16. Distance to permanent water	-
17. Distance to rocky area	-
Potential disturbance metrics	**-**
18. Frequency of paved roads	200, 500, 1000
19. Frequency of unpaved roads	200, 500, 1000
20. Frequency of urban area	200, 500, 1000
21. Distance to paved roads	-
22. Distance to unpaved roads	-
23. Distance to urban area	-

algorithm to account for the density of the observations in environmental space.

For the plants, the number of species was used as dependent variable per plot and we created two multiple regression models (one for woody plants and one for non-woody vascular plants) to predict species richness. The resulting models were transformed with the "box-Cox byte" algorithm and combined with equal weight (factor 0.5) to produce the overall "plant HSM". For each of the three groups of fauna, an overall HSM was created combining the specific HSMs by user-defined weight per species (Eastman 2001), which depended on the conservation value (Appendix). Finally, all HSMs per organism group were combined into an overall biodiversity HSM applying a new user-defined weight per group. The HSM for breeding Black vulture and Egyptian vulture (*Neophron percnopterus*) – the species with the highest conservation value in the area – were not included in the initial raptor HSM, but were used as Boolean data in a later step (see below) to highlight the priority areas for conservation of these two species.

16.2.4 Timber standing volume

We used the recent forest inventory for wood production of the local Forest Service (2006-2016) to produce quantitative maps of the distribution of standing wood volumes (basal area) (Consorzio Forestale del Ticino 2006). We used the stand level as spatial unit to summarize these data (417 sub-units of the division of managed forest, with an average size of 46.5 ± 18.9 ha). The timber volume was described as pine, oak and total volume (Consorzio Forestale del Ticino 2006). We used only the managed area of Dadia NP (zone B1), excluding the non-managed strictly protected areas (Fig. 16.1).

16.2.5 Establishment of the management scenarios

To obtain spatially explicit management plans at stand level, we reclassified the biodiversity thematic maps into four bins representing habitat suitability: (1) unsuitable, (2) marginal, (3) suitable and (4) optimal. We also reclassified the timber maps into four bins representing the standing volume: (1) minimum, (2) medium, (3) large and (4) maximum. We used the Natural Break method (ArcMap) for the biodiversity bin classification, and the four timber volume bins were defined by values of total standing timber volume of <500 m^3, 500-1,000 m^3, 1,000-2,000 m^3 and >2,000 m^3 per stand. We finally considered four possible general management actions at the stand level, in order to integrate biodiversity values into the timber management: (1) management without limitations (*free forestry*), (2) management with temporal restrictions, (3) management with temporal and spatial restrictions, and (4) management focussing on the ecological values (*ecological management*).

In this study, we implemented three management scenarios. The "*biodiversity scenario*" focused on the maximization of the biodiversity value (maximum environmental profit) in the managed forest. It was defined by the biodiversity models with each bin of habitat suitability leading to related management actions (Table 16.2), e.g. biodiversity bin 1 *"unsuitable"* leading to management action 1 *"free forestry"* and biodiversity bin 4 *"optimal"* to management action 4 *"ecological management"*. The "*timber scenario*" focused on the maximization of the economical benefits for the timber production (maximum economical profit) and was defined by the standing volume map with each bin of timber density leading to inverse related management actions (Table 16.2), e.g. timber volume bin 1 "*minimal"* leading to management action 4 "*ecological management"* or timber volume bin 4 *"maximum"* to management action 1 *"management without limitations"*. The third scenario was the "*trade off scenario*", which attempted to maximize the long-term net benefits for both biodiversity and society. The established trade off matrix considered both biodiversity and timber production at the same level, leading to the final determination of the management action for each stand (Table 16.2).

Table 16.2 Forest management categories determined by biodiversity and timber production under the scenarios *biodiversity*, *timber* and *trade off*.

Scenario		Biodiversity				Timber				Trade Off			
Timber bins		**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**	**1**	**2**	**3**	**4**
Biodiversity bins	**1**	FF	FF	FF	FF	EM	TSR	TR	FF	FF	FF	FF	FF
	2	TR	TR	TR	TR	EM	TSR	TR	FF	TR	TR	FF	FF
	3	TSR	TSR	TSR	TSR	EM	TSR	TR	FF	TSR	TSR	TR	TR
	4	EM	EM	EM	EM	EM	TSR	TR	FF	EM	EM	TSR	TSR

FF: free forestry, TR: temporal restrictions, TSR: temporal and spatial restrictions, EM: ecological management. Biodiversity bins: 1 unsuitable, 2 marginal, 3 suitable, 4 optimal; timber bins: 1 minimal, 2 medium, 3 large, 4 maximal.

We applied each scenario to each biodiversity data set as well as to the overall biodiversity HSM. For each scenario at the last step, we used the suitable and optimal areas for Eurasian black vulture and Egyptian vulture as Boolean variables as such: suitable and optimal areas for Eurasian black vulture were upgraded to the Management action "4" (ecological management) and for Egyptian vulture to the Management action "3" (temporal and spatial restrictions).

16.3 Results

In the following section, we present the resulting maps regarding habitat suitability, timber standing volume, and forest management categories. We further present the evaluation of the effectiveness of the different management scenarios in conserving biodiversity.

16.3.1 Habitat suitability maps

The species richness of vascular plants (351 plant species in 63 plots) was modeled using the eco-geographical variables as independent variables. The resulting regression model for woody plants was "Y = 4.3 + 2.01 northness – 10.29 frequency of openings + 2.53 frequency of mixed forest + 0.001 frequency of rocks + 0.001 distance to agricultural lands", while for non-woody plants it was "Y = 30.4 + 0.24 slope – 0.23 relative richness index + 5.02 frequency of mixed forest". Both models were significant at the level p=0.05 and were combined equally to the overall HSM for plants (Fig. 16.2a)

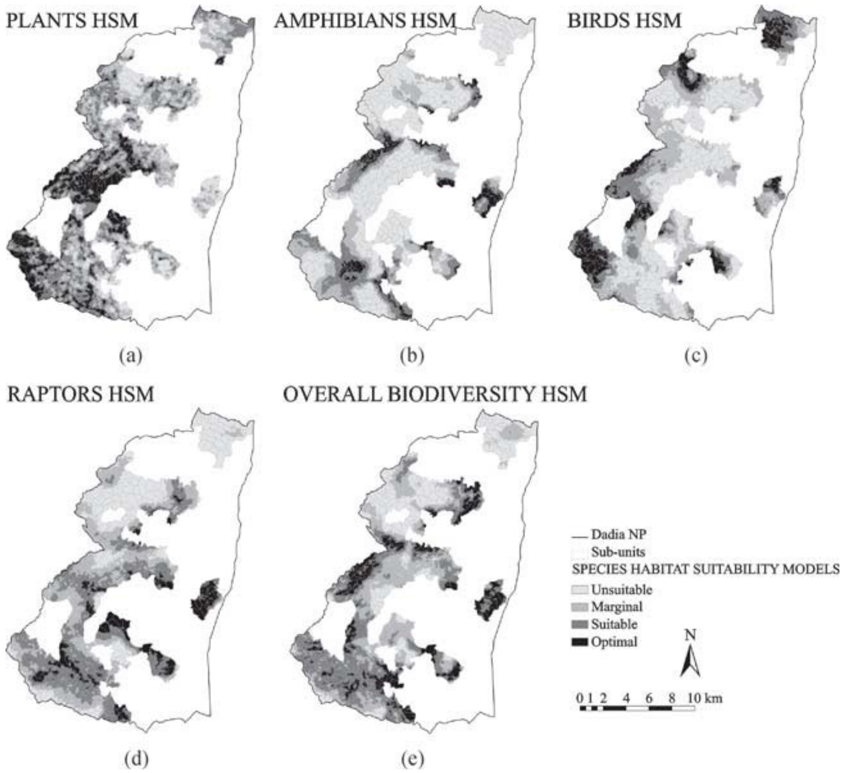

Fig. 16.2 Habitat suitability maps for (a) plants, (b) amphibians, (c) small birds, (d) raptors and (e) overall biodiversity in Dadia NP.

Amphibians (10 species in 53 plots) showed a pronounced specialization for certain habitats as their mean global marginality was 0.94 (range 0.63-1.35) and their specialization was 4.37 (range 1.59-12.56). Both groups, small birds and raptors, showed intermediate sensibility and differentiation of habitat use. The mean global marginality of small birds was 0.70 (range 0.35-1.05) and the specialization was 3.23 (range 1.13-6.93). For the raptor HSM, ten species of breeding raptors plus the Black stork had a relative abundance that enabled stable models. The mean global marginality for raptors was 0.63 (range 0.17-1.64) and the specialization was 2.05 (range 1.03-6.05). Finally, separate HSM were created for each taxon-group of animals (Fig. 16.2b,c,d) using species specific weights (Appendix). The combined overall biodiversity HSM resulted (Fig. 16.2e), applying the weights of 0.5 to raptors HSM, 0.25 to amphibians HSM, 0.15 to small birds HSM, and 0.1 to plants HSM.

16.3.2 Standing volume distribution maps

The mean pine wood volume was 1,533.2 $m^3 \pm$ 1,424.1 (sd) per stand, with a maximum value of 7,380.8 m^3 while the mean oak wood volume was 731.5 $\pm$ 658.1 m^3 with a maximum value of 4,785.3 m^3. The total timber volume ranged from 69 to 8,094 m^3 (Fig. 16.3), while the total volume per ha was 49.2 $m^3 \pm$ 26.2 and ranged per forest stand from 2 m^3/ha to 131 m^3/ha.

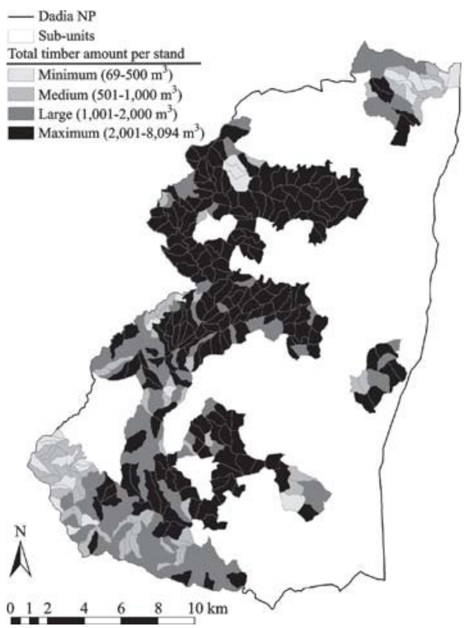

Fig. 16.3 Total timber standing volume of the managed forest area in Dadia NP.

16.3.3 Establishment of the management scenarios

We produced three thematic maps of spatially explicit management plans, based on the desired forestry policy in the management area (Fig. 16.4). At the timber scenario, where conservation priorities are considered exclusively in areas without economical value for timber, only 6% of the area was proposed for ecological management and 46% for free forestry. On the other hand, in the biodiversity scenario, where the most suitable areas remain unexploited, 18% of the managed forests were proposed for ecological management and

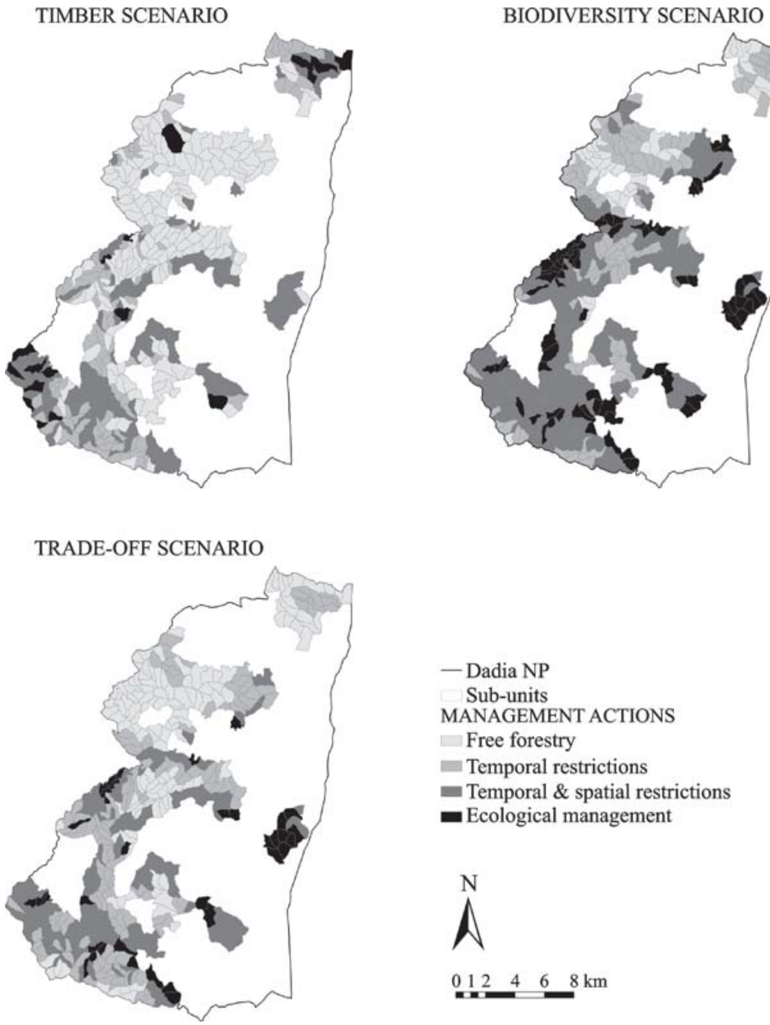

Fig. 16.4 Spatial forest management plans, presenting the distribution of the four forest management categories under the timber, trade off and biodiversity scenario.

11% for free forestry. The trade off scenario, taking into account both timber and biodiversity, lies in between, proposing 9% of the area for ecological management and 32% for free forestry.

The trade off scenario served both ecosystem services, biodiversity values and timber production (Fig. 16.5). In this scenario, 91% of the area with low suitability for biodiversity (*bins unsuitable and marginal*) was covered by the management category "free forestry", while the areas of high suitability for biodiversity (*bins suitable and optimal*) were intensively covered by the management categories "temporal and spatial restrictions" (47%) and "ecological management" (25%). For comparison, in the timber scenario, only 60% of the low biodiversity area was dedicated to free forestry and more importantly only 42% and 4% of the high biodiversity areas were classified as "temporal and spatial restrictions" and "ecological management", respectively (Fig. 16.5).

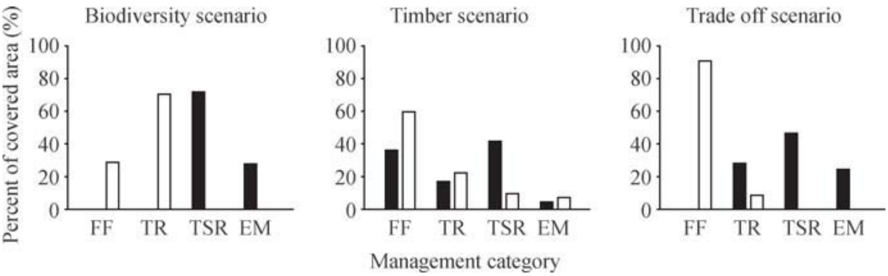

Fig. 16.5 Management and conservation of areas of differing suitability of biodiversity under the scenarios "Biodiversity", "Trade off", and "Timber". Black bars: forest stands of high suitability for biodiversity (bins suitable and optimal), white bars: forest stands of low suitability for biodiversity (bins unsuitable and marginal); FF: free forestry, TR: temporal restrictions, TSR: temporal and spatial restrictions, EM: ecological management.

16.4 Discussion

In the following section we discuss the need of integrating biodiversity into forest management, and several aspects regarding multi-taxa indicators, decision support systems, and the scenarios applied in this study.

16.4.1 Integrating biodiversity into forest management

New environmental policies call for increased attention to biodiversity issues in forest management planning, given that the loss and fragmentation of mature forest together with the structural diversity decline have threatened forest-dependent species (Andrén 1994; Siitonen 2001; Thompson et al. 2003; Angelstam et al. 2004; Poirazidis et al. 2004). Sustainable forestry and deadwood supply have recently emerged as two of the twenty-six headline indicators towards halting further biodiversity loss in Europe (European Environmental Agency 2007). In this frame, the approach developed in this study provides a useful tool for forest managers. We established biodiversity priority areas into the managed areas, providing a guideline for effective management strategies. We also developed habitat suitability models based on environmental features and we identified habitat associations that provide an important source of information for general habitat management issues. These models quantifying relationships between species and their habitats are considered nowadays one of the most efficient tools for forest management (Edenuis and Mikusinsky 2006). Sustainable forest management should be efficient, satisfying on one hand conservation goals while minimizing on the other hand socio-economic costs and the area removed from timber production (Pressey et al. 1997; Montigny and McLean 2005).

16.4.2 Species selection and multi-taxa indicator species

We modeled in this research habitat suitability for several groups of organisms, using totally 351 taxa of vascular plants, 10 species of amphibians and 23 species of birds for the assessment. For a successful use of habitat suitability models in forest biodiversity management an appropriate selection of species is required and multi-taxa bio-indication has several advantages (King et al. 1998; Angelstam et al. 2004; Rempel et al. 2004; Wrbka et al. 2008). Ecologically different taxa can show different patterns of biodiversity and it is assumed that even several species of one single taxon or guild are not enough for being representative (Schulze et al. 2004; Billeter et al. 2008; Cabeza et al. 2008). Also Edenius and Mikuszinski (2006) stressed the need for multi-species selection procedures in their recent review on the use of HSM in forest management. They have found only one study (out of 55 reviewed ones) that followed a multi-taxa approach, and only five papers of the review (9%) could be attributed to indicator species in the species selection procedure.

The indicator species approach has been criticized on conceptual grounds, such that no species share the same ecological niche, as well as on empirical grounds, i.e. untested or unverified relationships between the indicator and the species or species groups that the indicator supposedly covers (Lindenmayer et al. 2000; Rempel et al. 2004; Roberge and Angelstam 2004; Edenius and Mikuszinski 2006). In our study we used vascular plants, amphibians, small birds and raptors as indicator groups in habitat suitability models. Recent research confirmed that plants and birds are well performing surrogate taxa for overall biodiversity in Dadia NP (Kati et al. 2004b; see also Sauberer et al. 2004 for a Central European case study). Amphibians, due to their very specific habitat needs and life cycle, are important for being complementary and good indicators of habitat matrix permeability (Ray et al. 2002; Kati et al. 2004a, 2007; Cabeza et al. 2008). Raptors are top predators; requiring enough prey, large areas and limited disturbance, they indicate ecosystem health and perform well as indicators of biodiversity (Sergio et al. 2005; Sekercioglu 2006; but see also Cabeza et al. 2008). Raptors are also focal species of conservation efforts in the reserve, as their populations in Dadia NP are of regional importance (Poirazidis et al. 2004, 2007, 2010b; Skartsi et al. 2008).

16.4.3 Decision Support Systems and comparison of scenarios

Concerning limited funding and limited data sources, adaptive management is a useful tool for fast implementations (Angelstam et al. 2004; Duff et al. 2009). Ideally, an active adaptive management approach with iterated assessment and corrective action should be applied through continuous mutual learning by scientists, policymakers, managers and other actors until the targets are reached (Simberloff 1999; Brown et al. 2001; Angelstam et al. 2004; Steffan-Dewenter et al. 2007; Duff et al. 2009). The three scenarios, presented in this case study, are adaptive in terms of their main objectives and regarding their simplicity. The timber scenario is a simple approach to integrate conservation of biodiversity into forest management when timber production has the main priority. In this scenario more restrictive conservation management will be done only in forest stands with little timber. The biodiversity scenario can be followed when conservation is the key issue. Restrictions are proposed, where habitat suitability reaches maximum values, the performance regarding conservation is optimal, but the socio-economic benefits remain totally unused in forest stands with a high level of biodiversity. The trade off scenario as an alternative solution proved very useful to integrate timber extraction and nature conservation and an optimization of the benefits for society and biodiversity could be achieved. Compared with the timber scenario, free forestry is encouraged where habitat suitability is lower but forest stands of high biodiversity have more restrictions. A decision support system can be an effective mechanism to support technological and managerial decision making (Malczewski 2006) as it can combine multiple sources of information (models and data) into a single system that provides a tool to manipulate the information. With these capabilities, it supports decision makers in cognitive tasks that involve choices, judgment and decisions, in recognizing needs and identifying objectives, as well as in formulating and evaluating different courses of action (Garcia and Armbruster 1997). In the case of sustainable forest management, these actions are forest management scenarios, i.e. collections of rules and strategies regarding harvest scheduling and forest regeneration (Van Damme et al. 2003).

Timber harvesting and conservation of biodiversity are not necessarily mutually exclusive and some rules of temporal and spatial restrictions can optimize their coexistence (Lõhmus 2005; Brown et al. 2007). Integrating different data sources to a decision support system for spatial forest management planning can increase clearly the sustainability of forest management. Viable populations of indicators species and a high level of biodiversity can be maintained, without losing the socio-economic benefits of professional timber production. At the local scale, a selective targeting approach that identifies forest stands of potential high biodiversity and nature conservation value is essential. Once identified, these areas can be highlighted for inclusion in future local targets and management prescriptions altered accordingly (Bayliss et al. 2005). As maps of habitat suitability were initially created for individual species, our approach provides also a further resource for species specific conservation management. We recommend applying habitat suitability modeling to selected groups of indicator organisms to develop spatial management plans for managed forests. This enhances the sustainability of the management and promotes monitoring and evaluation of its effects on wildlife. The inclusion of further taxa as indicators of overall biodiversity into the existing decision support system is a prerequisite for continuous improvements of a sustainable forest management.

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Appendix

Selected species used for the habitat suitability models for amphibians, small birds and raptors, and user-defined weights (adding up to the value of 1 per group). SPEC values for avian "Species of European Conservation Concern" (BirdLife International 2004): 2- "concentrated in Europe and with an unfavorable conservation status"; 3- "not concentrated in Europe, but with an unfavorable conservation status"; 4- "concentrated in Europe, but with a favorable conservation status".

For the list of the 351 plant species, used for this analysis see Korakis et al. (2006), available by the authors.

Species	-	SPEC	Weight factor
Amphibians	**-**	**-**	**-**
Fire Salamander	*Salamandra salamandra*	-	0.2
Yellow-bellied Toad	*Bombina variegata*	-	0.15
Common Toad	*Bufo bufo*	-	0.1
European Green Toad	*Bufo viridis*	-	0.1
Common Spadefoot	*Pelobates fuscus*	-	0.1
Smooth Newt	*Triturus vulgaris*	-	0.1
European Tree Frog	*Hyla arborea*	-	0.1
Marsh Frog	*Rana ridibunda*	-	0.05
Balkan Stream Frog	*Rana graeca*	-	0.05
Agile Frog	*Rana dalmatina*	-	0.05
Small birds	**-**	**-**	**-**
Woodchat Shrike	*Lanius senator*	2	0.1
Ortolan Bunting	*Emberiza hortulana*	2	0.1
Black-headed Bunting	*Emberiza melanocephala*	2	0.1
Woodlark	*Lullula arborea*	2	0.1
Corn Bunting	*Milandra calandra*	2	0.1
Bonelli's Warbler	*Phylloscopus bonelli*	2	0.1
Green Woodpecker	*Picus viridis*	2	0.1
Olivaceous Warbler	*Hippolais pallida*	3	0.05
European Bee-eater	*Merops apiaster*	3	0.05
Orphean Warbler	*Sylvia hortensis*	3	0.05
Red-backed Shrike	*Lanius collurio*	3	0.05
Middle Spotted Woodpecker	*Dendrocopos medius*	4	0.05
Syrian Woodpecker	*Dendrocopos syriacus*	4	0.05
Raptors	**-**	**-**	**-**
Eurasian Black Vulture	*Aegypius monachus*	1	Special category
Egyptian Vulture	*Neophron percnopterus*	3	Special category
Golden Eagle	*Aquila chrysaetos*	3	0.3
Lesser Spotted Eagle	*Aquila pomarina*	2	0.2
Booted Eagle	*Hierraetus pennatus*	3	0.2
Black Stork	*Ciconia nigra*	2	0.1
Short-toed Eagle	*Circaetus gallicus*	3	0.1
Goshawk	*Accipiter gentilis*	-	0.05
Honey Buzzard	*Pernis apivorus*	-	0.03
Common Buzzard	*Buteo buteo*	-	0.01
Sparrowhawk	*Accipiter nisus*	-	0.01

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