Managing Forest Ecosystems

Klaus von Gadow
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Editors

Designing Green Landscapes

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Managing Forest Ecosystems

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Aims & Scope:

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

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

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

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

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Designing Green Landscapes

Foreword by Klaus von Gadow and Timo Pukkala

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Foreword

While the natural resources of the earth continue to diminish, “Green Landscapes” are being called upon to produce an increasing range of goods and services. A Green Landscape is a rural expanse of scenery that may comprise a variety of visible features. This book focuses on forested landscapes, although much of the theory and most of the practical applications are valid for any area of land. In many regions of the world, people depend on forests for their livelihood and well-being. Forests provide multiple services, – benefits generated for society by the existence of certain forest ecosystems and their attributes. The value of these benefits is often only recognised when they are lost after removal of the trees, resulting in flooding, loss of income and declining species diversity.

Forests provide multiple services. However, the amount and quality, and the particular mix of these services depend on the condition of the resource. Landscape design is a proven way to ensure that certain desired benefits will be available in space and time. It provides the foundation and an essential starting point for sustainable management.

This volume, which forms part of Springer’s book series *Managing Forest Ecosystems*, presents state-of-the-art research results, visions and theories, as well as specific methods for designing Green Landscapes, as a basis for sustainable ecosystem management. The book contains a wealth of information which may be useful to company management, the legal and policy environment and forestry administrators. The volume is subdivided into four sections.

The first section presents an introduction which clarifies the context and sets the scene, in particular focusing on tree landscapes. Virtually all forests are utilized by humans and many are subject to some kind of planned management. Foresters harvest trees to utilize the timber; they change the species composition and tree size distribution to attain some desirable structure; and they adapt rotation ages to increase the runoff from water catchments. Clearly, the dynamics of a forest ecosystem is not only determined by natural processes, but to a considerable extent by human interference. This calls for proper designs. The second section which is entitled “Assessing the Landscape” features three contributions. The focus is on landscape metrics, resource assessment and optimized data collection. Forest design needs spatially explicit data which accurately reflect the current condition of the resource at any given point in time. Landscape design requires continuous updating and keeping track of management activities and effective implementation of the "multiple path" approach to managing green landscapes depends on rigorous quantification of the current composition and structure of the landscape. The emphasis of the third section is on the numerical analysis of landscape design, which includes the specification of alternative treatment schedules for individual spatial units and the coordination of local and global objectives. The most important computational applications of mathematical programming, methods for integration of multiple services into landscape design, decentralized forest planning models within a cellular automata framework and approaches to analyzing the effects of climate change are among the highlights of this section. The fourth and final section deals with computer graphics and visualisation. These techniques support participatory planning and have become fundamental to landscape design. Participatory planning is a key to involve the public in landscape planning and the impacts of future silvicultural operations can be made visible with adequate tools.

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Part I
Introduction

Adaptive Design of Forested Landscapes

Klaus von Gadow, Tuula Nuutinen and Seppo Kellomäki

1 Introduction

The threat of climate change and associated environmental hazards are among the problems confronting political institutions and the scientific community today. Scientists are often asked to find quick solutions to environmental threats and a typical response is progressive differentiation and specialisation within the disciplines, or reorganisation of research institutions. Traditional boundaries between disciplines vanish and new fields of research are continually emerging. They operate individually or in varying associations with other fields, thus creating a wafting, "ameboid" landscape of science that has no well-defined structure. This process generates fresh ideas, new technologies and sometimes useful theories which may improve understanding of complex systems. Forest science is a good example.

In the context of this contribution, forested landscapes refer to landscapes which are characterized in some way by trees. They may include open and closed forests, savannah woodlands and commercial timber plantations. Forests provide multiple services, and their value is often only recognised when these services are lost after deforestation, resulting in soil erosion, flooding, reduced quality water, loss of species diversity and increased greenhouse gas emissions (Stern, 2006). Forest Services are benefits generated for society by the existence of certain forest types and their attributes. Water and nitrogen processes (Laurén et al., 2005); water yield from catchments (Bosch and Gadow, 1990); habitat for certain animal species (Kurttila et al., 2002) and carbon sequestration are determined, not only by natural processes, but primarily by forest management.

The natural processes providing different services are controlled by climatic and edaphic factors, with a consequence that forest services are vulnerable to changes in environmental conditions. For example, the growth of boreal forests in northern Europe is currently limited by a short growing season, low summer temperatures and short supply of nitrogen (Kellomäki et al., 1997). Therefore, the climate change

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with the increase in temperature may prolong the growing season and enhance the decomposition of soil organic matter, thereby increasing the supply of nitrogen (Lloyd and Taylor, 1994; Oren et al., 2001). This may substantially enhance forest growth, timber yields and accumulation of carbon in the boreal forests. Impacts on other services like on water yield are also expected (Kellomäki and Väisänen, 1995).

1.1 Natural and Planted Forests

Forest biomes are found in a variety of environmental conditions and have been subdivided into distinct types. The simplest classification distinguishes three types: the *tropical forests* which are characterized by a great diversity of species, occurring between latitudes of 23.5 degrees N and 23.5 degrees S; the *temperate forests* which have a growing season of 140–200 days and occur in eastern North America, northeastern Asia, and western and central Europe; and the *boreal forests* which represent the largest terrestrial biome, occurring between 50 and 60 degrees N in Eurasia and North America, with a growing season of about 130 days. Forests grow under a great variety of environmental conditions and it is not surprising that such a simple classification cannot portray the great diversity of forest ecosystem formations. Numerous attempts have been made by geographers, botanists, ecologists and foresters to capture this diversity using finer classifications.

Most authors differentiate six types:[1] the t*ropical rainforests* which are found in the Amazon lowland, central lowlands of Africa, and a belt from Sumatra, Indonesia to the islands of the western Pacific; *tropical deciduous (including Monsoon) forests* occuring in southern Asia, Myanmar, Thailand and Cambodia, and also in south-central Africa and South America bordering on the equatorial rainforest; *Mediterranean woodlands* which are found in a narrow coastal belt around the Mediterranean Sea, in the central and southern California coastal region, in central Chile, in the Cape Province of South Africa and in western Australia; *Temperate deciduous forests* which are native in eastern North America, Western Europe, and in eastern Asia; *Temperate rainforests* which are found at latitudes of both the Northern and Southern Hemisphere where precipitation exceeds 150 cm/year and falls during at least 10 months and where temperatures are cool throughout the year, but always above freezing; *Boreal Forests*, requiring a cold climate and adequate moisture, are found in two broad continental belts in North America, Europe and Siberia, and extend into lower latitudes at higher altitudes.

These ecosystems are not virgin forests, but subject to human use. They are managed to satisfy a great variety of objectives and to provide multiple services. Natural forests have often been replaced by fast growing commercial timber plantations and it has been estimated that worldwide, planted forests currently total about 123 million hectares. Tree planting for industrial plantations began at the end of the 19th century in some Southern Hemisphere countries, like South Africa and Australia to

[1] *vide* Folch and Camarasa (2000); Matthews et al. (2000), Hendrick (2001).

replenish scarce natural timber resources. It continued in earnest during the great depression, mainly to provide jobs, and then again after world war II in countries such as Australia, New Zealand and the United States to improve industrial wood supplies. During the 1960s, large-scale plantation programs were launched in many tropical and subtropical countries and by the year 2000 the global rate of tree planting for industrial purposes was estimated at 4.5 million ha per year (Cossalter and Pye-Smith, 2003). Plantations were established to stabilize domestic wood supplies, to provide jobs, to combat soil erosion and to reduce the pressure on the natural forest resources.

1.2 Human Use

Virtually all forests are utilized by humans and many are subject to some kind of planned management. Foresters modify the soil nutrient status by adding fertilizer; they harvest trees to utilize the timber and to improve the growing conditions of the remaining ones; they change the species composition and tree size distribution to attain some desirable structure; they may kill trees to increase the amount of deadwood and adapt rotation ages to increase the runoff from water catchments. Clearly, the dynamics of a forest ecosystem is not only determined by natural processes, but to a considerable extent by human interference. Social and ecological processes are intertwined and not all human activities have desirable outcomes. Socio-ecological systems are characterized by constant change and uncertainty, and sustainable use requires that the system is resilient. This means that societies must have the capacity to anticipate undesirable impacts and to develop adequate mechanisms of response (Folke, 2006).

The traditional mechanisms of response are based on the understanding that longterm forest development will follow some predefined “optimum” course. A typical example are the *rotation forest management* (*RFM*) systems which follow a sequence of standard silvicultural treatments from planting to the final harvest. It can be assumed that at least 280 million ha of forest land (Solberg’s categories M3, M4 and M5) and possibly more than 350 million ha (including parts of M2 and M1) are managed in some form of *RFM* system (Solberg, 1996).

Selective harvesting, also known as *Continuous cover forest* (*CCF*)[2] management, has a long tradition, but the methods of calculating sustainable harvest levels have been practiced for instance in a few temperate rain forests (Laughton, 1937; Breitenbach, 1974; Seydack et al., 1995), in North American deciduous forests (Leak, 1964, 1965; Hansen et al., 1987; Guldin, 1991) and in European forests (Susmel, 1980; Schütz, 1994). The stand age is undefined and forest development does

[2] The terminology is not always very clear: *CCF* management is characterized by selective harvesting and the use of natural regeneration; selective harvesting techniques are also practiced in the so-called *Near-Natural Forest Management*, favouring site-adapted tree species and some kind of “natural forest management”.

not follow a cyclic harvest-and-regeneration pattern. Instead, it *oscillates* around some “ideal” level of growing stock. This ideal structure is usually defined by some assumed optimum diameter distribution and/or species composition (Meyer, 1933). *CCF* systems are often preferred by private forest owners and may be attractive for public forest administrations in regions where environmental concerns and habitat conservation are important (Pommerening, 2001). Two examples of such CCF systems demonstrate the assessment of optimal sustainable management and harvest levels. The first refers to mediterranean pine forests (Trasobares and Pukkala, 2005), the second to a Grevillea robusta agroforestry system with maize in Kenya (Muchiri et al., 2002).

2 Traditional Models of Sustainable Forest Management

To successfully maintain an industry based on timber products, forest planning must ensure that there are always stands at the right stage of development and in sufficient number to yield the desired product mix from the forest. For planning purposes it is necessary (a) to produce accurate descriptions of future management activities and (b) to project the development of the growing stock and other important forest attributes. In the past this was achieved by means of yield tables (Schober, 1978). A yield table estimates the development of the remaining stand and the volume of the removed trees for forests which are managed in some standard way. A yield table does more than merely estimate timber yields. It produces a description of future management activity, expressed in terms of the volumes removed during thinning operations. It also projects the development of a forest through time, for a given site productivity and type of silviculture.

The yield table format has remained surprisingly constant during the past 200 years (Paulsen, 1795; Schwappach, 1890; Wiedemann, 1949; Schober, 1978). This may be an indication that the yield table format was a very practical design. It met the requirements for effective silvicultural planning, especially after the introduction of a system of tree classes and standard thinnings. The system of pre-defined thinning grades and yield tables, in combination with periodic inventories of the growing stock, represented a simple and effective planning framework. Standardized silviculture in combination with the model of the Normal Forest was the key to sustainable harvest control.

2.1 The Model of the Normal Forest

The model of the *Normal Forest* is an idealized standard which allows comparisons between the actual and the normal age class areas, growing stock volumes, growth rates and harvest volumes (Hundeshagen, 1826). The model is defined by two conditions: (a) a yield model estimating the development of the growing stock over age and (b) a rotation. It can be used as a simple tool to generate scenarios which provide useful information for timber companies. This application was demonstrated, for example, by Clutter et al. (1983) for pine plantations in Southern Georgia, USA. The growing stock increases with increasing rotation age, and this affects the annual sustainable cutting area, the total forest area required to deliver a certain quantity of roundwood, the total growing stock volume and the mean annual increment (MAI). The age of culmination of the MAI represents the minimum-landbase-rotation, i.e. the rotation where the area which is required to continuously supply a pulpmill with a given amount of timber is at a minimum.

2.2 Age Class Simulation and Area Change Models

The model of the normal forest can be used for comparisons between a current condition and some normal standard. Its value as a dynamic scenario model is limited however. Quite useful, though almost equally straightforward, are methods based on age class simulations. In the simplest simulation approach, the forest area is subdivided into *m* age classes and *n* felling periods. The available timber volume in the i'th age class $(\mathrm{i} = 1..\mathrm{m})$ is estimated using a suitable yield model. The planned total harvest volume for the jth felling period $(\mathrm{j} = 0..\mathrm{n})$ is prescribed. The simulation, which may be implemented in an *Excel* spreadsheet, shows the effects of a particular harvest strategy. An example was presented by Gurjanov and Gadow (2005) for the Lissino forest near St. Petersburg.

Obviously, the age class simulation method involves considerable aggregation over growing sites, forest types and management regimes and the predictions have to be interpreted with the necessary caution. However, an age-class simulation is often the only feasible way to predict the dynamic development of a forest resource for large timber growing regions with relatively uniform conditions.

Area change models, also known as *transition matrix models*, have been used by a number of authors for generating a scenario of forest development based on age class transitions. We may define $\mathrm{p_i(n)}$ as the probability that a randomly selected forest area is in state i at time n and t_{ji} as the conditional probability that state i moves to state j within a given time step, i.e. as the conditional probability that j will be reached provided the system is currently in i. Let $t_{j,1}$ be the probability that a forest stand which belongs to the j'th age class is harvested and thus moves to age class 1. Then $t_{j,j+1}$ is the conditional probability that a stand which belongs to the j'th age class survives and grows into the $(\mathrm{j}+1)$'th age class. One of the earliest and possibly the most famous application is Suzuki's *Gentan* model (Suzuki, 1971). Area change models have been used for timber supply projections, especially in Japan (Konohira and Amano, 1986) and in Europe (Kurth and Dittrich, 1987; Kouba, 1989). Another well-known forestry application is the EFISCEN model which was originally proposed by Sallnäs (1996) and later developed and used by Nabuurs et al. (1997).

The area change approach may have some potential as a scenario modelling tool, e.g. for forecasting the development of age structures on a regional and national scale. However, some work needs to be done to predict the transition probabilities more accurately (Randall and Gadow, 1990). The transition probabilities are not independent of the current age class distribution (state vector) and this may be one of the main problems associated with their use. Area change models may, however, be suitable for projecting the development of spatial patterns which are affected by natural processes (growth, fire or wind damage) and forestry activities (plantings, thinnings). However, credible applications are limited due to the fact that transition probabilities are usually not available.

2.3 Multi-period Harvest Scheduling with Linear Programming

Mathematical programming has a solid tradition in forest planning. The models include an objective function that must be maximized or minimized and a set of linear or non-linear constraints. In multi-period harvest scheduling models, the state of the forest at period $t + 1$ is a linear function of the forest at period t and the areas cut in period t. The objective may be to maximize the harvest volume or the net present value, or to minimize the cost of supplying a mill. The constraints are typically formulated such that a smoothing-out of production levels over time is achieved and at the same time a sustainable harvest can be maintained.

A complete survey of mathematical programming applications to timber harvest scheduling is beyond the scope of this paper. The interested reader may refer to relevant presentations by, for example, Clutter et al. (1983); Leuschner (1990); García (1990) and Buongiorno and Gilless (2002).

2.4 Residual Normalities in Continuous Cover Forests

Selective harvesting systems have a long tradition and similar methods have been used in different forest types. The most common approach involves recurring visits to a given stand and to harvest the accumulated increment at each visit. The principle resembles the normal forest. First, we have to define r, the number of years between harvests. To ensure sustainable yields, the forest is then subdivided into land parcels each covering one r'th of the total area. Each stand represents a state which is characterized by the number of years that have passed since the last harvest.

The main problem in a continuous cover forest is to determine which trees are to be removed and which are to remain. The decision may concern tree size or a combination of tree size, species and timber quality. The most common methods assume some standard or "normal" residual forest structure. The residual structure may be defined by a *normal* residual diameter distribution or a *normal* residual basal area distribution. The traditional *Plenterwald* harvesting systems practiced in Switzerland, France, Italy, Slowenia, Germany and in the Southern USA (Guldin, 2002), define an optimum forest structure by using an inverse J-shaped diameter distribution model. The target distribution is re-established by periodic removal of trees in the different diameter classes. In theory, the approach is very simple and logical. By comparing the real and the ideal diameter distributions, it is possible to determine the number of trees that should be harvested in each diameter class. Meyer

(1933) proposed the function $\ln(N) = \alpha - \beta \cdot D$ for specifying the normal residual diameter distribution. In a group selection (*Femel*) system which is characterized by group harvesting and gap regeneration Meyer's β parameter assumes values ranging from 0.08 to 0.15, depending on the maximum diameter of harvestable trees. In a single tree selection (*Plenter*) system the β-values are usually much lower, ranging between 0.05 and 0.07. In mixed fir, spruce and beech forests of the Trentino region of the Italian Alps, trees are harvested with reference to the model defined by Susmel (1980).

The definition of some ideal diameter distribution may simplify management, but usually there is no proof that a predefined structure is either economically sound or ecologically reasonable (Cancino and Gadow, 2002). There is no generally accepted ideal distribution of tree sizes, but a rather wide range of distributions found in stable *Plenter* and *Femel* forests (see Mitscherlich, 1952). Uneven-aged forests, natural or managed, may be found within a wide range of combinations of species mixtures and size classes.

An alternative criterion for defining the residual forest structure is a basal area distribution norm. An example of this approach is found in the *Northern Hardwoods*, a multi-species forest extending from the central Appalachians northwards towards Canada and from the Atlantic ocean westwards to Minnesota (Eyre, 1980). Hansen et al. (1987) proposed a residual basal area of 19.8 m^2/ha for a cutting cycle of 15 years and 19.8 m^2/ha for a 30-year cutting cycle.

Tree species are not specifically accounted for in the standards based on a specific residual diameter distribution or a residual basal area norm and this is sometimes seen as a shortcoming of the method. Tree *species and size* were simultaneously considered in a system developed for the indigenous evergreen forests occurring in the Southern Cape coastal belt of South Africa, between Mosselbay and Humansdorp, which are known as the *Knysna forests*. They represent a natural resource of great historical and scientific importance (Laughton, 1937; Breitenbach, 1974). In the old Knysna forest management model, developed during the 1970's, the first step involves a classification of forest types according to the moisture regime which represents a natural classification criterion affecting forest structure, productivity and species distribution. The second step involves a classification of different tree species and size classes on the basis of their current and future potential. Finally, a normal residual basal area distribution is defined for each forest type and tree class (Gadow and Puumalainen, 2000). The method has many practical advantages, but again, there is no generally accepted ideal distribution of basal areas over tree classes. And this is a major disadvantage of any method based on a silvicultural standard which homogenizes the landscape to a predefined "normality".

3 Standardization and Specialisation

Standards are agreements on technical specifications which are used as rules or definitions to facilitate manufacturing, trade and communication. In forestry, standard units are used to facilitate data assessment, analysis and reporting. A unit is a particular physical quantity, such as a meter or an inch, defined and adopted by convention. The current global standard is the International System (SI) of units, which is a modern form of the metric system. There are standard units of length, temperature, mass and time and these are needed to make comparisons and to express the numerical value of specific properties of an object, such as a tree. Standards are used in forest mensuration and it has been suggested that there is a need for standardisation in the development and documentation of forestry software, such as tree growth modelling systems (Pretzsch et al., 2002).

However, standardisation is not always meaningful. Foresters have been developing silvicultural standards as a universally agreed upon set of instructions for managing a forest. There are numerous examples of treatment programs prescribing a series of particular silvicultural events for the entire life of a forest from planting to the final harvest. The optimization of such standard treatment schedules was an important research topic during the 1960's until the 1980's (see for example, Brodie et al., 1978).

Standardized silviculture creates uniform habitats and this may reduce the diversity at stand and forest levels with undesirable effects for species which require specific habitat conditions. Furthermore, tree species choice and silviculture are influenced by changing policies, and changing economic and environmental conditions. Therefore, silvicultural standards which are designed for conditions which are assumed to remain constant from the time of regeneration may be outdated before the trees reach maturity.

A new idea leading to a change in silvicultural strategy may be generally accepted following debate and communication (Healey, 1992; Leskinen, 2004). But this acceptance is not permanent, because alternative policies are discussed all the time. One of these new proposals may eventually gain broad acceptance again, replacing the previous strategy. Amling (2005) could demonstrate such a succession of forest policies for the period 1945 to 2005 in the public forests of *Nordrhein-Westfalen* in Germany. A similar cyclic change of forest policy occurred in the former German Democratic Republic (Koch, 2005). In both cases, the phase-length of the cycles of policy changes proved to be much shorter than the lifespan of the trees. The changes may affect the type of harvesting practice (clearfelling vs selective harvesting), species selection (deciduous species vs conifers) and preferred forest structures (even-aged monocultures vs uneven-aged multi-species forests). It is not surprising, therefore, that many forests are in constant transition from one policy to another. This is a paradox, because, in theory, forest management has always been committed to longterm strategies. In reality, the changes of silvicultural practice are more frequent than is generally assumed. This fact is often ignored.

The strategic significance and economic benefits of industrial standardization are generally accepted. However, when designing the future of a forested landscape, the negative effects of using standard silvicultural programs may outweigh the advantages. Experience has shown that neither the need for certain products and services nor the growth-relevant conditions are constant. Consequently, it is necessary to continuously adapt silvicultural methods, often several times within the life of a tree. This process is sometimes referred to as "transformation forestry". The transformation never ends. It seems to have become a common feature of forest dynamics.

It should be pointed out in this context that spatial differentiation of forest management activities is required in addition to temporal differentiation. One reason which has been mentioned already, is the creation of a mix of habitat conditions. In addition, the provision of multiple services on the landscape level may be more efficient if individual stands or clusters of stands "specialize" in the production of particular services. For example, provision of recreation amenities, timber production, or maintenance of deadwood may be more effective if these activities are distributed wisely within the landscape. How much specialization is required (and how much joint production) depends on the trade-offs between different services at the stand level. In principle, standardization may decrease the production efficiency of multiple services in the forest.

4 The Multiple Path Concept

Conditions are changing and silvicultural treatment programs which were designed to provide an optimum mix of services over an entire rotation may soon turn out to be undesirable. Thus, a basic challenge is continuous adaptation to changing environmental and social circumstances. This is not trivial, because forest development is slow and rapid changes of the species composition, size distribution and many other features influencing the quantity and quality of services, are usually not possible. One approach which involves continuous assessment of changes while considering numerous constraints related to forest ecosystem dynamics, is the *Multiple Path* theory of forest design. The general concept was first published by Ware and Clutter (1971) for industrial forests in the South-Eastern USA. The basic idea was implemented in a variety of applications in North America (Forplan, 1986), South America (Schwichtenberg and Sánchez, 2005), New Zealand (Garcia, 1991), South Africa (Gadow and Bredenkamp, 1992), Northern Europe (Kilkki, 1987; Lappi, 1992; Pukkala et al., 1995; Hoen, 1996) and Central Europe (Chen and Gadow, 2002). The early applications were based on forest strata because spatially explicit stand-level planning was limited by ineffective algorithms and long processing times.

The *Multiple Path* theory assumes that a forested landscape is an aggregation of spatially defined land parcels of varying size and shape. These geographical units are known as forest stands or compartments. Each unit is characterized by a specific tree population with a given set of current attributes. There is a basic understanding that not only one, but a variety of treatment schedules or "management paths" may be potentially suitable for each stand. Each stand-path is characterized by a specific succession of management activities, unexpected hazards and growth, and has a value in terms of the services that it provides. Thus, designing a forested landscape involves the search for a combination of management paths which provides a desirable mix of services to the landowner. Following a proposal made by Pukkala and

Kangas (1993), the utility of a specific path combination may be derived as follows:

$$Maximize\ \mathbf{U} = \sum_{i=1}^{n} w_i \cdot u_i(q_i)$$

where U = the utility for a given path combination; n = number of services; w_i = relative weight of service i $\left(0 \leq w_i \leq 1; \sum_{j=1}^{m} w_i = 1\right)$; q_i = realized amount of service i for a given path combination; and $u_i\ (q_i)$ = partial utility function for service i $(0 \leq u_i\ (q_i) \leq 1)$.

This concept uses practical re-orientation time steps and has already proved its worth in large-scale applications (Nuutinen et al., 2000; Kangas et al., 2001; Pykäläinen et al., 2007). Numerous tools have been developed during the past decades to facilitate the realistic implementation of the theory, including improved methods to assess and describe forest spatial structure (Pommerening, 2002; Aguirre et al., 2003), better models of tree growth (Hynynen et al., 2002; Nagel, 2001; Pretzsch, 2001) and more effective algorithms that translate silvicultural prescriptions into tree selection algorithms (Gadow, 1988) and complex forest structural modifications (Albert, 1998; Hessenmöller, 2002). More effective methods to identify the preference structure of forest owners and other decision makers with regard to certain forest services (Leskinen, 2001) and quantitative models to relate particular forest attributes to specific services were developed (Pukkala, 2002; Bosch and Gadow, 1990). Until now, the use of process-based models in the forestry decision-making has been limited. This is, because applications of process-based models may need, for example, data not provided by conventional forest inventories. However, process-based models can provide the same prediction capacity under practical management situations as empirical models (Matala et al., 2003). Moreover, process-based models may help to understand, how forests grow and develop under different climate change scenarios (Mäkelä et al., 2000; Sands et al., 2000; Lindner, 2000; Gracia et al., 2001; Fontes et al., 2004; Hyytiäinen et al., 2004) and how management should be modified in order to avoid detrimental impacts and utilise potential opportunities provided by a possible climate change.

To be effective, adaptive design needs to consider the specific current reality of each individual stand in the landscape. The current mix of desirable services and their relative importance need to be assessed. These requirements may not remain constant over time. Furthermore, landscape level constraints cannot be ignored when taking decisions on stand level. It is also essential to integrate into the design as much expertise as possible from the relevant scientific disciplines. Thus, linking the spatial hierarchies, balancing the mix of services and designing their spatial arrangement, integrating varied forms of experience, limiting the time horizon to allow reliable forcasts, and initializing the attributes of all the stands in the landscape are important prerequisites of forest design. These will be discussed in the following sections.

4.1 Linking the Levels of the Spatial Hierarchy

Many landscapes, especially those where forestry has been practiced for long periods of time, are subdivided into well-defined geographical units, which are known as stands. Each management activity has an influence on the mix of services, not only within the stand, but also in the landscape as a whole. A local harvesting operation contributes to the global reduction of carbon stock, and affects the global mix of services. The different design levels are inseparable, despite the spatial fragmentation that is often encountered. However, the provision of services on the landscape level is often complicated by the relatively small size and large quantity of forest stands and by different objectives of the landowners. For one individual owner or a group of owners sharing the same objectives with regard to the services that the forest should deliver, an effective model which links the different geographical levels in the landscape was first proposed by Ware and Clutter (1971). More recently, the following additive utility model was proposed by Kurttila and Pukkala (2003):

$$\text{Maximize} \quad U = w_l \sum_{j=1}^{J} a_j u_j(q_j) + \sum_{k=1}^{K} w_k \sum_{i=1}^{I} a_{ik} u_{ik}(q_{ik}) \tag{1}$$

$$\text{subject to} \quad q_j = Q_j(\mathbf{x}) \qquad j = 1, \ldots, J \tag{2}$$

$$q_{ik} = Q_{ik}(\mathbf{x}_k) \qquad i = 1, \ldots, I, k = 1, \ldots, K \tag{3}$$

$$\sum_{m=1}^{M_{nk}} x_{mnk} = 1 \qquad n = 1, \ldots, N_k, k = 1, \ldots, K \tag{4}$$

$$x_{mnk} = \{0, 1\} \tag{5}$$

where U is the total utility; w_l is the weight of the landscape level; J is the number of goals at the landscape level; a_j is the relative local importance of landscape level management objective j; u_j is a scaled sub-utility function for management objective j; q_j is the value of objective j; K is the number of forest holdings; w_k is the weight of holding k; I is the number of holding-level management objectives (the same in all holdings); u_{ik} is a scaled sub-utility function in holding k for management objective I; q_{ik} is the value of objective i in holding k and a_{ik} is the relative importance of management objective i in holding k (the global importance of an objective can be computed as $w_l a_j$ or $w_k a_{ik}$). Q_j is an operator that calculates the value of objective $j(q_j)$, Q_{ik} is an operator that calculates the value of objective i of holding k, $\mathbf{x}$ is a vector of binary decision variables (x_{mnk}) that indicate whether stand n in holding k is treated according to management path m, $\mathbf{x}_k$ is a sub-set of $\mathbf{x}$ containing a discrete number of management paths of holding k, N_k is the number of stands in holding k, and M_{nk} is the number of alternative management paths in stand n of holding k. The sub-utility functions transform the absolute value of a particular variable measured in its own specific units to a relative sub-utility value. These functions were determined using the smallest possible target level and the largest possible value of the objective variable, and the respective priorities. The relative sub-utility values were weighted by the relative importance of the objective variable (for details see Kangas, 1992; Pukkala and Kangas, 1993).

Many forest maps and aerial photographs of forested landscapes show a rather fragmented pattern, which is the result of property boundaries on the one hand and spatially organized human activity on the other. These activities began decades and sometimes centuries ago. As a result, forested landscapes are typically fragmented. Local stand level design attempts to define an optimum path in each individual stand, in terms of the desired mix of services. Landscape level planning defines the services on the landscape level as periodic constraints. Thus the above model is effectively linking the stand and landscape levels of the spatial hierarchy.

4.2 Balancing the Mix of Services

The design of a forested landscape is determined by the current demands of society for a particular mix of forest services, and the potential to provide them. The demands may be diverse and wide-ranging, and they usually have to be satisfied simultaneously. The overall preference structure depends on the mutual importance of the services provided by a particular management path, and the behaviour of service-specific sub-utilities. The simultaneous achievement of a balanced set of services, the "*balancing*" problem of forest design, has been addressed in numerous studies dealing with multi-criteria analysis.[3] To be able to balance a set of services requires some kind of preference rating. For example, the overall utility of a particular path combination can be estimated using an additive utility function (cf Keeney and Raiffa, 1993).

There are two fundamentally different approaches to solving this decision problem. One is to evaluate the values of the different path combinations directly with respect to different services. This is known as a discrete approach (Alho et al., 2002) and has been demonstrated in several studies (Leskinen and Kangas, 1998). The discrete approach is practical when the number of stands and stand-wise path combinations is small or when the problem is limited to a few strategic level decision alternatives.

Another possibility is to estimate continuous functions that measure the utilities of the services. In reality the total number of standwise path combinations may be great. Therefore, the continuous approach may be more practical. Another advantage of the continuous approach compared to discrete techniques is that the preferential uncertainties can be measured and incorporated into the decision analysis process (for details refer to Alho et al., 2001; Leskinen et al., 2003). The uncertainties may be seen as differences between the observed preference ratings and a theoretical fitted model. They can be analysed by standard statistical inference

[3] see, for example, Henne, 1976; Saaty, 1980; Steinmeyer and Gadow, 1994; Poschmann et al., 1998; Schmoldt et al., 2001; Kangas et al., 2001; Leskinen, 2001; Pukkala, 2002; Meixner and Haas, 2002; Albert, 2003.

including hypothesis testing and confidence interval estimation. This is interesting because it enables the analyst to evaluate differences more objectively. Some of the differences between certain path combinations may not be significant.

4.3 Designing the Spatial Arrangement of Services

Another important challenge of landscape design involves the spatio-temporal organisation of forest stands. Each stand has unique attributes and the characteristics of neighboring stands may differ substantially, causing abrupt "spatial breaks" within the landscape. Management activities in certain stands may influence adjacent stands, for example by increasing the risk of wind damage after felling operations. The selection of the most suitable treatment path in a particular stand may depend on the characteristics of neighboring stands. Consequently, it is often necessary to coordinate silvicultural activities, not only in a temporal, but also in a spatial context. In addition there may be ecological and/or economic reasons for considering the spatial design of a forest. For example, clustering of stands to increase undisturbed core areas was found necessary to improve habitat quality (e.g. Harrison and Fahrig, 1995; Kurttila, 2001, Öhman and Eriksson, 1999; Nalle et al., 2004). In addition, it is sometimes useful to cluster stands for harvesting, especially when the areas are small (Lu and Eriksson, 2000; Heinonen et al., 2007). The use of spatial objectives implies that the sizes, shapes and juxtapositions of different forest stands and planned management operations are taken into account when designing a particular forested landscape (Kurttila, 2001; Baskent and Keles, 2005).

Adjacency goals and constraints can be used to disperse or aggregate certain features. The constraints may be formulated as a "unit restriction model" (URM) or an "area restriction model" (ARM; Murray, 1999). URM refers to operational restrictions of any two adjacent stands while ARM specifies constraints that are valid for certain groups of spatial units so that the specified open area limit will not be exceeded. In addition to dispersing the cuttings, it is sometimes desirable to spread the characteristics of stands at different points in time. Spreading the fire risk or the habitat suitability are two examples.

There are several ways to define the spatial closeness of different forest stands (Bailey and Gatrell, 1995; Chen and Gadow, 2002; Kurttila et al., 2002). For example, the length of the common boundary between stands may be useful, or the distances between stands, or both. The number of neighborhood relations between stands may increase considerably if all stands within a certain radius from the stand border are considered as neighbors. Hurme et al. (2007), for example, used a neighbourhood distance of 500 m.

Neighbourhood information of spatial units within the landscape, combined with the knowledge of their properties at a given point in time, allows the calculation of different landscape metrics. This in turn makes it possible to analyse spatially explicit patterns of biophysical features, including soils, topography, climate, vegetation, land use, and drainage systems. Associations between those patterns may be quantified and used to measure habitat conditions in areas of varying size. As a result, such metrics can also be used as objective variables in an optimization process.

The use of finely-grained spatial data involving raster cells or other small calculation units is facilitated by improved remote sensing technology (Næsset et al., 2004; Suvanto et al., 2005), but it also increases the decision space (Holmgren and Thuresson, 1997; Lu and Eriksson, 2000; Hyvönen et al., 2005). Finely-grained data may have to be aggregated into feasible treatment units (Heinonen et al., 2007).

Traditionally, forest subdivision and stand delineation was mainly based on timber production characteristics and subjective design by planning experts. More efficient use of scarce resources may require finely-grained spatial entities. The advantages include improved flexibility and increased efficiency, particularly when multiple forest products are considered. However, the use of a large number of small treatment units, which are the actual decision variables, can slow down some standard heuristic optimization methods to impractical levels. As a result, techniques that decompose the problem into stand-level sub-problems have been developed and tested (Pukkala et al., 2007). Practical management may however require predefined and fixed stand boundaries for orientation, work scheduling and bookkeeping.

4.4 Assessing Different Services

Because of the long-term environmental and social implications of forest resource management, forest research has always had to transcend boundaries. Forest scientists had to join up with other disciplines, typically the biological, mathematical and social sciences, to ensure that new and specialized research results are applied to forest management problems. Research about carbon balances is a good example. Carbon sequestration by forests is one way to slow-down climate change and is a prominent example of the environmental services of managed forest ecosystems. The "carbon balance" measures how much carbon is sequestrated or emitted as a consequence of natural stand dynamics and management activity.

According to Pukkala (2007), the carbon balance of a stand during a certain time period can be calculated from the change of biomass resulting from tree growth, the biomass of trees harvested during the period, the biomass of trees which died during the period, the decomposition of dead trees which are present in the stand, and the decomposition of trees harvested from the stand (including roots, cutting residues and removed timber assortments). The calculation of the carbon balance requires models for all the above processes rather than a single model or index. The "decomposition" of the harvested timber also needs to be simulated. This can be done if the average lifespan of products (saw logs, pulpwood, firewood) are known. The carbon balance of a forest is equal to the sum of the carbon balances of the individual stands.

The service production functions for the stands can be used in landscape-level design. The simplest way would be to use the sum (e.g. with carbon balance) or mean (e.g. scenic beauty) of the stand-level indices as a landscape-level objective or constraint. If these are insufficient, various landscape metrics may be computed from the stand-level indices. Landscape metrics are variables that measure the sizes, shapes, relative arrangement and connectivity of habitat patches (or just stands with certain features) as well as their total area (McGarigal and Marks, 1995). These metrics can be used as objective variables in forest landscape design.

4.5 Management Paths and the Design Window

Most foresters do not manage landscapes in their everyday work, their primary unit of concern is the stand, its location, size and particular attributes. Stands have unique identifiers and they are the primary units of spatial organisation, resource assessment and bookkeeping. Each stand development follows a succession of management events and each event has an effect on the services that can be provided. We define the sequence of events as a *path* and the time period during which these events take place as the *design window*. Forest stands have a history of past management, which cannot be changed, and their future development cannot be predicted indefinitely. The *design window* is a limited period of time, starting at the current point in time t_0 and ending at t_1. It is assumed that future management paths and their contribution to a particular mix of environmental services can be predicted during the period $t_0 \ldots t_1$. To be able to describe a path for this period, three important aspects need to be considered: (a) the management events $(\mathrm{E_i})$ at time i; (b) natural growth $(\Delta \mathrm{W}_{i..i+1})$ in response to the i'th event and (c) unexpected hazards $(\mathrm{r}_{i..i+1})$ during the period between events i and $i + 1$ (cf. Yoshimoto, 2001).

Simulations in pure spruce stands (Einsiedel, 2004) and multi-species beech stands (Wagner, 2004) have shown that different paths may have virtually the same value if several criteria are considered simultaneously. This finding is interesting because it shows that in certain stands there may be several equally useful management paths. Generating multiple options, and evaluating them with regard to the services that they produce, is the basic paradigm of the multiple path concept. Besides experience-based path generation, where a qualified expert defines several (usually between 1 and 5) sequences of activities for each stand, there are more sophisticated techniques, including rule-based methods, all-possible-paths methods and single tree methods.

Rule-based methods of path generation may use decision trees which are derived from silvicultural prescriptions. An early example was presented by Gadow (1988) for South African pine plantations. More recent applications of this approach include the rules for Spruce forests in Germany, developed by Sánchez-Orois (2003) and Vilčko (2002). The *all-possible-paths* method was used by Seo et al. (2005) and Hinrichs (2006) who defined a maximum number of harvest events, and several possible stand density categories and thinning weights at each harvest event. A plausibility test is applied to ensure that only those paths are accepted which meet certain restrictions of practicality. A *single tree method* of path generation was presented by Ziegeler and Vilčko (2005) for even-aged Scotch Pine stands. They proposed that the entire stand may be characterized by a representative neighbourhood group of 6 trees. There are two possible destinies for each tree: (a) the tree is harvested at the possible thinning event 1, 2, 3,. . ., or N, and (b) the tree is not harvested and remains in the stand. At the start there are n trees. Each tree may be harvested at any thinning event, but only once. Thus, the sum of the destinies for the entire group in the sample is equal to the product $N * N * \ldots * N$ (n times), producing N^n possible paths.

4.6 Initializing the Landscape

Traditionally, region-oriented forest assessment was carried out periodically at regular time intervals, and the assessment was usually based on the measurement of field sample plots which were systematically distributed in the landscape. The sample plot results were generalised to produce statistical summaries and mean values about the regional forest resources at landscape or property level. The data were then used in periodic forecasting and planning.

A stand is the reference unit for forest resource assessment and design, and the stand-oriented assessment provides spatially explicit information which is essential for timing forest operations. Consequently, up-to-date forest resource data from all the forest stands within the landscape are needed to make predictions and to evaluate decisions concerning future activities. Remote sensing material is used widely in practical applications, sometimes in combination with nonparametric regression methods, to estimate forest attributes (e.g. Tomppo, 1992; Tokola et al., 1996) or to detect changes (Varjo, 1997; Tuominen et al., 2003) following a previous field assessment. More recently, airborne laser scanning (ALS) has provided promising results. When applied simultaneously with aerial photographs it is possible to derive good estimates even for individual tree species (e.g. Packalén and Maltamo, 2006).

The "landscape" may cover a single ownership or multiple forest properties owned by individual owners following their own objectives when managing their forests. Obviously, the ownership composition plays an important role in the initialisation of landscape parameters. There are different approaches to update stand attributes which have been modified by management activities. Stand attributes may be re-measured immediately following a harvest event in the field. The assessment may be integrated into the actual cutting operation, e.g. by storing the data of removals in the board computer of the harvesting machines, or when silvicultural operations are carried out in the forest (Rasinmäki and Melkas, 2005). The assessment may also be organised as a separate field checking (Koivuniemi and Korhonen, 2006). Managers sometimes keep track of stand modifications caused by natural or human disturbances, and use computational methods to update changes in stand attributes (Anttila, 2005).

A similar approach involves an assessment after the trees have been marked and before they are cut. This is known as a *harvest event assessment* or *thinning control* and has been practiced in Australian and South African commercial plantations, and in uneven-aged, multi-species stands in France and Germany. The idea is to provide data for timber buyers about forthcoming yields and to forestall the harvest until the operation is approved. An important objective is to obtain data for preventive analysis of the impact of the structural and ecological changes that will occur if the harvest event does take place. Considerable savings of time and effort may be achieved if the time intervals between successive inventories and the intensity of the assessments are adapted to the information needs.

Irrespective of the particular method, keeping track of operational events can greatly improve the information flow from the forest to management. Post- and pre-harvest event assessments are effective mechanisms for ecological monitoring and operational control. But above all, they provide necessary data for initializing the landscape.

5 Application Examples

When considering the practical application of adaptive landscape design, as outlined in the previous sections, a distinction should be made between regional harvest scenarios and multiobjective ownership planning. The former has strategic relevance on a national or regional scale. The latter refers to a particular forest property with a specific set of objectives. In the following, examples from the practical forestry in Finland are used to illustrate both application types.

5.1 Regional Harvest Scenarios

Since World War II, forestry and the forest industry have been playing an important role in the national economies of some countries, especially in North America and Northern Europe. Therefore, different forest policy measures were defined to encourage forest utilisation and management to provide the raw material for forest industry, both in the short and long term. In Finland, for example, policy measures include strategic forest planning at the national level, co-ordinated by the Ministry of Agriculture and Forestry (MAF) as well as regional and property level forest planning for non-industrial private forest owners (NIPF) carried out by regional forestry centres (RFC).

In Finland, approximately 52% of the forest land is owned by NIPF, 8% by companies, 35% by the State and 5% by others (Metsätilastollinen, 2006). Strategic forest planning at the national level includes forests in all ownership categories and has been supported by calculations based on sample plot and tree data from the national forest inventory (NFI) which was started in Finland as early as 1920s (Ilvessalo, 1927). After World War II, forest financing programmes such as HKLN (Heikurainen et al., 1960) and MERA I–III (MERA, 1968; Ervasti et al., 1965, 1970) were designed to support intensive forest management and forest improvement. The timescale of these programmes was several decades. Since the 1980s the Forest 2000 Programme (1986), its revision (Komiteamietintö, 1992) and their successors such as national forest programmes (NFP) supported by regional forest programmes (RFP) have broadened the interests in forests and forestry beyond timber production. The first NFP for the period 2000–2010 was published in 1999 (Ministry, 1999) and the second will be finished in 2007. The third round of RFPs was finished in 2005.

Since the 1980s, the Finnish MELA system has been used in the analyses of wood production possibilities and the impacts of different harvest levels at the national and regional scale.

Besides the regular assessment of the regional harvest potentials based on the rounds of 8th (e.g. Salminen and Salminen,1998), 9th (e.g. Nuutinen et al., 2005a) and 10th (Nuutinen and Hirvelä, 2006) NFI, the large-scale applications of MELA include several rounds of national timber production analysis since the middle of the 1980s for the Forest 2000 Programme and its revision (Siitonen, 1990), the National Forest Programme 2010 (Nuutinen and Salminen, 1999) as well as three rounds of regional forest programmes in 1998, 2000 (Nuutinen et al., 2005b) and 2005 (Nuutinen et al., 2005c), analysis on multiple-use potential for the preparation of the *Lapland Forest Strategy* (Kajala, 1996), an analysis on the effects of nature conservation strategies for the *Committee for Financing Forest Protection and Employment* (Nuutinen et al., 1996) and reports from research projects (e.g. Siitonen and Nuutinen, 1995 – see Fig. 1; Nuutinen et al., 2006).

The practical implementation of the MELA software for more than 30 years – the years of intensive changes in IT technology – is even today unique in the world. MELA can be compared with some other systems such as FORPLAN and its successor SPECTRUM in the United States (Johnson and Rose, 1986); IFS/FOLPI in New Zealand (Manley, 1996); Hugin (Lundström and Söderberg, 1996) and Indelningspaket (Jonsson et al., 1993), and their successor Heureka (Lämås and Eriksson, 2003) in Sweden; Avvirk (Eid and Hobbelstad, 2000) and its successor GAYA-JLP (Eid, 2000) in Norway.

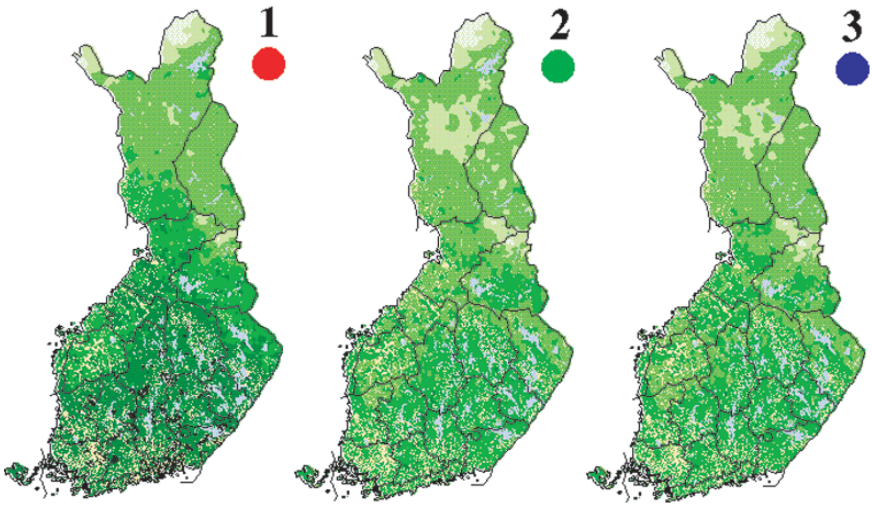

Fig. 1 Three simulations with the MELA system to evaluate the effects of different harvest levels on the per ha growing stock in 2040. Legend and data sources: darker shading of green indicates higher growing stock volumes (Siitonen and Nuutinen 1995, NF1); water bodies in blue and agricultural land in yellow (National Land Survey of Finland). Processing of maps by Harri Kilpeläinen

5.2 Ownership Planning

Regional harvest scenarios provide strategic information for a forest region as a whole, across all forest ownership categories. This information is not only used in national and regional policy formation, but also by timber processing companies when considering new investments and capacity expansions. The purpose of ownership planning is very different. Each individual forest owner or management unit of a timber company has specific growing conditions, tree species and age distributions. The management objectives and constraints are very particular. Detailed, spatially explicit information is usually available and the owner requires a design which is tailormade to the local conditions.

MELA is not only used in regional scenarios, but also in ownership planning. The system has been installed by a number of clients in state, company and private forestry since the 1980s. The approach is based on simulation of management schedules at stand level and optimisation at forestry unit level and was originally presented by Kilkki (1968) and later implemented as a computerised system (Kilkki and Siitonen, 1976; Siitonen et al., 1996). In company forests stand data are used in the company-level calculations, and specific management proposals for individual stands are derived to support operational planning. In state forests, MELA has been used to calculate the impacts of different strategic scenarios in regional land-use planning (Asunta et al., 2004) and in landscape ecological planning (Karvonen, 2000). In private forests the annual coverage of new forest management plans prepared by 13 RFCs is between 900 000 and 1000 000 hectares. MELA is an important component of the SOLMU/LUOTSI information system used in their planning process. A major advance in the MELA system development was the introduction of the JLP software (Lappi, 1992).

Another system which has been designed for ownership planning in Finland is MONSU (Pukkala et al., 1995; Pukkala, 2004). The Monsu software has essential components for simulating the decision space (e.g. a model of biological processes), for preference rating (ranking objectives and constraints), for writing a planning model based information from simulations and preference ratings, for solving the model using either mathematical programming or specific heuristics, and for visualising and analysing the results, using sensitivity analysis tools. One of the interesting features of MONSU is the fact that it integrates many non-timber benefits into numerical forest design. MONSU is considered more applicable for multiple-use forestry than MELA and is therefore used to idely outside Finland.

6 Conclusions

Virtually all forests are utilized by humans and possibly the majority are subject to some kind of planned management. Foresters harvest trees to utilize the timber and to improve the growing conditions of the remaining ones; they change the species composition and tree size distribution to attain some desirable structure; they kill or mutilate trees to increase the amount of habitat in deadwood and adapt rotation ages to increase the runoff from water catchments. Thus, the dynamics of a forest ecosystem is not only determined by natural processes, but to a considerable extent by human interference. Social and ecological processes are intertwined and socio-ecological systems are characterized by constant change and uncertainty. Sustainable use requires that such systems are resilient, which means that societies must have the capacity to anticipate undesirable impacts and to develop adequate mechanisms of response.

We have shown that the traditional methods of standardising silviculture are not suitable in meeting the varied and changing demands of society. Standardized silviculture creates uniform habitats and this may reduce spatial diversity with undesirable effects for species which require specific habitat conditions. Furthermore, tree species choice and silviculture are influenced by continually changing policies, and changing economic and environmental conditions. Therefore, silvicultural standards which are designed for conditions which are assumed to remain constant from the time of regeneration may be outdated before the trees reach maturity.

Thus, the basic challenge of forest design is to continuously adapt to changing environmental and social circumstances. This is not trivial, because forest development is slow and rapid changes of the species composition, size distribution and many other features influencing the quantity and quality of services, are usually not possible. One approach which involves continuous assessment of changes while considering numerous constraints related to forest ecosystem dynamics, is the *Multiple Path* theory of forest design which assumes that a forested landscape is an aggregation of spatially defined land parcels, or “stands” of varying size and shape. Each of these geographical entities is characterized by a specific tree population with a given set of current attributes. There is a basic understanding that not only one, but a variety of treatment schedules or “management paths” may be potentially suitable for each stand. Each stand-path is characterized by a specific succession of management activities, unexpected hazards and growth, and has a value in terms of the services that it provides. Thus, designing a forested landscape involves the search for a combination of management paths with the aim of providing a desirable mix of services to the landowner.

Several systems have been developed for the practical application of the multiple path principle. We have briefly introduced two examples, MELA and MONSU. Both are prominent examples of the practical application of the theory.

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Part II
Assessing the Landscape

Landscape Metrics, Scales of Resolution

Samuel A Cushman and Kevin McGarigal

1 Introduction

Effective implementation of the "multiple path" approach to managing green landscapes depends fundamentally on rigorous quantification of the composition and structure of the landscapes of concern at present, modelling landscape structure trajectories under alternative management paths, and monitoring landscape structure into the future to confirm whether management is having the expected effects on landscape structure. Indeed, quantification of current conditions, anticipation of future changes and monitoring these changes as they occur are the three foundational elements of adaptive management and the key foundation for the multiple path approach to landscape design.

In this chapter we review five key components of landscape analysis. First, we discuss the importance of appropriate models of landscape structure. Second, we review the critical issue of scale and its enormous influences on landscape analysis. Third, we review the importance of evaluating the spatio-temporal context of the present landscape, and discuss several approaches to establishing it. Fourth, we define the meaning landscape metrics, review the major components of landscape pattern, and describe a number of the most useful landscape metrics for characterizing each component of landscape structure. Fifth, we discuss linking landscape analysis to simulation modelling to infer the expected effects of alternative management scenarios using landscape trajectory analysis. The topics in this chapter are adapted from much more complete treatment in McGarigal et al. (in press) and Cushman and McGarigal (2006).

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2 Defining the Model of Landscape Structure

2.1 Classification Paradigms

There are many ways to model or represent landscape structure. For example, in the *island biogeographic* model the landscape is represented as a simple binary landscape consisting of habitat patches and a background matrix. In this case, habitat is represented as occurring in discrete patches distributed in a homogeneous matrix. In contrast, the *landscape mosaic* model represents the landscape as a spatially heterogenous mosaic of several patch types. The appropriate model of landscape structure depends on the issues of importance to managers or researchers. The same landscape can be represented in many different ways, and the investigator or manager must select a model that adequately represents the landscape in a manner and scale relevant to the issues or organisms of importance.

2.2 Choice of Digital Data Format

There are different data formats for digitally representing a model of the landscape, and there are practical implications to the choice of formats. Vector and raster data formats are both commonly used in landscape analysis. Vector format represents each patch as a polygon, where the boundaries of the polygon conform to the actual boundaries of the patch in the real world. In contrast, raster format represents the landscape as a lattice of grid cells, in which each cell is assigned a value corresponding to the patch type or class comprising the majority of the cell. Patches are defined by contiguous cells of the same cover type. Because vector and raster formats represent lines differently, metrics involving edge or perimeter will be affected by the choice of formats (McGarigal et al. in press). Edge lengths will be biased upward in raster data because of the stair-step outline, and the magnitude of this bias will vary in relation to the grain or resolution of the image. The key point here is that many landscape metrics are not invariant to the data format, and some metrics are defined only for vector or raster formats. In general, the raster format has gained popularity over the vector format because of the ease of conducting complex spatial computations on grids.

3 Determining the Appropriate Scale of Analysis

Measured landscape pattern is a function of scale. To summarize McGarigal et al. (in press), the spatial scale of ecological data is defined by extent and grain (Forman and Godron 1986; Turner et al. 1989; Wiens 1989a). *Extent* is the area within the landscape boundary. The spatial extent of an investigation, from a statistical perspective, is the area defining the population we wish to sample. *Grain* is the size of the individual units of observation. A fine-grained map might represent information as 0.1-ha units, whereas a map with an order of magnitude coarser resolution would have information structured into 1-ha units (Turner et al. 1989). Extent and grain define the upper and lower limits of resolution of a study, and inferences about scale-dependency are constrained by the extent and grain of the data (Wiens 1989a). One cannot extrapolate beyond the extent of the sampled population, nor can one infer pattern of objects smaller than the grain of the data.

It is critical that extent and grain be defined appropriately to represent the ecological phenomenon or organism under study, or there will be a good chance of reaching erroneous conclusions (McGarigal et al. in press). In practice, extent and grain are often dictated by the scale of the imagery (e.g., aerial photos, Landsat images) used or the technical capabilities of the computing environment. However, it is more ecologically meaningful to define scale from the perspective of the organism or ecological phenomenon under consideration (McGarigal et al. in press). Typically, however, one does not know the appropriate scale parameters. In such cases, it is advisable to choose a finer grain than is believed to be important because the grain sets the minimum resolution of investigation (McGarigal et al. in press). Once set, we can always resample to a coarser grain. In addition, one can specify the minimum patch size to be represented in a landscape, and this can easily be manipulated above the grain size. Indeed, it is often useful to reanalyze the same landscape using progressively coarser minimum patch sizes to better assess landscape heterogeneity across a range of scales (Thompson and McGarigal 2002).

There are often substantial practical implications of the choice of grain and extent. Many of the landscape metrics used to quantify landscape structure are highly sensitive to grain (McGarigal et al. in press). Metrics involving edge or perimeter will be affected, with edge lengths biased upwards in proportion to the grain size. Metrics based on cell adjacency information such as the contagion index of Li and Reynolds (1993) will be affected as well, because grain size effects the proportional distribution of adjacencies. In this case, as resolution is increased (grain size reduced), the proportional abundance of like adjacencies (cells of the same class) increases, and the measured contagion increases. Similarly, the measured degree of habitat fragmentation may vary with extent, especially if the habitat is not uniformly distributed throughout the entire study area (McGarigal et al. in press).

The ratio of grain to extent for a given landscape also warrants consideration. If the ratio is small (i.e., a coarse-grained map), then the landscape dynamics are likely to be dominated by boundary effects, analogous to the bias associated with small sample size in statistics (McGarigal et al. in press). Moreover truncation of patches artificially by the landscape boundary can have a dramatic influence on certain metrics. Also, if the landscape extent is small compared to the scale of the organism or ecological process under consideration then any metric will have questionable meaning. Metrics based on nearest neighbor distance or employing a search radius can be particularly misleading in such situations (McGarigal et al. in press).

Any model of landscape structure requires an explicit identification of scale. In any landscape structural analysis, it is incumbent upon the investigator or manager to select a scale (i.e., extent, grain, minimum mapping unit) that is appropriate to the phenomenon under consideration, as interpretation of landscape structure is constrained scale. In addition, observed patterns or relationships should be evaluated relative to the limitations imposed by the scale of the data. In most applications a multi-scale assessment of habitat loss and fragmentation is recommended (McGarigal et al. in press), in which a relevant landscape model is established across a broad range of scales (i.e., grain and extent).

4 Defining the Relevant Spatiotemporal Context for Assessing Landscape Structure

4.1 Spatial Context

Each landscape has a context or regional setting, regardless of scale and how the landscape is defined. The importance of the landscape context is dependent on the phenomenon of interest, but typically varies as a function of the "openness" of the landscape (McGarigal et al. in press). The "openness" of the landscape depends not only on the phenomenon under consideration, but on the basis used for delineating the landscape boundary. For example, from a geomorphological or hydrological perspective, the watershed forms a natural landscape, and a landscape defined in this manner might be considered relatively "closed". Conversely, from the perspective of a bird population, topographic boundaries may have little ecological relevance, and the landscape defined on the basis of watershed boundaries might be considered a relatively "open" system. Local bird abundance patterns may be produced not only by local processes or events operating within the designated landscape, but also by the dynamics of regional populations or events elsewhere in the species' range (Wiens 1981, 1989b; Vaisanen et al. 1986; Haila et al. 1987; Ricklefs 1987).

Fragmentation metrics quantify the amount and configuration of habitat within the designated landscape boundary only. Consequently, the interpretation of these metrics and their ecological significance requires an acute awareness of the landscape context and the openness of the landscape relative to the extent of the landscape and the issues under consideration (McGarigal et al. in press). In particular, a focal habitat may be quite rare and fragmented in distribution at one landscape extent, but form a highly connected matrix at a broader or finer extent. Moreover, the ecological consequences of the measured habitat fragmentation may vary depending on the landscape extent in relation to the character of the landscape context. For example, a fragmented habitat distribution at one scale nested within a regional landscape context in which that habitat is abundant and highly connected may function differently than if that habitat is similarly rare and fragmented within the regional context as well (e.g., McGarigal and McComb 1995). Likewise, an abundant and unfragmented habitat distribution at the focal scale may not function as expected if the habitat is rare and highly fragmented within the broader context. Hence, any assessment of habitat loss and fragmentation should include at least a qualitative assessment of habitat conditions within the broader regional context.

4.2 Temporal Context

Fragmentation metrics quantify the pattern of habitat at a snapshot in time. Yet it is often difficult, if not impossible, to determine the ecological significance of the computed value without understanding the range of natural variation in habitat pattern. For example, in disturbance-dominated landscapes, habitat patterns may fluctuate widely over time in response to the interplay between disturbance and succession processes (e.g., Wallin et al. 1996; He and Mladenoff 1999; Haydon et al. 2000; Wimberly et al. 2000; McGarigal et al. 2001). It is logical, therefore, that landscape metrics should exhibit statistical distributions that reflect the natural spatial and temporal dynamics of the landscape. By comparison to this distribution, a more meaningful interpretation can be assigned to any computed value. Despite widespread recognition that landscapes are dynamic, there are few studies quantifying the range of natural variation in landscape pattern metrics (Neel et al. 2004). This remains one of the greatest challenges confronting landscape pattern analysis.

5 Choosing Metrics of Landscape Structure

Clearly, given the number and variety of components of landscape structure affected by habitat loss and fragmentation, it is unreasonable to expect a single metric, or even a few metrics, to be sufficient (Cushman et al. submitted). Therefore, a truly multivariate approach is warranted in most applications. Unfortunately, the selection of a suite of fragmentation metrics is constrained by the lack of a proper theoretical understanding of metric behavior. The proper interpretation of a landscape metric is contingent upon having an adequate understanding of how it responds to variation in landscape patterns (e.g., Gustafson and Parker 1992; Hargis et al. 1998; Jaeger 2000; Neel et al. 2004). Failure to understand the theoretical behavior of the metric can lead to erroneous interpretations (e.g., Jaeger 2000). Neutral models (Gardner et al. 1987; Gardner and O'Neill 1991; With 1997) provide an excellent way to examine metric behavior under controlled conditions because they control the process generating the pattern, allowing unconfounded links between variation in pattern and the behavior of the index (Gustafson 1998). Unfortunately, neutral models of fragmentation under various forms of fragmentation (e.g., random, contagious, disperse, corridor, edge, and nuclear) have not been used in a comprehensive manner to evaluate a broad array of metrics. Therefore, it is not possible to reliably identify the "best" measures of habitat loss and fragmentation. Rather, the measures described below provide a comprehensive, yet parsimonious, suite of metrics for quantifying habitat loss and fragmentation, and are recommended until such time as a more thorough investigation of metric behavior is completed.

There are five major spatial components to habitat loss and fragmentation: (1) habitat extent, (2) habitat subdivision, (3) patch geometry, (4) habitat isolation, and (5) habitat connectedness (McGarigal et al. in press). Numerous landscape metrics have been developed for each of these components (e.g., Baker and Cai 1992; McGarigal and Marks 1995; Jaeger 2000; McGarigal et al. 2002). However, these categories are not discrete and many landscape metrics measure properties that relate to several components. Thus, a simple classification of metrics into these categories is not straightforward. Nevertheless, this classification has practical utility because it ensures that a comprehensive suite of metrics is selected. In the sections below, we identify and describe a parsimonious suite of metrics for measuring habitat loss and fragmentation (Table 1). It is important to note that the specific metrics included here do not represent a comprehensive list of useful fragmentation metrics and do not necessarily include the "best" metrics as considered from any one perspective. Rather, given the overwhelming number and variety of available metrics, it was our intent to suggest a parsimonious suite of metrics that measure different aspects of habitat loss and fragmentation and that when taken together may provide a comprehensive assessment of habitat loss and fragmentation in most applications.

5.1 Habitat Extent

Habitat extent represents the total areal coverage of the target habitat in the landscape and is a simple measure of landscape composition, represented by the following metric:

Percentage of Landscape. A straightforward and intuitive metric that measures habitat extent in relative terms is the *percentage of the landscape* (PLAND) comprised of the target habitat, defined as follows:

$$PLAND = P_i = \frac{\sum_{j=1}^{n} a_{ij}}{A} \cdot 100$$

where P_i equals the proportion of the landscape occupied by the ith patch type (the focal habitat); a_{ij} is the area (m^2) of the jth patch; and A is the total landscape area (m^2). PLAND approaches 0 when the corresponding patch type (class) becomes increasingly rare in the landscape. PLAND equals 100 when the entire landscape is comprised of the focal patch type; that is, when the entire image is comprised of a single patch. PLAND is a relative measure, as opposed to total class area, and

Table 1 Proposed parsimonious suite of metrics for measuring habitat loss and fragmentation

Spatial component	Metric name	Acronym in FRAGSTATS
Habitat Extent	Percentage of Landscape	PLAND
Habitat Subdivision	Patch Density	PD
	Clumpy	CLUMPY
	Landscape Division	DIVISION
Patch Geometry	Total Core Area Index	TCAI
Habitat Isolation	Class-level Similarity Index	SIMILAR_AM
Habitat Connectedness	Correlation Length	GYRATE_AM

therefore may be used for comparing among landscapes of varying sizes. PLAND is not affected in any way by the spatial distribution or configuration of habitat patches.

5.2 Habitat Subdivision

Habitat subdivision deals explicitly with the degree to which the habitat has been broken up into separate patches (i.e., fragments), *not* the size, shape, relative location, or spatial arrangement of those patches. Because these latter attributes are usually affected by subdivision, it is difficult to isolate subdivision as an independent component. Subdivision can be measured in a variety of ways, but it is perhaps best defined by the absolute or relative number of patches, the degree of habitat aggregation or clumpiness, and the degree of subdivision derived from a geometric view based on the cumulative distribution of patch sizes, as represented by the following metrics:

Number of Patches or Patch Density. A simple and direct measure of habitat subdivision is given by the *number of patches* (NP) or, alternatively, *patch density* (PD). Unfortunately, both of these measures are difficult to interpret by themselves without also considering habitat area. Nevertheless, regardless of habitat area, as the number of habitat patches increases, technically the habitat becomes more fragmented (subdivided into disjunct patches). For this reason, this metric is often reported as a basic descriptor of habitat subdivision. The choice between NP or PD largely depends on the application. If more than one landscape is involved and they are different sizes, then PD is the more logical formulation because patch number is standardized to a per unit area. However, if a single landscape or multiple landscapes of equal extent are involved, the two formulations are equivalent and the choice becomes one of personal preference.

Clumpiness. A useful measure of habitat subdivision is given by the *clumpiness index* (CLUMPY) which measures the degree to which the focal habitat is aggregated or clumped given its total area. CLUMPY is calculated from the adjacency matrix, which shows the frequency with which different pairs of patch types (including like adjacencies between the same patch type) appear side-by-side on the map. CLUMPY is scaled to account for the fact that the proportion of like adjacencies (G_i) will equal P_i for a completely random distribution (Gardner and O'Neill 1991), and is defined as follows:

$$CLUMPY = \frac{G_i - P_i}{P_i} \quad for \quad G_i < P_i \& P_i < .5, \quad else \quad \frac{G_i - P_i}{1 - P_i}$$

$$\text{with } G_i = \left(\frac{g_{ij}}{\sum_{i=1}^{n} g_{ik} - \min e_i} \right)$$

and where g_{ii} is the number of like adjacencies (joins) between pixels of patch type (class) i, g_{ik} is the number of adjacencies (joins) between pixels of patch types (classes) i and k; min e_i is the minimum perimeter (in number of cell surfaces) of patch type (class) i for a maximally clumped class, and P_i is the proportion of the landscape occupied by patch type (class) i.

The formula is contingent upon G_i and P_i because the minimum value of G_i has two forms which depend on P_i. Specifically, when $P_i \leq 0.5,\ G_i = 0$ when the class is maximally disaggregated (i.e., subdivided into one cell patches) and approaches 1 when the class is maximally clumped. However, when $P_i \geq 0.5,\ G_i = 2P_i - 1$ when the class is maximally disaggregated and approaches 1 when the class is maximally clumped. Given any $\mathrm{P_i}$, CLUMPY equals -1 when the focal patch type is maximally disaggregated; CLUMPY equals 0 when the focal patch type is distributed randomly, and approaches 1 when the patch type is maximally aggregated. Thus, CLUMPY has a straightforward and intuitive interpretation that indicates whether the focal habitat is more or less clumped than expected by chance alone. Note, because CLUMPY is based on "cell" adjacencies, it is sensitive to the grain size or resolution of the landscape. As the grain size is reduced (i.e., the landscape is mapped at a finer resolution), the proportion of like adjacencies will increase and the landscape will appear to be more clumped. Thus, this metric, like most others, must be interpreted in reference to the resolution of the map and is perhaps best used as a comparative index for comparing among different landscapes or the same landscape over time.

Degree of Landscape Division. Jaeger (2000) presented a new suite of indices that measure habitat subdivision from a geometric point of view and are calculated from the cumulative distribution function of the habitat patch sizes. *Degree of landscape division* (DIVISION) equals the probability that two randomly chosen places in the landscape under investigation are not situated in the same contiguous habitat patch. Thus, as the habitat becomes increasingly subdivided into smaller patches, the probability increases that two randomly chosen locations will belong to separate patches. DIVISION is defined as follows:

$$DIVISION = \left(1 - \sum_{j=1}^{n} \left\{\frac{a_{ij}}{A}\right\}^2\right) \cdot 100$$

where a_{ij} is the area (m^2) of patch ij and A is the total landscape area (m^2). DIVISION equals 0 when the landscape consists of a single patch, and approaches 100 as the proportion of the landscape comprised of the focal patch type decreases and as those patches decrease in size. DIVISION is maximum when the focal patch type consists of single one-cell patches. Conceptually, DIVISION and its related metrics (the splitting index and effective mesh size, not described here) are similar to simple measures of number of patches and average patch size, but they posses desirable mathematical properties that make them superior to the simpler measures. In particular, they have low sensitivity to very small patches, behave monotonically to the various fragmentation phases, and are sensitive to structural differences in the landscape mosaic (Jaeger 2000). DIVISION is closely related and effectively redundant with the area-weighted mean size of focal habitat patches, which explains its insensitivity to very small patches (i.e., small patches are given very little weight). Note, unlike CLUMPY, DIVISION (at the class level) is effected by both habitat extent and subdivision, thus confounding these two components. Consequently, this metric must be interpreted in conjunction with PLAND and is perhaps best used as a comparative index for comparing among different landscapes with the same habitat extent.

5.3 Patch Geometry

Patch geometry deals explicitly with the spatial character of habitat patches. Given the myriad aspects of patch geometry, there exists a wide variety of patch geometry metrics. Here we focus exclusively on "core area" because it integrates several aspects of patch geometry relevant to habitat fragmentation. Core area is defined as the area within a patch beyond some specified depth-of-edge effect distance. Holding patch area constant, as a patch becomes more convoluted and complex in shape, core area decreases because less of the patch is greater than the specified depth-of-edge distance from the perimeter. Core area measures are particularly sensitive to habitat subdivision because, as just noted, they are sensitive to basic perimeter-area relationships. For example, holding the total area of habitat constant, the subdivision of habitat automatically decreases core area because of the increase in the ratio of habitat perimeter to area. Concomitant habitat loss and subdivision results in an even greater decrease in core area. Although there are many alternative ways to index core area, they all measure the same basic integrated aspect of patch geometry and are therefore largely redundant. Consequently, a single metric is sufficient, as follows:

Core Area Index. The *core area index* (CAI) is basically an edge-to-interior ratio like many shape indices (McGarigal et al. 2002), the main difference being that the core area index treats edge as an area of varying width and not as a line (perimeter) around each patch. CAI is computed at the patch level, as follows:

$$CAI = \frac{a_{ij}^c}{a_{ij}} \cdot 100$$

where a_{ij}^c is the core area (m^2) of patch *ij* based on specified depth-of-edge distances (m), and a_{ij} is the area (m^2) of patch *ij*. In other words, CAI equals the percentage of a patch that is core area. CAI can be averaged across all patches of the focal habitat (weighted by patch area) to provide a suitable class-level metric, or an equivalent *total core area index* (TCAI) can be calculated directly, as follows:

$$TCAI = \frac{\sum_{j=1}^{n} a_{ij}^c}{\sum_{j=1}^{n} a_{ij}} \cdot 100$$

TCAI is equivalent to the *area-weighted mean core area index* (CAI_AM) reported in FRAGSTATS (McGarigal et al. 2002). TCAI equals 0 when there is no core area (i.e., every location within the focal habitat is within the specified edge influence distance from the habitat edge); that is, when the habitat contains no core area. TCAI approaches 100 when the habitat, because of patch size, shape, and edge width, contains mostly core area. TCAI is a relative index; it does not reflect patch size, class area, or total landscape area; it merely quantifies the percentage of available area, regardless of whether it is 10 ha or 1,000 ha, comprised of core. Consequently, this index does not confound area and configuration; rather, it isolates the configuration effect. For this reason, the core area index is probably best interpreted in conjunction with total habitat area, or its relativized equivalent – PLAND.

5.4 Habitat Isolation

Habitat isolation deals explicitly with the spatial and temporal *context* of habitat patches, rather than the spatial character of the patches themselves. Unfortunately, isolation is a difficult thing to capture in a single measure because there are many ways to quantify context. Isolation can be measured in the spatial dimension in several ways, depending on how one views the concept of isolation. Here, we focus exclusively on a measure of neighborhood similarity, or, conversely, contrast, within a specified ecological neighborhood surrounding each habitat patch. Isolation is measured by the degree of contrast (i.e., the magnitude of differences in one or more attributes between adjacent patch types) between the focal habitat and neighboring patches.

Similarity Index. The *similarity index* (SIMILAR) is a patch-level measure of neighborhood similarity (McGarigal et al. 2002). It considers the size and proximity of all patches, regardless of class, whose edges are within a specified search radius of the focal patch. SIMILAR is computed at the patch level, as follows:

$$SIMILAR = s_{ij} = \sum_{k=1}^{n} \frac{a_{ijs} \cdot d_{ik}}{h_{ijk}^2}$$

where a_{ijs} is the area (m^2) of patch ijs within the specified neighborhood (m) of the focal patch; $\mathrm{d_{ik}}$ is the similarity (coefficient between 0–1) between the focal patch (type i) and the kth patch within the specified neighborhood; and h_{ijs} is the distance (m) between the focal patch and each neighboring patch ijs, based on patch edge-to-edge distance. SIMILAR can be averaged across all patches of the focal class (weighted by patch area) to provide a suitable class-level metric, as follows:

$$SIMILAR\_AM = \sum_{j=1}^{n} \left[s_{ij} \left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}} \right) \right]$$

SIMILAR_AM equals 0 if all the patches surrounding the focal habitat patches have a zero similarity coefficient (i.e., maximum contrast). SIMILAR_AM increases as the habitat patches are increasingly surrounded by patches with greater similarity coefficients and as those similar patches become closer and more contiguous in distribution. The similarity index quantifies the spatial context of a (habitat) patch in relation to its neighbors of the same or similar class; specifically, the index distinguishes sparse distributions of small and insular habitat patches from configurations where the habitat forms a complex cluster of larger, hospitable (i.e., similar) patches. Note, in a binary landscape consisting of the focal habitat class and a non-habitat matrix, as might be represented under an island biogeographic perspective on landscape structure, the similarity index reduces to the proximity index given in FRAGSTATS [see also Gustafson and Parker (1992) and Whitcomb et al. (1981)], which considers only patches of the focal class. Note, the similarity index must be interpreted carefully in relation to habitat extent. Specifically, because it is computed at the patch level and considers only the composition of the landscape surrounding each habitat patch, not the area of the patch itself, it can exhibit erroneous behavior under certain circumstances. For example, when the focal habitat comprises most (or all) of the landscape and is connected into a single matrix-forming patch, SIMILAR_AM equals 0 because there are no other (similar) habitat patches in the neighborhood, implying isolation when in fact the habitat is both abundant and highly connected. Thus, SIMILAR_AM is best interpreted in conjunction with PLAND and its use should probably be limited to landscapes in which the focal habitat is not matrix forming (i.e., typically say PLAND $<$ 30–40%) and consists of multiple patches.

5.5 *Connectedness*

Connectedness integrates all of the above components and involves both a structural component (continuity) and a functional component (connectivity). As noted previously, it is difficult to isolate connectedness in a metric owing to the myriad constituent components. Consequently, the range of available connectedness metrics is somewhat limited; although, this is an area of active development in landscape pattern analysis so new measures will likely become available in the future. Here we focus on a single continuity metric and a single connectivity metric. Note, however, that there is considerable redundancy between these metrics and the suggested isolation and subdivision metrics.

Correlation length. A useful measure of the continuity or structural connectedness of habitat is the *correlation length index* (CLI) (Keitt et al. 1997), which is derived from the patch *radius of gyration* (GYRATE), as follows:

$$GYRATE = g_{ij} = \sum_{r=1}^{z} \frac{h_{ijr}}{z}$$

where h_{ijr} is the distance (m) between cell *ijr* [located within patch *ij*] and the centroid of patch *ij* (the average location), based on cell center-to-cell center distance; and z is the number of cells in patch *ij*. GYRATE equals the mean distance (m) between each cell in the patch and the patch centroid and represents the average distance an organism can move and stay within the patch boundary. The correlation length index (CLI) is computed as the *area-weighted average patch radius of gyration* (GYRATE_AM), as follows:

$$CLI = GYRATE\_AM = \sum_{j=1}^{n}\left[g_{ij}\left(\frac{a_{ij}}{\sum\limits_{j=1}^{n} a_{ij}}\right)\right]$$

where a_{ij} is the area of patch *ij*. CLI equals 0 when the habitat consists of single-cell patches and increases as the patches increase in extensiveness. Correlation length is intuitively appealing, because as patches become smaller and more compact, they extend over less space and therefore provide for less physical continuity of the habitat across the landscape. Large and elongated patches extend over greater space and provide for greater connectedness of the habitat. Thus, holding habitat area constant, as the habitat becomes more subdivided during fragmentation, correlation length decreases. Correlation length can be interpreted as the average distance an organism might traverse the map, on average, from a random starting point and moving in a random direction, i.e., it is the expected traversibility of the map (Keitt et al. 1997).

Traversibility Index. A useful measure of the functional connectivity of habitat is given by the *traversibility index* (TRAVERSE) based on the idea of ecological resistance (McGarigal et al. 2002). The premise is that a hypothetical organism dispersing from a focal habitat cell in a highly traversable neighborhood can reach a large area with minimal crossing of “hostile” cells. This metric uses a resistance surface to determine the area that can be reached from each cell in the focal habitat. The focal cell gets an organism-specific “bank account,” which represents, say, an energy budget available to the organism for dispersal from the focal cell. The size of the account is selected to reflect the organism’s dispersal or movement ability. Each patch type (including the focal patch type) is assigned a cost based on a user-specified resistance matrix. Specifically, relative to a each focal patch type, each patch type is assigned a resistance coefficient in the form of weights ranging from 1 (minimum resistance, usually associated with the focal patch type) to any higher number that reflects the relative increase in resistance associated with each patch type. The metric is computed by simulating movement away from the focal cell in all directions, where there is a cost to move through every cell. Under the best circumstances (i.e., minimum resistance), there will exist a maximum dispersal area based on the specified account or energy budget. Note that assigning a (small) cost for traveling through the focal community (typically a cost of 1) results in a linearly decaying function. Moving through more resistant cells costs more and drains the account faster. Thus, depending on the resistance of the actual landscape in the vicinity of the focal cell, there will be a certain area that a dispersing organism can access. This area represents the least-cost hull around the focal cell, or the maximum distance that can be moved from the cell in all directions until the "bank account" is depleted. This dispersal area, given as a percentage of the maximum dispersal area under conditions of minimum resistance, provides a measure of the relative traversability of the landscape in the vicinity of the focal cell. Averaging this index across cells of the focal habitat provides a class-level index of traversability, computed as follows.

$$TRAVERSE = \frac{\left(\sum_{r=1}^{z} t_{ir} \Big/ z_i \right)}{t_{\max}} \cdot 100$$

where t_{ijr} is the least cost hull area around the rth cell in the focal patch type (i); z_i is the total number of cells in the focal patch type (i); and $t_{\max}$ is the maximum least cost hull area around a cell of the focal patch type (i) given minimum resistance. TRAVERSE equals 0 when the focal habitat consists of one or more single-cell patches surrounded by hostile patch types that prevent any movement (i.e., function as barriers). This is achieved when the resistance coefficient for the neighboring patch type(s) is greater than the user-specified bank account for the focal cell. TRAVERSE equals 100 when the focal habitat is surrounded by minimally resistant patch types. TRAVERSE is computed at the cell level and then averaged across cells in the focal patch. As a result, this metric requires substantial computations and may take considerable time to compute for a large landscape. In addition, this metric requires the user to specify an appropriate resistance matrix containing coefficients for each pairwise combination of patch types, as well as a scaling factor that governs the size of the maximum least cost hull; that is, the size of the area surrounding the focal cell that is accessible given minimum resistance.

6 Landscape Trajectory Analysis

Once the model of landscape structure, the appropriate spatial scales, and spatio-temporal reference framework have been determined it is possible to meaningfully evaluate the current condition of the landscape in terms of the area and configuration of landscape elements. To evaluate the meaning of these conditions it is essential to have a quantitative goal or benchmark with which to compare. Specifically, it is desirable to compare current conditions to a desired condition which expresses the actualization of the management objectives. Importantly, for these desired landscape conditions to be meaningful in evaluating current conditions they must be expressed in terms of quantitative values and ranges for a selection of landscape metrics. Once desired conditions are thus expressed it is possible to evaluate the current condition in terms of its departure from desired conditions. Cushman and McGarigal (2006) present the analytical framework for evaluating landscape structure with respect to past conditions or desired future conditions. The framework is based on several key ideas, including (a) quantifying the multivariate character of landscape structure; (b) evaluating current conditions with respect to historic ranges or desired future conditions; (c) measuring changes in landscape structure over time; and (d) quantitatively comparing the rates and patterns of landscape change among mulitple landscapes over time.

6.1 Landscape Structure Space

Landscape structure space is derived from a p-dimensional space, where each dimension represents a different landscape metric. We refer to this p-dimensional space as *landscape metric space*, denoting that the dimensions are defined by the individual landscape metrics (but in standardized form). Because many landscape metrics are partly redundant (Riitters et al. 1995) it is often preferable to obtain orthogonal combinations from an unconstrained ordination technique (McGarigal et al. 2000), such as principal components analysis (PCA) (e.g., McGarigal and McComb 1995; Cushman and Wallin 2000) or nonmetric multidimensional scaling (NMDS). In this manner, the p-dimensional space is reduced to an m-dimensional space, where hopefully $m << p$. We refer to this reduced m-dimensional space as *landscape structure space*, denoting that the dimensions now represent composite structure gradients whose exact definitions will vary among data sets depending on the suite of landscape metrics measured and idiosyncracies of the specific landscapes. Representing multivariate measurement of landscape structure as an m-dimensional landscape structure space greatly facilitates analysis when multiple metrics are measured simultaneously. The challenge of describing each landscape across all measured metrics is replaced by describing the relative locations and rates and directions of change in a much reduced landscape structure space. This makes for much more concise and meaningful analysis. Note, however, that the trajectory analysis described below can just as easily be conducted on the original p-dimensional landscape metric space, although the solution is not as concise. Ultimately, the choice of approach depends on how successfully the variance structure of the measured landscape metrics can be summarized by ordination. In the description that follows, we will presume the use of an ordination approach.

6.2 Location and Distance

The location of a landscape in structure space is defined by the coordinates of the landscape on each axis of the landscape structure space. These coordinates are simply the values of the orthogonal axis scores (e.g. from PCA or NMDS). Location is defined by the position vector describing the direction and distance of the landscape from the origin of the structure space. In simplest terms, the position vector describes the location of the landscape in the m-dimensional structure space. The position vector is most easily handled in component form, which decomposes its components along each axis of the structure space:

$$r_i = x_{i1} + x_{i2} + \ldots + x_{im}$$

The coefficients x_{im} are the scalar elements of the position vector, defining the magnitude of displacement of the ith landscape (r_i) from the origin along each of the m dimensions of the structure space.

To evaluate the current landscape condition it is essential to quantitatively calculate the distance of the landscape from a reference or desired condition in landscape structure space. This distance is defined as the Euclidean distance between the location of the landscape and the reference or desired condition:

$$\overline{d}_i = \sqrt{\sum_{k=1}^{m} (X_{ijk} - X_0)^2}$$

where x_{ijk} is the score for landscape i $(1 \ldots \mathrm{n})$ at time j $(0,\ t)$ on axis k $(1 \ldots \mathrm{m})$. This is simply the Pythagorean theorem applied to m dimensions. Displacement in m dimensions of the current from the reference or desired condition provides a measurement of how much the overall landscape structure departs from management objectives. Alternatively, displacement can also be expressed in component form:

$$\overline{d}_i = \Delta x_{i1} + \Delta x_{i2} + \ldots + \Delta x_{im}$$

where Δx_{ik} is the difference in scores between time $j = t$ and 0 for landscape i $(1 \ldots \mathrm{n})$ on axis k $(1 \ldots \mathrm{m})$. In this form, displacement is defined as the change in the landscape structure space along each dimension. This form facilitates interpretation because displacement can be described directly in relation to the particular aspects of landscape structure represented by each dimension.

6.3 Landscape Trajectories

Evaluating current conditions with respect to reference and desired conditions provides a measure of departure to guide management. However, it is insufficient to fully inform management decisions as to what management strategy will most effectively lead to the creation of the desired conditions on the landscape of interest. Dynamic landscape simulation models are useful in this task (Cushman and McGarigal 2006). By explicitly simulating future landscape conditions under alternative management regimes, involving different areas, frequencies and patterns of treatment or restoration, it is possible to create plausible futured landscapes. The concepts of location and distance in landscape structure space can then be used to determine which management alternatives most efficiently lead to creation of desired landscape condition (Cushman and McGarigal 2006). Specifically, multiple plausible alternative scenarios should be simulated. As landscape dynamics simulation models are stochastic it is important to produce a sample of replicate runs on each scenario (usually 100–500), and to output results at a temporal time-step sufficiently fine to match the temporal scale of the management and disturbance processes and to be relevant to management decision making (usually 1–10 years).

Once these simulations have been run it is relatively simple to calculate the mean and variability of landscape structure in each scenario at each time step. Then it is straightforward to compare management alternatives in terms of the degree and rate to which they lead to changes from current landscape structure to desired landscape structure (Cushman and McGarigal 2006). In this comparison, calculation of distance from initial condition and from desired condition at each time step is very useful, as are calculations of divergence among scenarios and velocity of change, in total, and the component of velocity toward or away from desired conditions (Cushman and McGarigal 2006). Divergence is defined as the Euclidean distance between the location of different landscapes at the same point in time:

$$\overline{g}_j = \sqrt{\sum_{k=1}^{m} (X_{ajk} - X_{bjk})^2}$$

where x_{ijk} is the score for landscape i (a,b) at time j $(0\ldots\mathrm{t})$ on axis k $(1\ldots\mathrm{m})$. Like displacement, this is simply the Pythagorean theorem applied to m dimensions, the difference being that here the distance is between two different trajectories at the same point in time, instead of the distance a single trajectory has moved away from the starting point after any time period.

Velocity is a measure of both the rate and direction of change in the landscape trajectory:

$$\overline{v}_i = \Delta r_i / \Delta t$$

where Δr_i is the change in position vector for the ith landscape and Δt is the interval under consideration. Velocity can described as the rate and direction of movement between two points in m-dimensional landscape structure space. Velocity can also be expressed in component form:

$$\overline{v}_i = (\Delta x_{i1} + \Delta x_{i2} + \ldots + \Delta x_{im}) / \Delta t$$

where Δx_{ik} is the difference in scores for landscape i $(1\ldots\mathrm{n})$ on axis k $(1\ldots\mathrm{m})$ for that interval. In this form, velocity is defined as the rate of change along each dimension of the landscape structure space.

Trajectory analysis allows quantitative assessment of current landscape structure with respect to desired conditions, by calculating the *distance* in structure space between current and desired conditions. It also allows assessment of the displacement, divergence, velocity and acceleration of landscape change under alternative scenarios. Comparing the *displacement* of each scenario from initial conditions over time provided a means to evaluate the nature and extent of change from the original state. Computing the *divergence* among scenarios over time provided an objective measure of the relative differences among scenarios in their impacts on marten habitat area and fragmentation. Between displacement and divergence in landscape structure space we have the means to quantify the impact of scenarios both with respect to initial conditions and relative to each other. These are the two comparisons most needed by landscape managers to provide information on the effects land management alternatives. Also, quantifying *velocity* and *acceleration* through landscape structure space provides rigorous description of the rates and directions of changes in landscape structure. These measures facilitate quantitative comparison among scenarios and allow a quantitative comparison of which alternative management scenarios most effectively and rapidly lead to achieving the desired future landscape condition.

Landscape trajectory analysis is also the appropriate tool to monitor changes as management occurs. Simulation models produce expected results, but the results of such models must be treated as spatial hypotheses that must be verified and tracked. Thus, the key final component in landscape analysis is monitoring changes in landscape structure as they occur, comparing them to expectations from simulation models, and with movement toward or away from desired conditions. Using measures of *distance, divergence, displacement, velocity* and *acceleration* managers can monitor the changes in landscape patterns as they occur in a quantitative framework that allows direct comparison to desired conditions, and allows for evaluation of effectiveness of the current management approach. This is essential to provide the critical feedback needed to guide adaptive landscape management.

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Detailed Assessment Using Remote Sensing Techniques

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1 Design of Assessment

1.1 Measurements and Variables of Interests

The variables of interest in forest inventory usually determine the amount of trees by means of stem volume or biomass as well as stand structural information, health data or plant physiological data. To be able to estimate volume variables other characteristics, such as tree size (diameter), tree height or total number of trees may also have to be determined. Furthermore, forest growth, tree species and different area information (site classes, stand development stages) are usually considered. Forest inventory is usually based on principles of sampling theory. Typical examples of such inventories are National Forest Inventories (NFI's). As an exception of sampling based inventory is stand level inventory for forest management purposes. In Finland, forest management planning in private forests is usually based on information collected by forest compartments (stands) (Poso 1983). The area coverage of this method is usually 100%, i.e. it produces accurate spatial information on landscape level.

The stratification of the forest area into homogenous strata or stands has been used to improve the efficiency of field sampling in multi-phase sampling designs or in connection with the stand-wise forest inventory for forest planning. Sometimes photographs are used to measure only for stratification, but also for measuring so called remote sensed plots. Tarp-Johanssen (2001) distinguishes the digital analysis of forest images in two sections: 2D and 3D methods. 2D methods operate in the image domain of individual images alone whereas the 3D methods combine information from several images to operate in the physical 3D-object domain. This statement could be refined here such that 3D methods combine auxiliary information, which can also be other views, about the scene to operate in the object space (Korpela 2004).

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Visual interpretation of large-scale aerial photographs for single tree characteristics such as species, height, crown width and area, and breast height diameter has been tested in many experimental studies. Passive sensors, like twin-camera fixed-base, single-camera sequential photographic systems and digital photographic systems, have been used to observe visible part of view from airborne vehicle. Stereo model measurements have been a common practice in some countries, like Norway, Canada and Germany. Aerial images have already been used for decades in practical forestry in Finland for stand delineation, which still remains basically the only de facto standard in data collection for forest management planning purposes nowadays (Anttila 2005).

The most common practice when acquiring digital images is still to capture the images with an analogue frame camera and digitize them afterwards by scanning the film. Digital aerial cameras are becoming more common, but these systems still have some limitations, such as spatial resolution and the maturity of the technology for large-scale orthophoto production. Digital aerial cameras have started to replace analogue cameras, however (Leberl & Gruber 2003), and they can be assumed to be used practically always in conjunction with a Global Positioning System and Inertial Measurement Unit (GPS/IMU). This enables in-situ positioning of the camera and its rotations, i.e. an external orientation is available instantly for each image without further processing. Camera position and orientations can be used as a preliminary guess while resolving the eventual external orientation parameters, in some cases even without any Ground Control Points (GCP). This will naturally reduce the costs of image production.

Active sensors, like laser and other radar systems allow also the observe the non-visible part of forests, such as co-dominant and suppressed canopy layers. The widely used active RS technique for forest inventory purposes is currently Airborne Laser Scanning (ALS), a type of RADAR system that differs considerably in response from the conventional RS data sources. It produces a 3D point cloud which approximates the surface of the earth, i.e. the physical dimensions are measured directly. The basic 3D feature of laser data gives excellent possibilities for the utilisation of laser data in forestry. Each laser hit has x, y and z values which allows the determination of geographical location of the information. In nature the height value z describes the vertical structure of vegetation, usually trees in forestry applications.

Airborne ranging laser systems utilize light detection and ranging (lidar). Airborne profiling lidars have been used already in the 1980s to predict forest variables (e.g. Nelson et al. 1988). Subsequently, scanning sensors of small footprint diameter (usually from 10–30 cm) and discrete return have made it possible to obtain aerial coverage of accurate height information on the forest canopy (Næsset 1997). On the other hand, it is also possible to utilise spaceborne large footprint lidar (e.g. Drake et al. 2002) or terrestrial laser (Lowell et al. 2003) in forestry applications. However, here we concentrate only on airborne laser scanning applications.

In forestry applications, the typical flight altitude is 200–2000 meters above ground level. The scanning angle has usually varied between 7–15 degrees and the pulse density on the ground is mostly between 0.1–10 hits per square meter. An essential part of laser data processing includes the generation of digital terrain model

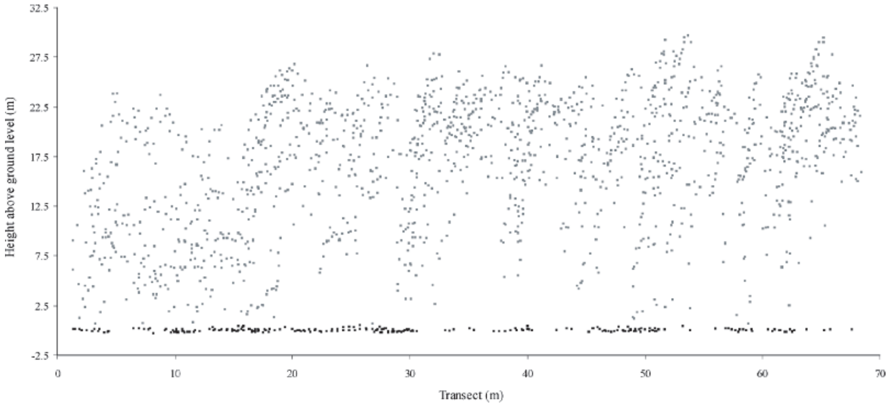

Fig. 1 Side view of a 5 meters deep transect of ALS data for a mature spruce stand acquired with an Optech ALTM 2033 sensor at an altitude of 1500 metres above ground level. Only the first pulses are depicted in the figure. The pulse density is about 0.7 measurements per square metre. The black dots represent pulses classified as ground hits

(DTM). It must be noted that lidar technology can simultaneously map the ground beneath the tree canopy as well as canopy height. To obtain vegetation height values either raster-based canopy height model (CHM) can be computed as the difference between the original digital surface model and the constructed DTM or the heights for laser points can be obtained by subtracting the corresponding DTM values. The DTM can be used in many other forestry applications as well, because it determines e.g. slope and small-scale ground height variation within forest stand.

The usability of laser is based on the fact that the percentiles of the height distribution of laser scanner data are related to the vertical structure of the tree canopy (e.g. Magnussen & Boudewyn 1998). The two main approaches to the derivation of forest information from laser scanner data have been the laser canopy height distribution approach, usually applied for low resolution data, i.e., pulse density below one hit per square meter and the individual tree based approach, usually applied for high resolution data (e.g. Næsset 2004).

1.2 Balance Between Field Data and RS Data in Data Collection

When a forest information system is established, the first task is the identification of information needs in attribute, time and at the area level. The most critical factor is to decide the smallest area unit of interest and proceed with suitable forest inventory methodology. The limitations of high-resolution data in regional surveys are mainly the cost and difficulties to automatically interpret the detailed information with complex texture (Hyppänen 1996). However, remote sensing has been used to classify or stratify forests, thus reducing variation and increasing sampling

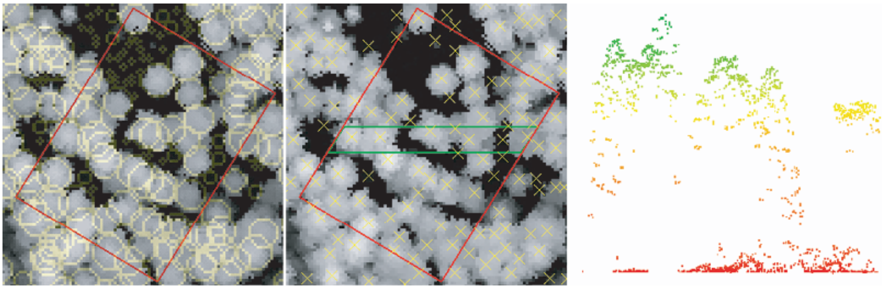

Fig. 2 Canopy height model and crown width circles modelled from height (left image), detected individual trees, and the laser point cloud of five meters wide cross-section (center image) of the first pulse data (right image)

efficiency in multilevel sampling designs. Forest inventory systems based on remote sensing and their implementations can be categorized in the following way:

a) Remote sensing oriented systems; the main attention has been paid to the identification of different forest classes, including species classes and volume categories, meant for thematic map production purposes (e.g. Walsh 1980; Horler & Ahern 1986; Brockhaus & Khorram 1992). Ground truth is collected from subjectively chosen training areas with some field observations.
b) Field data oriented systems; the calculation of inventory results is based on field sample plots and the inventory system has been established according to a predesigned sampling frame. Remote sensing data have been used as auxiliary information mainly in two-phase sampling schema (e.g. Poso et al. 1984, 1987; Köhl 1990) or in calibration approach (e.g. Mäkinen et al. 2006).
c) Updating oriented systems; the main interest is to fulfil the information needs of the database and control the quality of data. The base information has mostly been collected with more accurate inventory systems, and remote sensing data are used for monitoring purposes and to allocate field checking (Varjo 1997). The existing data can be utilized as auxiliary data in the planning of a data collection procedure.

The classification above characterizes only the main differences and features in the implementation of methods, although most of the methods can not strictly be categorized to only one group. Combining the mentioned systems is not common and there are only few studies, where whole a remote sensing based forest inventory system uses both proper field data and auxiliary map data sources. The main methods for the estimation of forest characteristics have been stratification (or classification), regression analysis, and non-parametric estimation using reference field sample plots. The remote sensing material can be also used for direct measurements of forest objects. Variables like tree height and crown width can be measured with stereoscopic instruments or pattern recognition techniques.

As forest information is required for units of a defined size (i.e. 1–10 ha), the same sampling design alternatives as for inventories on the compartment level can be considered. Stratified sampling may results in estimates for units that show the same characteristics and may vary considerably in size. The sampling design alternatives for stand level could be e.g.

- Simple random sampling (only field assessments)
- Regression estimates (combining data from field samples and complete high resolution remote sensing data by regression techniques)
- Two phase sampling with regression estimators (combining data from field samples and samples in aerial photography by means of regression techniques)

The attribute used to compare sampling design alternatives should be selected. The optimization process of inventory concept needs several assumptions:

(1) The correlation between attribute estimates in remote sensing products and field data
(2) The cost for remote sensing information for target area
(3) The variance of variable of interest
(4) The size of the unit of reference
(5) The cost for the assessment of on field plot and remote sensing data based plots needs to be set
(6) Assumptions about registration errors of field plots and remote sensing based plots

It is evident that the accuracy of stand attributes is closely related to the costs of the inventory chain. Furthermore, it has been suggested that the choice of method and the intensity and timing of the inventory should be based on the costs of inventory design and the economic losses that can be expected to arise from poor decisions based on erroneous data. This concept is called "decision-oriented inventory planning", and cost-plus-loss analyses have been proposed as a viable solution for achieving this (Cochran 1977; Ståhl et al. 1994; Eid 2000). Cost-plus-loss analyses have also been used to evaluate RS-based forest inventory systems (Eid et al. 2004; Juntunen 2006). Thus, high accuracy in forest variable estimation is not an objective in itself, as costs must always be considered as well.

Only part of the necessary information can be collected using the inventory techniques based on aerial photography and the accuracy has in general been poorer than that achieved with traditional field inventories. There are therefore only a few operational inventory systems which rely only on use of aerial photographs. In any case, aerial photographs have been used extensively as a part of the inventory chain, as explained earlier. It must also be remembered that field measurements will always play an important role in RS-based forest inventories. The increasing precision of remote sensing information can reduce need of field data significantly and pay more attention to field variables which can not be observed with remote sensing data (rare existing plant, biodiversity variables etc.).

It has been indicated in several studies that essential growing stock characteristics, e.g. mean height, basal area and stand volume, can be accurately predicted using ALS data (Magnussen & Boudewyn 1998; Lim et al. 2003; Næsset 2004), and as the costs of ALS data have decreased considerably (Eid et al. 2004), ALS-based forest inventory applications have emerged as an interesting and realistic alternative for operative forestry purposes (see Næsset 2004b). Concerning total characteristics, such as the total volume of all tree species, the results have been even better than those achieved with traditional stand level field inventory techniques. There are also some studies in which species specific growing stock has been estimated. For instance, Packalén (2006) presents an approach to predict simultaneously volume, stem number, basal area and diameter and height of the basal area median tree for Scots pine, Norway spruce and deciduous trees and for total characteristics as sums of the species-specific estimates by combining ALS data with aerial photographs. The accuracies achieved in the estimation of species-specific characteristics were as good as those achieved in current inventory practise, and in the case of total volume estimates the accuracy was even substantially better.

1.3 Inventory Strategy: from Trees to Stand Data

Two strategies are commonly applied in remote sensing based stand inventories. Either trees are directrly interpretated from remote sensing data or stand level remote sensing variables are used to estimate size distribution of trees.

When trees are interpretated directly, image segmentation techniques are used to delineate individual trees. The interpretation of aerial photographs at the single tree level have been studied substantially (Gougeon 1995; Dralle & Rudemo 1996, 1997; Pollock 1994, 1996; Larsen 1997, 1999a, 1999b; Larsen & Rudemo 1997, 1998; Brandtberg & Walter 1998; Brandtberg 1999, 2002; Wulder et al. 2000; Pitkänen 2001; Haara & Nevalainen 2002). Forest inventory systems are also being developed to incorporate automated or semi-automated aerial photo interpretation (Pinz 1999).

In the case of ALS data if the number of laser pulses is increased to more than, say, 5 measurements per square metre, individual trees can be recognised (Brandtberg 1999; Hyyppä & Inkinen 1999; Persson et al. 2002; Popescu et al. 2002; Solberg et al. 2006). For example, in the study by Hyyppä and Inkinen (1999) the number of laser pulses was increased to more than 10 measurements per one square metre to distinguish individual trees. Furthermore, Holmgren and Persson (2004) have shown that it is even possible to recognise species of individual trees.

1.4 Inventory Strategy: from Stand to Tree Data

Tree level measurements using field inventory are usually too laborious and expensive to apply operationally. For this reason diameter distributions are often used as surrogates of individual tree measurements and these distributions are estimated based on stand level attributes. Similar kind of approach can also be used with

Table 1 Examples of outlined inventory schemes (models) utilising photogrammetric individual tree measurements or 3D laser measurements. (see Korpela 2004)

Work Phase	Inventory scheme		
	A - No field visits	**B** - Sample of field plots	**C** - Field visits in stands of interest
I Preparatory work	Image (and laser data) acquisition, and planning of inventory		
II Field inventory	–	{Position, sp, d, h, C_W} – field observations e.g. from systematic layout of field plots. Satellite positioning and plot measurements	{Position, sp, d, h, C_W} – field observations in every stand. Satellite positioning and plots measurements
III Use of Remote Sensing Data	Semi-automatic measurement of tree top position and height		
IV Use of Remote Sensing Data	–	Training sets for image based species classification	Training sets for image based species classification
V Use of Remote Sensing Data	Interpretation of tree species and measurement of crown width		
VI Calculation visible objects	Indirect estimation of dbh and derivation of volumes. (Calibration/validation procedures in B and C).		
VII Calculation on non-discernible objects	The estimation of forest variables for the non-discernible tree stratum.		
VII Calculation of stand level attributes	Computation of the aggregated results, density maps etc. for stands or other reporting units		

remote sensing material. Numerous approaches to describing diameter distributions have been presented in recent decades, e.g. those of Bailey and Dell (1973); Hafley and Schreuder (1977); Borders et al. (1987); Maltamo and Kangas (1998), who introduced the use of Weibull and Johnson SB functions, percentiles and non-parametric distributions, respectively. One special approach, adopted mainly in Finland, has been the use of the basal area diameter distribution (Maltamo & Kangas 1998: see also Gove & Patil 1998), the main idea being that, instead of using tree frequencies, the distribution can be estimated and scaled according to the basal area of the underlying trees. Recent developments have included an advanced methodology to estimate Weibull distribution parameters (Gove 2003; Cao 2004) and the measurement of quantile trees to calibrate the predicted distribution (Mehtätalo 2005; Mehtätalo & Kangas 2005). ALS data have also been used to construct theoretical diameter distribution models in Norway and Finland (Gobakken & Næsset 2004; Maltamo et al. 2006a). Both studies employed the Weibull function and percentile-based distributions and were based on the use of basal area diameter distribution.

Automated stand delineation task by image segmentation and estimation of forest variables using spectral and textural information (Holopainen 1998; Muinonen et al. 2001) has been used to calculate stand characteristics. Forest variables are derived either from image pixel level estimates (Holopainen 1998) or directly from mean stand values (Anttila 2002). Similary, laser data variables are calculated for specific grid cells (Næsset 2004a), plots (Packalén 2006) or directly to the whole stand (Suvanto 2005).

2 Tree Attribute Mapping

The development of automatic tree level inventory methods has attracted a lot of attention in recent years. These methods rely on the delineation of individual tree crowns to segments. Both 2D and 3D techniques have been introduced for this purpose. The idea behind individual tree detection is that their crown dimensions can be correlated with tree characteristics of primary interest such as diameter at breast height and tree height, after which generic models that use diameter and height as explanatory variables may be used to estimate the stem volume. In 2D methods only the crown diameter or area is used to estimate other tree characteristics, while in 3D methods tree height can also be determined. Because of the direct determination of height, instead of estimating it using crown size, 3D methods produce much better estimates for stem characteristics of interest. Tree level inventory approaches produce ideal input data for forest management systems which operate at the single tree level. One disadvantage, however, is that suppressed trees which are not visible from the above cannot be detected and the individual tree interpretation is also generally difficult in a dense forest. Also, tree level inventories often have high costs attached to them.

A common 2D method for isolating tree crowns is first to detect the tree tops by assuming that, after proper smoothing, the local intensity maxima on the aerial

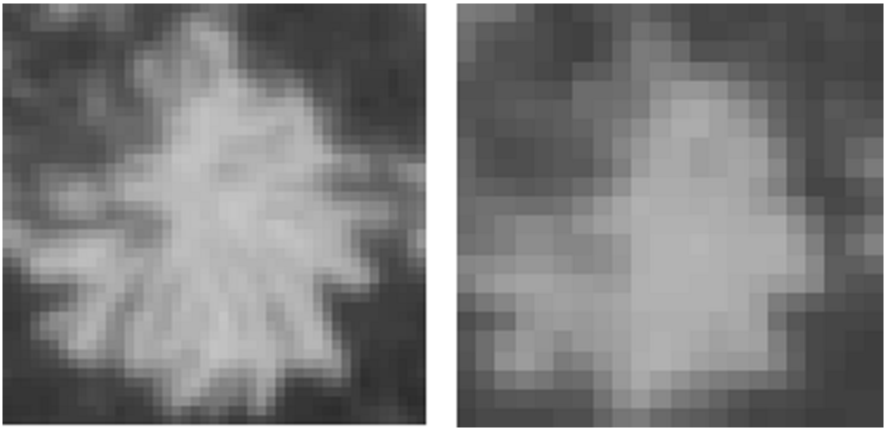

Fig. 3 1:5000 and 1:10000 near-nadir views to a dominant spruce tree. pixel sizes on ground are 12.5 and 25 centimetres respectively. The diameter of the crown is approximately 4.5 metres. (Korpela 2004)

images will represent tree tops, after which the crowns are segmented, e.g. by means of a region-growing algorithm using local maxima as seed points (Dralle & Rudemo 1996; Lowell 1998; Wulder et al. 2000; Pitkänen 2001; Anttila & Lehikoinen 2002). Template matching is a method in which the presence of predefined instances of sub-images, called templates, e.g. trees, are detected in the image. The template is moved over the image and a measure of similarity is used to determine whether an

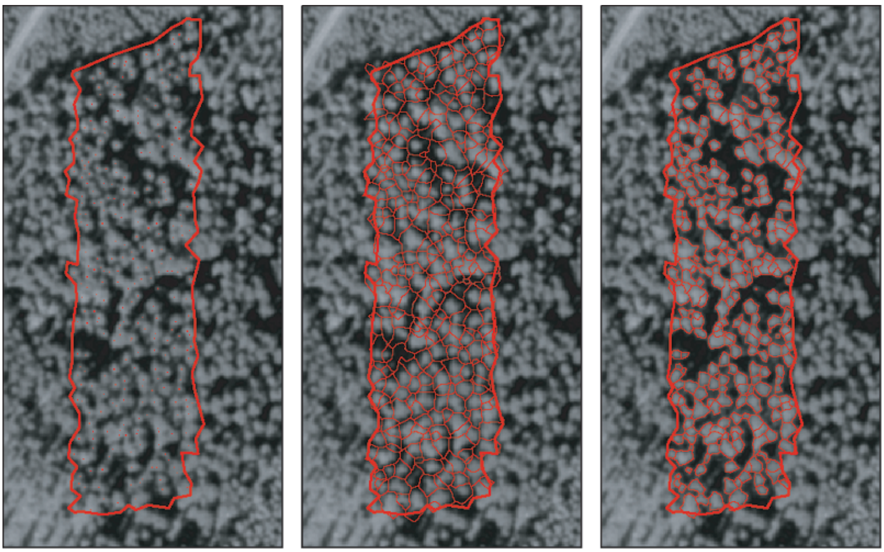

Fig. 4 Identification of tree top candidates, watershed delineation of objects and removal of background from tree objects (Ikonen 2004)

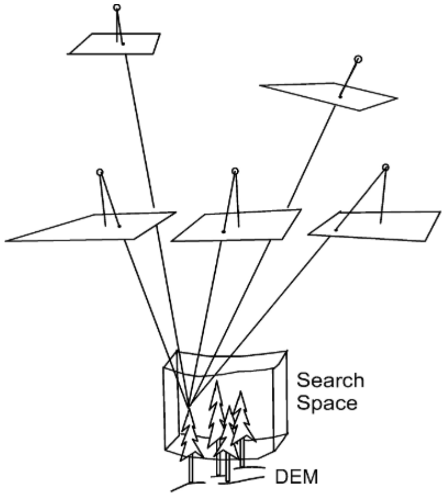

Fig. 5 Multiple images can be used in extraction of tree attributes (Korpela 2004)

object, e.g. a tree, exists in a particular location or not. To speed up the algorithm, prior information about candidate locations may be used. This type of technique has been developed and tested by many authors (Pollock 1996, 1999; Larsen 1997; Larsen and Rudemo 1998; Olofsson et al. 2006). Korpela (2000, 2004) extended the template matching approach used by Larsen and Rudemo (1998) to the 3D application, which enables tree height determination as well. Template matching can take into account the effect of varying image-object-sun orientation, which cannot be considered in the local maxima approach (Korpela 2004). Other approaches to the detection of individual tree crowns are valley following (Gougeon 1995), multiple-scale analysis (Brandtberg & Walter 1998) and edge detection (Pinz 1999).

The methods used in single tree detection in laser scanning have been merely applied from similar studies using high and very high resolution aerial imagery (see e.g. Hyyppä et al. 2007 for review). Brandtberg (1999); Hyyppä and Inkinen (1999) were the among the first to demonstrate the individual tree based forest inventory using laser scanner using tree finding with maximas of the CHM and segmentation for edge detection. Persson et al. (2002) further improved the crown delineation. In their study the proportion of correctly detected trees was 71%. In the same study area as Hyyppä and Inkinen (1999); Maltamo et al. (2004) the proportion of detected trees was only about 40%. This was due to the multilayered and unmanaged stand structure of the study area (Fig. 6). Some other attempts to use Digital Surface Model (DSM) or CHM image for individual tree crown isolation or crown diameter estimation have reported by e.g. Brandtberg et al. (2003); Leckie et al. (2003); Straub (2003); Popescu et al. (2003); Morsdorf et al. (2003); Tiede and Hoffman (2006).

One problem of individual tree detection on aerial images or on raster canopy height models is handling of tree crowns of different sizes. On laser scanner data one size attribute, height, is directly available. This gives possibilities to develop

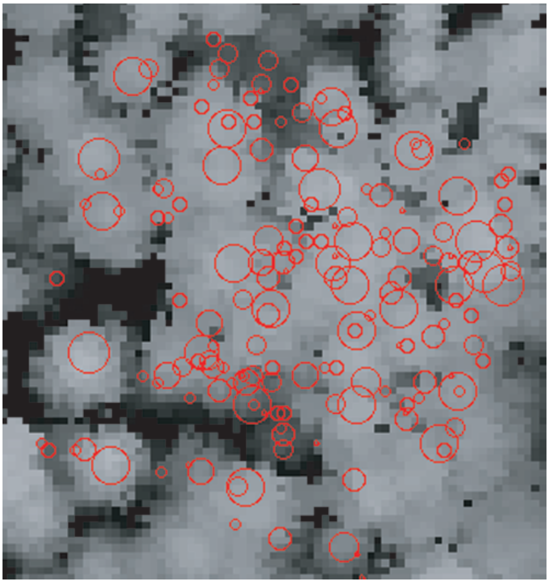

Fig. 6 Field measured trees (circles according to tree size in DBH) and corresponding CHM in unmanaged forest

processing methods that adapt to the object size. In the study by Pitkänen et al. (2004) three adaptive methods were developed and tested for individual tree detection on CHM. In the first method, the CHM was smoothed with canopy height based selection of degree of smoothing and local maxima on the smoothed CHM were considered as tree locations. In the second and third methods, crown diameter predicted from tree height was utlised. The second method used elimination of candidate tree locations based on the predicted crown diameter and distance and valley depth between two locations studied. The third method was modified from scale-space method used for blob detection. Instead of automatic scale selection of the scale-space method, the scale for Laplacian filtering, used in blob detection, was determined according to the predicted crown diameter.

Holmgren and Persson (2004) stated that it is possible to separate Scots pine and Norway spruce using ALS data at individual tree level. The portion correctly classified trees on all plots was 95%. Moffiet et al. (2005) proposed that the proportion of laser singular returns is an important predictor for the tree species classification. Brandtberg et al. (2003) used laser data under leaf-off conditions for the detection of individual trees. Additionally, tree species classification results of different indices suggest a moderate to high degree of accuracy using single or multiple variables between classifications of deciduous trees.

One specific feature of ALS based single tree detection is that tree height estimates are underestimated. Hyyppä et al. (2007) reviewed the literature concerning this aspect. Hyyppä and Inkinen (1999) reported individual tree height estimation with an RMSE of 0.98 m and a negative bias of 0.14 m while Persson et al. (2002) reported an RMSE of 0.63 m and a negative bias of 1.13 m. Both test sites were dominated by Norway spruce and Scots pine. Persson et al. (2002) further explained their greater average tree height underestimation as resulting from lower ALS sampling density (ab. 4 pulses per m^2). Naesset and Okland (2002) stated that the estimation accuracy significantly decreased by a lower sampling density. Gaveau and Hill (2003) reported the tree height estimation of broadleaf woodland, since the previous studies had concentrated on coniferous forests. A negative bias of 0.91 m for sample shrub canopies and 1.27 m for sample tree canopies was observed. Leckie et al. (2003) concluded that some of the 1.3 m underestimation could be accounted by the undergrowth. Yu et al. (2004) found a systematic underestimation of tree heights by 0.67 m for the laser acquisition carried out in 2000 and 0.54 m from another acquisition in 1998. Since the accuracy of conventional field measurements may not be adequate for detailed assessment of the bias, Maltamo et al. (2004) used 29 pines accurately measured with a tacheometer, and found a 0.65 m underestimation of height for single trees, including annual growth that was not compensated for in the plot measurements. They also found that the precision of 0.50 m for individual tree height measurements was better than reported earlier (e.g. Hyyppä & Inkinen 1999; Persson et al. 2002). In the studies of Rönnholm et al. (2004); Brandtberg et al. (2003) it was shown that the tree height can also be reliably estimated even under leaf-off conditions for deciduous trees.

Possibility to characterize suppressed trees that cannot be detected has also been of interest in some studies. The use of original point clouds instead of DSMs or CHMs makes this possible. Since, some of the laser pulses will penetrate under the dominant tree layer, it is also possible to analyze multilayered stands. In Maltamo et al. (2005), the existence and number of suppressed trees was examined. This was carried out by analysing the height distributions of reflected laser pulses. Histogram tresholding method of Lloyd was applied to the height distribution of laser hits in order to separate different tree storeys. Finally, the number and sizes of suppressed trees were predicted with estimated regression models. The results showed that multilayered stand structures can be recognised and quantified using quantiles of laser scanner height distribution data. However, the accuracy of the results is dependent on the density of the dominant tree layer.

Correspondingly, Mehtätalo (2006) used a theoretical approach to describe small trees. The probability of a tree being observed was related to its height and was equal to the proportion of the forest area not covered by taller trees. Mehtätalo (2006) presented mathematical formula which was based on the following assumptions: (i) trees are randomly located within the stand and crown diameters within a stand are uncorrelated, (ii) tree height increases as a function of crown diameter, (iii) the tree crown forms a circle around the tree tip, and (iv) a tree is invisible if the tree tip locates within the crown of a taller tree. Furthermore, different approaches were proposed for the correction of the censoring effect upon the observed distribution of crown areas. The used approach provided accurate estimates for the distribution of crown areas and the number of stems.

Yu et al. (2004) demonstrated the applicability of airborne laser scanners in estimating forest growth and monitoring harvested trees, using datasets with a point density of about 10 points/m^2 over a two-year period. Out of 83 field-checked harvested trees, 61 were detected automatically and correctly (Fig. 7). All the mature harvested

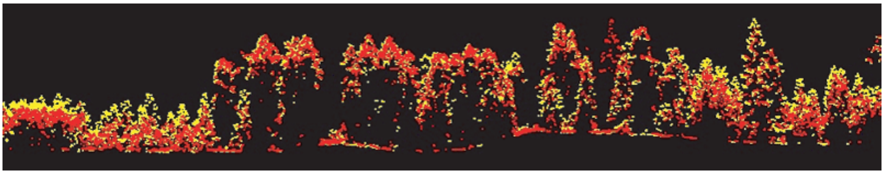

Fig. 7 An example of 3 m wide canopy profiles for a 150 m long cross-section of ALS data. Height growth of individual trees as well as one fallen tree can be observed. Light gray: ALS data 2003, Dark gray: ALS data 1998 (Yu et al. 2006)

trees were detected; it was mainly the smaller trees that were not. Forest growth was demonstrated at plot and stand levels using an object-oriented tree-to-tree matching algorithm and statistical analysis. St-Onge and Vepakomma (2004) concluded that sensor-dependent effects are probably the most difficult to control in multi-temporal laser surveys for growth analysis purposes. Naesset and Gobakken (2005) concluded that over a two-year period, the prediction accuracy for plotwise and standwise change in mean tree height, basal area and volume was low when a point density of about 1 point/m^2 and canopy height distributions were used. They also reported that certain height measurements, such as maximum height, seemed less suitable than many other height metrics because it tends to be less stable. In Yu et al. (2006) 82 sample trees were used to analyse the potential of measuring individual tree growth of Scots pine in boreal forest. Three different type of variable was extracted from the point clouds representing each tree; they were the difference of highest z value, difference between DSMs of tree tops and difference of 85, 90 and 95% quartiles of the height histograms corresponding to a crown. The results indicate that it is possible to measure the growth of an individual tree with multi-temporal laser surveys.

ALS-based tree-level forest inventories may become a realistic alternative in the near future. Tree-level inventories require denser ALS but technological development will mean that costs will decrease rapidly. An approach in which aerial photographs are not needed for species recognition would also be interesting at the individual tree level, but more development work must still be done in the fields of individual tree recognition, tree species classification and the inventory chain as a whole before tree-level inventories can be valid operationally.

3 Stand Attribute Mapping

3.1 Data Collection Unit

The unit for which stand attributes are to be estimated may be an operative stand, a "microstand" or a plot. A microstand is a homogeneous patch of an area that is usually smaller than an operative forest stand (Hyvönen et al. 2005). Microstands are always created using image segmentation techniques, and thus are also referred to as (forest) segments. The major difference between microstands and operative stands is that microstands are not designated to be used as basic treatment unit; they are used for inventory purposes only. Even though all methods entail a natural inventory unit, the applicability of the method is not tied to this particular level. Plots, for instance, can be aggregated to form stands, e.g. using a grid approach where each cell in the grip represents a plot. Due to averaging, up-scaling generally increases the accuracy at the aggregated level.

3.2 Visual Interpretation

Remote sensing based forest inventory methods can be categorized in many ways. One natural classification is to divide them into techniques that use visual interpretation and those that use computer-aided interpretation. In the era of analogue photographs it was naturally only visual interpretation that was used. These methods usually employed photogrammetric techniques which allow 3D interpretation, e.g. measurement of tree height. In the era of digital images visual interpretation is still frequently used, e.g. in stand delineation.

The forest inventory approaches in which visual interpretation of aerial photographs has played an important role are mainly based on two-phase sampling. The principle is that aerial photographs are used for stratification in the first phase and in the second phase a subset of plots or stands are measured in the field and results are generalized to the whole area (Poso 1972; Mattila 1985; Spencer & Hall 1988; Biggs & Spencer 1990). This scheme has also been used operationally in many countries, especially in large-area forest inventories, e.g. in northern Finland within the Finnish NFI. Numerous other visual interpretation methods for forest inventory purposes have also been studied at the tree and stand levels using both 2D and 3D techniques during past decades (Nyyssönen 1955; Aldrich et al. 1969; Talts 1977; Needham & Smith 1987; Næsset 1991; Gagnon et al. 1993; Anttila 1998). According to Anttila (2005), Nordic research into visual interpretation has mostly been aimed at stand level inventories.

Typically in Scandinavia, forest stands are delineated before the field inventory using aerial photographs, and the stand borders are corrected during the fieldwork as required. Digitized colour-infrared (CIR) photographs are typically used for delineation purposes. The use of CIR images enables especially easy interpretation of broadleafed trees. Stand characteristics by tree species are assessed in the field using subjectively placed angle-count sample plots, i.e. large trees are more likely to be sampled. The forest stock characteristics to be estimated are mean age, basal area or stem number and the diameter and height of the basal area median tree. All other stand attributes, such as growing stock volume, are estimated from these measurements employing theoretical diameter distributions (see 1.4).

3.3 Computer-aided Interpretation

The most commonly used feature in remote sensing and image analysis is the spectral feature describing the tonal variation in portions of the electromagnetic spectrum, i.e. the radiance observed by the camera system. Spectral features on aerial images are often referred as tones, or just pixel values. Unfortunately, the observed radiance is affected by several forms of spectral distortion, such as light fall-off effects and variations in atmosphere and imaging geometry (King 1991; Pellikka 1998; Lillesand et al. 2004). In addition, the camera system, film and post-processing can cause undesirable variation in the pixel values. The most significant source of spectral distortion, at least where forests are concerned, is the bi-directional reflectance effect. This can be perceived in aerial photographs as variation in brightness caused by the fact that the tree crowns in the direction of incoming radiation expose their shady sides to the camera and those in the opposite direction their illuminated sides (Holopainen & Wang 1998). Due to spectral distortion, the same object is not represented with congruent pixel values in different parts of the image or in different images. When tone is used in the computer-aided interpretation of forest characteristics it is essential to minimize these spectral anomalies. Several empirical methods in particular and also some semi-empirical ones have been developed and tested for this purpose.

Another widely used feature of pictorial data is texture, which contains information about the spatial distribution of tonal variations within an image. According to Tuceryan and Jain (1999), texture measures can be divided into four categories: statistical, geometrical, model-based and signal processing, in which probably the most commonly used are statistical methods. First-order statistics estimate properties of individual pixels (e.g. variance) without taking into account spatial interaction between neighbouring pixels, whereas second and higher-order statistics estimate properties of two or more pixel values occurring at specific locations relative to each other (Ojala & Pietikäinen 2001). The most widely used statistical method is that introduced by Haralick et al. (1973), a second-order method that relies on a grey-tone spatial dependence matrix, from which all the texture measures are derived. Haralick's textural features have been used extensively in remote sensing and forest inventory applications, although numerous other textural methods have been tested, too (Weszka et al. 1976; Marceau et al. 1990; Franklin & Peddle 1990; Muinonen et al. 2001; Coburn & Roberts 2004; Tuominen & Pekkarinen 2005). The reason for the use of texture is that it makes sense to utilize all the meaningful information available in the images, and tone itself is not concerned with spatial variation at all. The combined use of tone and texture has often been found to improve the accuracy by comparison with the use of tone alone.

Automatic forest inventory methods which exploit aerial photographs at the stand, microstand or plot level mostly employ tone and texture, but quite different approaches have been adopted, too, such as photogrammetric 3D methods (Korpela & Anttila 2004). One advantage of extracting features from a larger area, e.g. a plot, is that a single pixel in a high-resolution image does not contain as much information as a spatially grouped set of pixels. Thus, the mean pixel value and the variation in pixel values within a plot, stand or sometimes microstand are used as explanatory variables when estimating forest attributes. To mention a few recent studies, approaches of this kind have been employed in Hyyppä et al. (2000); Shugart et al. (2000); Holmström et al. (2001); Anttila (2002); and Hyvönen et al. (2005).

ALS data can also be utilized in stand level, however, most often the basic calculation unit is a plot. After the calculation of stand variables to each grid cell the results can be averaged to stand level as well. At plot level canopy height distribution approach with low-resolution data is applied (Næsset 1997, 2002; Lim et al. 2003; Holmgren 2004). The size of the grid in relation to ALS data is dependent of the point density used. For example, the use of ALS data including 1 laser hit per square meter already results about 250 hits for circular sample plot which radius is 9 meters.

To apply canopy height distribution approach, the orthometric laser scanning heights are converted to above-ground heights (i.e. canopy heights) by subtracting the DTM at the corresponding location. The ground hits are excluded by assuming that any point with a canopy height less than e.g. 2 meters is a ground hit and that the remaining points are canopy hits. After that the first and last pulse height distributions for each sample plot can be created from the canopy height hits. For example, 5, 10, 20,. . ., 90, 95 and 100% $(\mathrm{h}_5, \ldots, \mathrm{h}_{100})$ percentiles for canopy height can be computed (see Næsset 2004a) and the corresponding proportional canopy densities $(\mathrm{p}_{05}, \ldots, \mathrm{p}_{100})$ are also calculated for these quantiles. In addition, the proportions of canopy hits vs. ground hits are usually determined. All metrics are calculated separately for the first and last pulse data. For example, h_{10} means the height value where 10% of all laser hit heights of vegetation are accumulated. Correspondingly, p_{10} is a value between 0 and 1 indicating the proportion of laser hits which is accumulated at the height of 10%.

Above mentioned characteristics of the distribution of canopy heights has been used as predictors in regressions or non-parametric models for estimation of mean tree height, basal area and volume (e.g. Lefsky et al. 1999; Magnussen et al. 1999; Means et al. 1999; Næsset 1997; Naesset & Okland 2002; Næsset 2002; Lim et al. 2003; Maltamo et al. 2006b). In addition to the prediction of stand mean and sum characteristics diameter distributions of a forest stand has also been predicted by using the statistical canopy height distribution based approach (e.g. Gobakken & Næsset 2004; Maltamo et al. 2006a).

Packalén & Maltamo (2007) applied the non-parametric k-MSN method to predict species-specific forest variables volume, stem number, basal area, basal area median diameter and tree height simultaneously for Scots pine, Norway spruce and deciduous trees as well as total characteristics as sums of the species-specific estimates. The combination of ALS data and aerial photographs was used in this study. The predictor variables derived from the ALS data were based on the height distribution of vegetation hits, whereas spectral values and texture features were employed in the case of the aerial photographs. The k-MSN method is a non-parametric nearest neighbour method which uses canonical correlation analysis to produce a weighting matrix for selecting the k Most Similar Neighbours from the reference data, i.e. observations which are similar to the object of prediction in terms of the predictor variables. The k-MSN method is the same as the MSN method described by Moeur and Stage (1995) except that the estimates for the target observation are calculated as weighted averages of the k nearest observations. The results showed that this approach can be used to predict species-specific forest variables at least as accurately as from the current stand-level field inventory for Finland (Fig. 8).

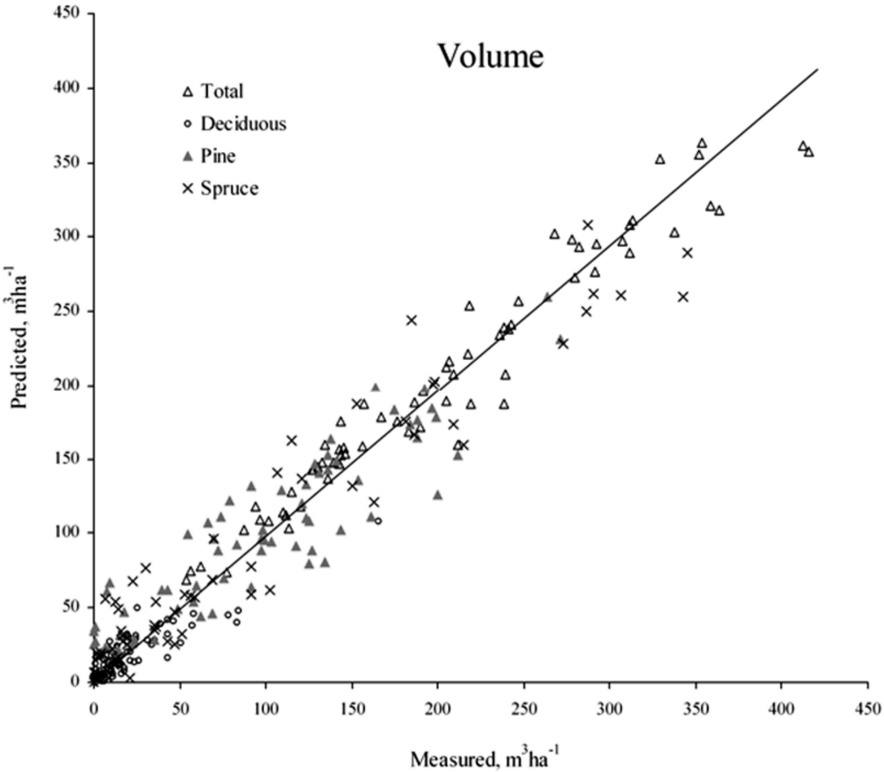

Fig. 8 Measured versus predicted volumes by tree species and total volume averaged at the stand level (Packalén & Maltamo 2007)

4 Conclusions and Future Prospects

Detailed structure of forest can be assessed with alternative techniques and strategies. The forest inventory task always try to assess many variables and partly field work can be replaced with remote sensing. If commercial growing stock is under interest, need for field work can be very small (Korpela & Tokola 2006). The quality of remote sensing based information is increasing and new technology make cost efficient use of high-resolution data possible. The reliability of RS-based estimates depends, however, always on forest structure and variability. Increasing number of species and multiple canopy layer structure decrease accuracy (Table 2).

In the near future, forest managers and decision makers can expect that accurate timber sortiments specific tree data and stand terrain information is available as a standard product. The inertial GPS devices in airborne vehicles will reduce need for manual work during the image processing and digital sensors provide multiple views to forest objects. The development in airborne laser scanning has been very fast and costs in data acquisition have already decreased substantially. The data of spectrometers and SAR data will bring own feature to this field. The current digital

Table 2 Subjective outlook of reliability figures of different remote sensing based stand volume estimates in forest inventory areas categorized according to stand size and heterogeneity of the area (CV = coefficient of variation)

	Stand level	Optical satellite (eg. Landsat TM)	Radar (eg. JERS)	Aerial Photographs, <1:20000	Laser scanning
		RMSE, %	RMSE, %	RMSE, %	RMSE, %
Large stands, large CV%	Total volume/biomass	35	35	30–35	5–15
	Species, 3 different	70	80	60	
Large stands, small CV%	Total volume/biomass	25	25	15–25	5–10
	Species, 3 different	55–70	60–80	70	
Small stands, large CV%	Total volume/biomass	40–60	40–60	35–45	10–20
	Species, 3 different	100	100	75–90	
Small stands, small CV%	Total volume/biomass	35–50	35–50	25–35	10–20
	Species, 3 different	90	100	70–85	

airborne sensors can easily adapt more wider spectral range and detailed spectral resolution. IFSAR techniques will be important when weather conditions are difficult and large areas inventoried. There are several promising results in large stand landscapes for timber volume prediction, although species specific results are difficult to obtain (Santoro et al. 2006; Magnusson 2006). The data fusion techniques and the emergence of high resolution remote sensing technologies (Magnusson 2006) will definitely improve our chances to make better decisions than earlier. The major challenges are still in species identification in uneven-aged and mixed natural forests.

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Assessing Landscape Attributes

Christoph Kätsch

1 Introduction

The future of land and land use is one of the most challenging global issues of the 21st century. Trends in some important key factors driving land use and land management like the permanent growth of global population reaching 8 Billion in 2020 (Anonymous 1999), urbanisation and others indicate that the demand for land, water and biological resources will increase significantly in the next few decades. It is a global problem, but developing countries in particular, with emerging economies will face severe pressure on their natural resources with unpredictable consequences for functioning and stability of the natural systems. Rapid growth connected with strong dependencies on natural resources threatens the natural, economic and social equilibrium in these countries. In addition contemporary concern with climate change adds an element of fundamental uncertainty on future development in these regions.

In order to meet the challenges, proper land use planning must be seen as an indispensable prerequisite for balancing land use and its adaptation to limited resources and increased demands. In comparison to typical land use planning approaches in the developed regions of the globe the planner in developing countries often has to deal with a host of fundamental problems which include issues regarding food security, unemployment and a more or less uncontrolled migration of poor population into the booming urban areas. The dynamic developments in areas such as these make it quite difficult if not impossible to implement plans and local as well as regional authorities' often loose control of the rapid changes in land use. Uncontrolled residential development in rural areas for instance is a severe problem affecting both the natural as well as the cultural landscape of the region.

Planning is generally based on informational processes irrespective of the context in which it is done. The planner needs to be in possession of all relevant knowledge and information regarding the present and future state of the object under planning in order to compile a plan suitable to meet objectives and goals. Planning of future

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land use or green landscapes therefore normally follows a multi-stage approach starting with the setting of objectives for the landscape to be developed followed by collecting and analysing all relevant data about the landscape and its elements. Land use options for the planning region then have to be assessed, decided and compiled into the final plan. All phases mentioned are driven by information which must be seen as a key resource for appropriate planning and decision making.

Landscapes however are complex systems which incorporate several coupled subsystems including the natural system and the socio-economic system which is driven by a human factor. Modern science is far away from fully understanding the systems dynamic and the consequences of interactions between the natural and social subsystems. It is often hampered by missing or poor quality of actual data as well as methodological and analytical difficulties emerging from the systems complexity. Nevertheless progress in many scientific disciplines has contributed to a better understanding of landscapes in some parts.

As far as collecting data about landscapes is concerned, modern Remote Sensing Techniques need to be mentioned in this context. The better perspective from above leading to an objective representation of the landscape at a specific point in time along with sophisticated digital image processing techniques are key-technologies in modern landscape inventory and monitoring. In the processes of transforming the raw data in information, modern Geographic Information Science provides powerful and effective tools for storing, handling and analysing landscape data and its spatial relationships.

Rules for capturing, processing and analysing relevant data however come from the scientific discipline dealing with specific land cover types or land use systems respectively. Forest science for example has its own highly developed metrics system which is based on more than 200 years of experiences in Forestry from all around the globe. More recently methods and indicators describing the edges between different land cover types have been developed. They allow a more comprehensive, multidisciplinary approach when looking for complete landscapes.

This paper presents the current state of the art of inventorying green landscapes in general but with a specific focus on forest landscapes. Information gathered during this phase of landscape design is needed in initializing the state of the landscape for a specific point in time. Due to a very wide field of applications and a large number of scientific disciplines involved it will not be possible to consider all relevant publications.

In the third chapter some examples of inventorying and planning landscapes dealing with specific land use problems in developing countries will be presented. With the south western parts of South Africa as example, the challenges these countries will have to handle typical for most countries on the African continent will be discussed. On the one hand side this country can be seen as one of the mayor drivers in global nature conservation. Due to a unique diversity in flora and fauna the country hosts several World Heritage Sites (e.g. Table Mountain and the Greater St Lucia Wetlands Park) of which one is situated within the Fynbos Floral Kingdom in the south western part. Beyond this more than 30 million hectares of forests and woodlands in the northern and eastern parts of the country form an important resource base for rural communities. On the other hand scarce water resources, high exposure to natural disasters like wild fires and enormous pressure caused by a booming economy and a continuous influx of immigrants from other African countries are some of the major problems leading to severe changes in the landscape.

2 Methods and Techniques in Assessing Landscape Attributes

Initializing landscape attributes in a planning context is a multidimensional problem dealing with conceptual, informational and methodological questions. From the conceptual perspective it should be considered that understanding the complex interaction between land resources, land cover and the human impact on it requires a systems analytical approach. An isolated view on a specific type of land use like forestry, agriculture or others might be misleading in this context. Landscapes are complex systems and only a comprehensive approach will allow identifying all entities and the relationships which define interdependencies between different landscape elements. Beyond this it forms a methodological precondition for managing the relevant information with the help of modern information systems.

At a more abstract level the landscape system is basically formed by three components as generalized entities namely Land, Land Cover and Land Use (Fig. 1).

Every component has its spatial dimension and its properties can be described with a more or less diverse set of attributes. According to a definition given by

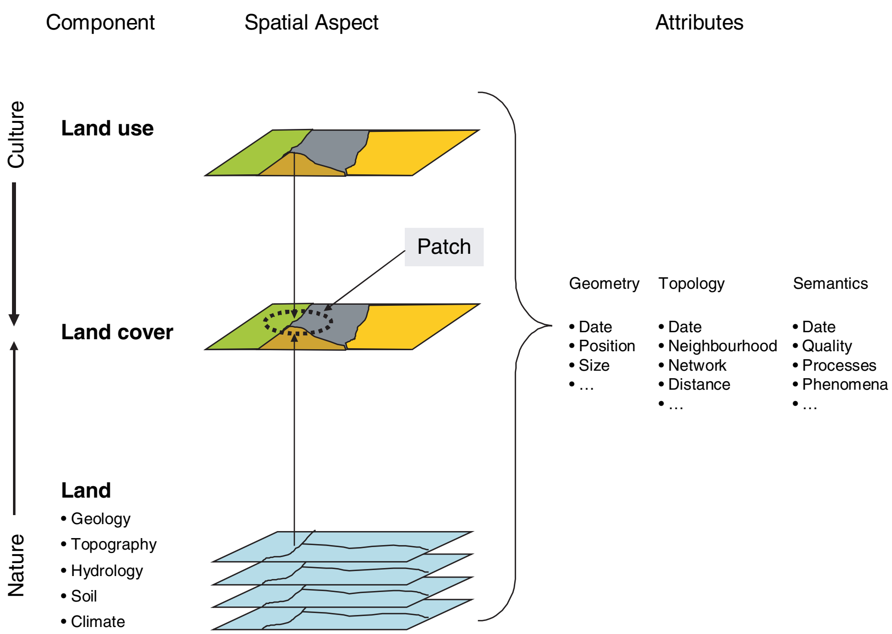

Fig. 1 Elements of the landscape system (adapted from Zebisch 2004)

FAO/UNEP in 1999 (Anonymous 1999), Land and Land Resources refer to a delineable area of the earth's terrestrial surface, encompassing all attributes of the biosphere immediately above or below this surface. They include those of the near-surface climate, the soil and terrain forms, the surface hydrology (including shallow lakes, rivers, marshes and swamps), the near-surface sedimentary layers and associated groundwater and geo-hydrological reserve, the plant and animal populations, the human settlement pattern and physical results of past and present human activity (terracing, water storage or drainage structures, roads, buildings, etc.). Land Use is characterized by the arrangements, activities and inputs by people to produce change or maintain a certain land cover type (Di Gregorio and Jansen 1998). Land Use defined in this way establishes a direct link between land cover and the actions of people in their environment whereas Land Cover is the observed (bio) physical cover on the earth's surface (Di Gregorio and Jansen 1998). All three components can be broken down into several subsystems to a more or less arbitrary level.

The actual state of the landscape is finally represented by its attributes. According to Burrough (1986) the system "Landscape" can be described by 3 different groups of attributes namely

- attributes describing the parcel or patch geometry, including size, position and others
- attributes indicating the topology of a patch meaning its neighbouring situation, the density or distance to other similar patches
- attributes representing the quality, potential as well as ongoing processes within the landscape (Semantics).

At this level all attributes mentioned so far are generalized meaning that they are not related to a specific discipline or land use type. The situation changes when information or purpose-oriented knowledge of a specific land cover type or a specific purpose is produced. This is done during the process of initializing attributes of the landscape under observation.

Besides the specific land use type which determines information needs, the scale at which the information is needed is the main factor for selecting the appropriate information. Scale normally refers to the spatial-temporal resolution in which landscapes are inventoried (scale of observation) or processes within the system are observed (process scale). Traditional land use planning procedures are normally done at a meso-scale (minimum unit 1 ha) whereas forest planning resolution might even be at micro-scale (minimum unit 1 m^2).

Depending on specific user requirements or the land use system, huge sets of information in form of indicators were developed. A wide set of attributes describing geometry and topology of landscape patches have been proposed by McGarigal and Marks (1995). They developed the landscape metrics statistics, a set of indicators describing the spatial character of a specific landscape. "Patchiness" or "Fragmentation" for example are important attributes of landscapes providing valuable information about the degree of disturbances caused by human activities or other factors. In less disturbed landscapes which still can be found in some parts of Southern Africa, the spatial properties of the landscape mosaic are of particular interest when designing green landscapes. Here the design of "ecological networks" which allow a sustainable use of landscape without severe losses in biodiversity is a major challenge in landscape design (Samways 2007). Forest science has developed a similar but considerably more complex set of indicators and tools to describe and analyse forest structures in all its 3 dimensions (e.g. Gadow 1993; Gadow and Füldner 1992, Gadow and Stübner 1994; Zenner, 1998).

By far the biggest number of attributes can be found in the third group. Depending on the purpose, qualitative properties of landscapes, parcels or patches can be described by hundreds of different indicators, each developed for specific land cover and land use types. These attributes are subject to the complex land evaluation and assessment procedures which form a major part of any land use planning concept. Furthermore this group includes attributes describing the processes which may change the landscape. Current or expected changes in landscapes form some of the most important planning information and its prediction, assessment and interpretation is one of the most challenging and important tasks within the planning procedure. In order to meet the challenge and to understand dynamics in landscapes a "Land-change science" has emerged as a foundational element of global environment change and sustainability science in the last few years (Rindfuss et al. 2004).

In complex environments such as green landscapes with its biophysical, social and economic levels, the question which information is relevant in the planning context is often controversial. This information which is related to the landscape attributes mentioned above should be fit to help in quantitative and qualitative evaluation of the landscape's current state. Furthermore costs for its production should be according to the benefits expected from its use.

2.1 Data Capture

In order to capture the data base for "Green Landscape Design" a host of different approaches and methods were developed in recent times. A large body of publications details the actual state of knowledge in very different disciplines including natural and social sciences. It includes besides others direct measurements or field surveys, remote sensing with digital image processing techniques as a tool to collect data about the land cover on huge areas and regionalisation including sophisticated spatial interpolation techniques which are useful in estimating landscape attributes for any point in the area of interest.

3 Remote Sensing

Detecting attributes of remote objects without being in physical contact with it has become an indispensable tool in monitoring landscapes. This is due to two different technical developments which form integrated parts of a modern Remote Sensing System namely; a sophisticated sensor component and advanced image processing methods. Both segments have gone through significant changes in the last decade allowing for more detailed and more accurate data capture and a much broader thematic depth.

The sensor segment in particular has grown significantly over the last decades. Today's state of the art is characterised by two main developments: the passive remote sensors and the active sensors. Passive sensors depend on existing electromagnetic radiation – mostly the sun light – which is reflected or emitted by the objects of interest. This group includes optical sensors like the traditional aerial photograph and multispectral digital cameras which normally cover the visible part of the electromagnetic spectra and some parts of the infrared spectral band. More recently imaging spectrometry has gained growing attention in remote sensing. Hyperspectral imaging is the simultaneous acquisition of images in many narrow, contiguous, spectral bands. Each pixel in the remotely acquired scene has an associated spectrum similar to the spectra of the material/mineral obtained in the laboratory. These capabilities open the door to a more qualitative assessment on the different objects shown on the images.

Active sensors produce their own electromagnetic radiation to collect data about remote objects. RADAR[1] sensors applied in remote sensing of the earth surface use microwaves in wavelengths between 2, 4 cm and 30 cm to illuminate the area under investigation while receiving the reflectance coming back. LIDAR[2]-systems apply a similar principle like the radar sensors. The systems actively emits laser light which is scattered back by the surface illuminated. The time for the light to travel out to the target and back to the instrument is used to determine the range to the target.

Analysing remotely sensed image data normally includes methods of image interpretation or pattern recognition for deriving more qualitative data on the one hand and methods for measuring objects shown on the images using photogrammetry. The most commonly used method in image interpretation is to identify spectral and textural similarities in the multidimensional spectral space and linking the findings to different land cover categories. Standard classification procedures provide more quantitative data allowing detecting geometric and topological attributes of the land cover classes.

The forestry sector and forest science in particular have developed extensive experience in initialising forest attributes using remote sensing techniques. After mapping the Brazilian Forest Cover (Carneiro 1991) in the 1970s the modern methods where employed all around the globe. Studies using Landsat imagery indicated the successful discrimination of age classes and tree species of coniferous and deciduous forest areas in southern Germany. The field verification yielded a final mapping accuracy of between 70% and 85% (at the 95% confidence limit) for the six distinct classes (Keil 1988). Schardt (1990) even reported a sensational accuracy ranging from 92% to 100% when looking at the discernment of a variety of tree species in South Germany. In Scandinavia, several studies have centred on the evaluation of

[1] RADAR: RAdiowave Detection And Ranging

[2] LIDAR: LIght Detection And Ranging

land use and forest areas (Jaakkola et al. 1988; Gjertsen et al. 1999). The digital analysis of Landsat TM as well of SPOT imagery revealed, that coniferous and deciduous forests in Sweden could be identified with an overall accuracy of 90%. However, the effort to discriminate different tree species as well as to determine the structure of age classes turned out to be less rewarding (only 64%). Several applications at this level were reported for South African forests. The first time South Africa came into action, was in 1974, when a set of satellite images covering the entire country, was acquired and analysed (Malan 1974). One of the first forest maps for South Africa derived from satellite image was produced in 1988 (Malan et al. 1980; Zel 1988). Wannenburgh and Mabena (2002) for example undertook a national indigenous forest inventory for South Africa based on Landsat ETM data using a hybrid approach based on semi-automatic classification and visual interpretation. From other parts of the globe similar investigations are reported showing that 10 to 12 land use classes can be derived from multispectral satellite images whereas 2 to 3 forest types or age classes depending on the general forest environment can be mapped with sufficient accuracy (Hildebrandt 1996; Lillesand 2004).

With high resolution satellite images showing a much more heterogeneous picture of the landscape the need for different image classification procedures has grown. Basically two different approaches for image classification have been developed so far: a pixel based classification and a contextual so-called object based classification method. Whereas the first approach classifies each image pixel within a multispectral image space employing at least 2 spectral bands, the second approach uses image segmentation before the classification itself is done. The segmentation process analyses the images by looking into spatially linked groups of pixels which have a similar shape or texture. These pixels are then interpreted as belonging to a parcel of land with the same land cover. In the second phase the segments are classified and the thematic map is produced. The combined approach is in particular beneficiary when classifying high-resolution images. Many investigations reveal that the combined approach allows for more accurate classification (e.g. Desclée et al. 2006; Brooks et al. 2006; Shiba and Itaya 2006; Yan et al. 2006; Kosaka et al. 2005; Hese and Schmullius 2004b). When classifying Korean forest landscapes from Ikonos images Hyun-Kook (2002) found a significant improvement of the overall accuracy from 56,6% to 65,7%. Other authors claim similar results (e.g. Meinel et al. 2001).

More recently efforts have been made to support inventorying landscapes with automatic object recognition methods. Stolz (1998) developed a knowledge-based classification system for automated land cover classification. The system uses Fuzzy-Logic approaches in identifying typical landscape objects and produces highly accurate thematic maps for the area of interest. Other approaches apply semantic network modelling or neural networks (e.g. Kunz et al. 1998; Schilling et al. 1994). Automatic recognition of trees and other vegetation objects on images has not yet reached a practical state applicable to all forest types (Kätsch 2001). Nevertheless some investigations have been done in recognising tree species. Pinz (1988) developed a system for automatic detection and recognition of tree species in Austria. Brandtberg (1997) applied image segmentation processes for recognising different tree species on highest resolution photographs. He used criteria linked to the specific branch system of a tree species. More recently Eriksson (2004) developed an automatic species classification system based on individually segmented tree crowns in high resolution aerial imagery. Detecting two conifer species (Spruce, Scots Pine) and two deciduous species (Birch, Aspen) the system yielded an overall accuracy of about 77%. Korpela (2004) found similar results for Scots Pine and Silver Birch on multi-scale aerial images.

Remote Sensing techniques have intensively been used in initialising typical attributes of the traditional forest metrics. Direct measurements of geometric stand properties like mean stand heights or dominant stand heights can be performed by using digital photogrammetric methods. Modern digital photogrammetric plotters support the automatic extraction of surface models which can be used to estimate stand heights with acceptable accuracy. Intensive research in this field, done in central European Forests showed that an absolute overall accuracy of $+1.4\,\mathrm{m}$ to $-1.0\,\mathrm{m}$ can be achieved on large scale multispectral aerial photographs (e.g. Kätsch and Stöcker 2000). Despite encouraging results when applying this technique for mean stand heights it should be mentioned that heights of single trees can often not be derived from digital surface models due to matching errors of the extraction software. On high resolution satellite imagery similar measurements are possible but may result in much less accurate results. For forest plantations in South Africa Kätsch (2000) found average accuracy of $\pm 4\,\mathrm{m}$ to 5 m when using "across track" stereo models derived from the Indian remote sensing system IRS 1c/1d. This low accuracy does not allow using photogrammetric stand heights for forest management decisions.

In order to further automate single tree measurements on aerial photographs several different approaches to delineate tree crowns from the images have been proposed. Pollock (1996) developed a template based matching approach which was refined by other authors (e.g. Larsen and Rudemo 1997; Olofsson 2002). Korpela (2004) extended this to a 3D-template matching system. Other methods include region based crown segmentation (e.g. Pinz 1989; Pitkänen 2001; Culvenor 2002; Pouliot et al. 2002; Eriksson 2003), contour oriented approaches (e.g. Gougeon 1995; Brandtberg and Walter 1998; Brandtberg and Warner 2006) as well as probabilistic methods (e.g. Perrin et al. 2006). According to Eriksson et al. (2006) none of the methods available today is robust enough to handle all types of forests. In a comparative study testing three different existing delineation methods on different forest types he concluded that in planted, more open forests a Marked Point Process (MPP) in which trees are represented as geometric objects produces the best results whereas dense, closed forests hamper the extraction procedure significantly. In order to apply the best delineation methods for each forest type he recommends a pre-segmentation of the image.

Besides the photogrammetric techniques used in forest inventory, the growing potential of LIDAR has to be mentioned. Based on distance measurements between a plane and the surface, object heights can generally be derived with a very high accuracy. Forests and any kind of vegetation however, with irregular density and height variations cause specific problems when using this technique. Research from all parts of the globe shows the potential of this technology (e.g. Nilsson 1996; Koch and Friedländer 2000; Amano et al. 2001; Persson et al. 2002; Weinacker et al. 2002; Katzenbeisser and Kurz 2004; Morsdorf et al. 2004; Blaschke et al. 2004; Koukoulas and Blackburn 2005; Anderson et al. 2005; Rowell et al. 2006; Lee 2006). Reports regarding the accuracy of stand and tree-height measurements vary depending on the scanning technique and the forest conditions. Nilsson (1996) reported an underestimation of the stand height in Scotts Pine stands by 2,1 m–3,7 m. Hyyppä et al. (2000) assessed the accuracy of tree heights derived from LIDAR-data in a Finish test site. The authors calculated a standard error of about 1m in comparison to terrestrial height measurements. Under more complex forest conditions Funahashi et al. (2001) recorded errors higher than 50 cm depending on the vertical orientation of the tree crowns. Holmgren et al. (2003) developed regression models for calculating the basal area weighted mean height based on LIDAR data. The models allow estimating the mean height with an accuracy of about 10%–11% of the average height.

Results of single tree extraction from forest stand surface models using the "Watershed algorithm" (Straub 2004) reveals that about 80% to 85% of the visually counted trees were identified and only 50% of the trees known for its position in the stand were detected (Vögtle et al. 2004). Maltamo et al. (2004) employed high-density laser scans in enumerating single trees in a boreal nature reserve. 80% of the dominant trees were detected and height measurements on 29 Scots Pines yield an overall accuracy of about ±50 cm. Results for structurally heterogeneous Spruce forest presented by Solberg et al. (2006) revealed that 93% of the dominant trees were detected whereas only 19% of the suppressed trees could be counted on the dataset. Research recently carried out in South African Eucalypt stands leads to the conclusion that this promising technique is too costly at this stage (Hattingh 2004).

The potential of using ground based LIDAR-technology in capturing complete forest stands and landscapes for mensuration and visualisation purposes is currently under investigation in several parts of the globe (e.g. Thies et al. 2003; Aschhoff et al. 2004; Haala et al. 2004; Culvenor 2005). The high resolution at cm-scale and fairly automated data analysis procedures make the modern technique to a promising tool in assessing forest stand attributes.

The step towards the assessment of qualitative biophysical or biochemical attributes of landscape elements was taken by using imaging spectrometry based on complex airborne or space-borne sensor systems (e.g. Niemann et al. 1999; Udelhoven et al. 2000; Law et al. 2001; Goodenough et al. 2002; Schlerf et al. 2002; Sampson et al. 2003). It combines the advantages of the better perspective when looking from above with a deep view into the whole range of reflectance spectra of the objects shown. The expectations of this technical concept are, on the one hand, that it will make it easier to classify different objects on an image with much higher reliability. On the other hand, generating data on the current physiological or bio-physical and biochemical status of living vegetation might become possible. Nevertheless, the improved perspective along with a complete coverage of the earth surface and the objects on it is associated with two severe problems which hamper easy analysis of the hyper-spectral data. First, it is the quality of the atmosphere which is more or less pervious to electromagnetic radiation. A more humid, dusty atmosphere will change the spectra reflected from the earth's surface in a different manner than a drier, clear one. Secondly, the spatial resolution of imaging spectrometers mounted on aircraft or satellites is limited. Even if a high-resolution image with pixel size in the sub-decimetre range is available, the reflectance which is detected by the sensor will most probably come from very different objects. Leaves of trees, crops, weeds and bare soil will be registered in one pixel, showing a more or less undefined mixed pixel.

Meanwhile some of the problems mentioned above have however been solved or minimised. With additional terrestrial reference spectroscopy during the flight campaign, the process of radiometric correction and end-member analysis can be improved significantly. In particular when mapping minerals, which normally have a very distinct spectral reflectance curve, methods of atmospheric correction, noise reduction and spectral un-mixing have proven to be generally successful in detecting and recognising these minerals automatically. Furthermore, several results published on mapping living vegetation using traditional cluster-analytical approaches show that the thematic depth or resolution of classifications can be significantly improved when using hyper-spectral data (e.g. Martin et al. 1998; Sandmeier and Deering 1999; Darvishsefat et al. 2002). Whereas multi-spectral images can be classified into 10–12 categories (Forest, Agricultural Crops, Water...) when using supervised classification (Schmitt-Fürntratt 1991), a hyper-spectral dataset can basically be classified into 25 or more categories, allowing a much more detailed insight into the land use pattern. For this reason hyper-spectral remote sensing is the only available approach to get information on biodiversity in natural vegetation types typical for South African forest landscapes.

A more difficult situation must be reported when talking about evaluating the biophysical and biochemical properties of living vegetation with this modern technology. Beside the basic difficulties with atmosphere and mixed pixels, the leaves of a tree vary considerably in their spectral reflectance properties. Even under very clearly defined conditions in a laboratory, spectra will vary depending on the sun's angle, the leaf's position and of course, its current physiological status (e.g. Horn 2005). The scientific community is currently arguing about the right approach on how to solve these problems. Under discussion are two different approaches: the one is a more empiric one, the other one more deductive.

The classical empirical approach, which was in some parts developed for analysing multispectral images, uses indices or ratios from specific bands of the spectra which were identified as being of particular importance for any biophysical or biochemical property of a plant. Two chlorophyll indices called the Pigment Specific Simple Ratio (PSSRa) and the Pigment Specific Normalised Difference (PSNDa), as proposed by Blackburn (1998), should be mentioned for example. The PSSRa is a simple ratio of reflectance at two optimal wavelengths, in this case 810.4 nm and 676.0 nm for chlorophyll a $(\mathrm{PSSRa} = \mathrm{R}_{810.4}/\mathrm{R}_{676.0})$. Another method is proposed by Chapelle et al. (1992). The Ratio Analysis of Reflectance Spectra (RARS) uses an empirically derived mean "reference" spectrum from a chlorophyll-saturated plant in order to divide each hyper-spectral pixel spectra of the image to highlight specific absorption features of chlorophyll a, b and major carotenoid pigments. McNairn et al. (2001) applied these methods to several corn and bean fields in Canada.

The second approach, which is in particular promising for the retrieval of biophysical parameters of vegetation, uses radiative transfer model inversion. These models provide a theoretical collection of typical spectra for different crops, plants and land covers, depending on their properties. Furthermore, two factors mentioned above, namely the sun angle and the view angle, which changes the spectra significantly, can be included. By inverting the model, biochemical or biophysical properties of vegetation can be predicted. The development of reflection models applicable to any forest situation is currently investigated at many institutions (e.g. Chen and Leblanc 1997; Shabanov et al. 2000). Some models have been successfully tested for forestry purposes. Atzberger (2000, 2003, 2005) used inverted reflectance models to derive canopy variables for European conifer stands. A model to derive biophysical properties of landscapes with different land uses based on hyperspectral data and additional land use information was proposed by Dorigo et al. (2005). They estimated Chlorophyll content and the leaf area index (LAI) for different cover types which are important attributes in assessing agricultural crops and forest trees.

The analysis of hyperspectral imagery will form a major part of future research in the field of remote sensing of vegetation as it forms the only promising technology which may be of help when investigating processes in living vegetation. If this technology becomes a feasible alternative to terrestrial approaches, it holds great potential in the agricultural sector, forestry and global environmental assessment.

First investigations into the potential of active radar remote sensing in mapping landscape led to more disappointing results (e.g. Keÿler 1986). Many more recent investigations using more advanced sensor technology however reveal that this technology allows collecting quantitative as well as qualitative data about the landscape and about forests in particular (e.g. Imhoff 1995; Strozzi et al. 1998, 2000; Horrell and Inggs 2000; Mougin et al. 1999; Schmullius et al. 1999, Schmullius and Wagner 2000; Hallikainen et al. 2001; Fransson et al. 2000). This includes information about forest structure, species composition as well as timber volume or biomass estimates. Many of the authors mentioned above applied Interferometry which uses signals from two radar sensors taken at the same time. This technology allows modelling of 3-dimensional structures in the landscape. In the more humid tropical regions around the globe radar remote sensing provides the only approach to monitor landscapes and forests continuously as this technology is fairly insensitive to closed cloud cover (e.g. Oliver 1998; Leysen et al. 1998).

The wide variety of remote sensors available covering a wide range of technical capabilities suggests a multi-sensor approach when initialising attributes for green landscapes. Depending on specific requirements and budget restrictions given in most cases a multi-stage and multiphase approach is recommended. Systems such as these can cover information needs at the micro-scale with terrestrial LIDAR applications which would feed their information into the meso-scale level using airborne and high resolution space borne imagery and so on. A similar system was proposed by Hirata et al. (2004) allowing an efficient and economical viable landscape inventory.

Understanding the dynamics and the processes in forming or changing landscapes is crucial for any planning approach. Since the early days of Remote Sensing the analysis of time series has significantly contributed to raising awareness about changes in the global environment. Changes in this sense refer to either changes in spectral signatures reflected from the objects on the earth's surface; which might be useful in estimating qualitative as well as quantitative changes or to changes in geometry measured within a set of two 3D surface models. Both approaches have been investigated quite intensively for different purposes leading to a set of well established methods in image based change detection.

Detection of changes in spectral reflectance values between a set of digital images taken at two different points in time is normally done using indices showing whether a change happened or not. A large number of indices were proposed to date. They can be classified into different categories: indices based on pixel values (punctual comparison), textural parameter based indices (contextual comparison) and information measurement based indices (Shannon and Weaver 1963). The first group applies a pixel value comparison by computing simple image differences. These calculations can be done between the reflectance based greyscale values or any derived image data like the NDVI[3] or similar. This method normally requires a proper co-registration of the two images and should include a spectral calibration of the image pairs. Textural indices describe the variation or diversity of pixel values within an "observation-window". Functions like variance, contrast or homogeneity have been widely used in these approaches. As all indices developed thus far, they have their advantages and disadvantages in a specific change detection problem. Le Hégarat-Mascle and Seltz (2004) proposed to use a set of indices in a fusion procedure. With the same aim in mind Wiemker et al. (1997) developed a change detection strategy which integrates various concepts in order to make change detection robust against varying recording conditions. It combines textural parameters from local neighbourhoods, principal component analysis and an unsupervised classification using cluster analysis based on Bayesian Rules which minimizes the probability of error. Hame et al. (1998) presented an unsupervised method (AutoChange) for change detection and identification of forest areas and Hese and Schmullius (2004a) applied a classification approach based on neural network classification for detecting changes in forest cover in the Siberian forests. Altogether automatic change detection based on direct comparison of pixel or arrays of the images remains a difficult process leading to overall accuracies of about 80%–90% in forested areas.

The automatic determination of changes in the dimensions and structure of forest stands using image time series is limited to highest resolution aerial photographs or LIDAR data. Richter (2001) estimated height growth in Spruce stands by calculating the difference between the two stand surface-models derived from high resolution stereo images. In average he found good relations between the real height growth of the forest stand, but due to positional errors and artefacts derived during the automatic terrain extraction procedure calculations of height growth of single

[3] NDVI: Normalized Difference Vegetation Index.

trees was not successful. Higher accuracy for single tree observations was achieved by using LIDAR (Anderson et al. 2005; St-Onge and Vepakomma 2005; Yu et al. 2004). Yu et al. (2006) detected height growth of single spruce and pine trees with a satisfactory relation to the field measurements ($R^2 = 0.68$)

3.1 Estimating Attributes by Regionalization

Regionalization techniques include different methods to transform spatial information from one scale to the other (Kleeberg and Cenus 1992). The most important concepts available include, scale change (aggregation/segregation), translocation and transformation using a transformation function.

Scale change by aggregating information is a well known and often applied technique in rescaling of information. Forest inventory concepts for example are mostly based on these methods by combining data taken on sample plots to a higher level. This "downscaling" may be sufficient in most cases, some attributes however require a specific approach. In particular attributes describing forest structure cannot be aggregated to an average structure with a simple arithmetic mean calculation. Similar problems have been reported from the social sciences which deal with individual behaviour as actors in a changing landscape (Rindfuss et al. 2004).

Translocation is often used in initializing attributes of land resources like soil characteristics, hydrological issues or the climate. Terrestrial data representing these land attributes are mostly measured in regular or randomly distributed grids. Site data including soil information, climate data as well as data taken in social surveys for instance are therefore available in a more or less coarse network making it difficult to apply a full spatial coverage of fully initialized landscape attributes. Modern geo-statistics provides a powerful toolset for estimating a complete surface formed by *x, y* and *z*-axis. In spatially distributed attributes x,y describe the location while z is the elevation within the 3-dimensional space representing variable of interest. One of the most commonly used methods uses a triangulation procedure which aims at creating a network of triangles which represent the surface with a set of planar triangular facets. These planar triangles can be refined by using fifth-order polynomial surface for each triangular area using the nodes of the triangles. Elevations from surrounding triangles are also used to compute the slope and change of slope in different directions around each node. Other methods apply different smoothing procedures like cubic spline adaptation (minimum curvature) or polynomials. This method calculates the best-fit surface by minimizing the sum of the squared deviations between the input values and the calculated surface.

One of the more complex methods is based on the assumption that two points in the space under observation become more similar to each other the nearer they are located. This spatial autocorrelation has to be described in a semi-variogram used in the specific interpolation procedure which is called "Kriging" (Matheron 1963; Röttig 1996; Wackernagel 2003). The algorithm interpolates an elevation value for each output raster cell by calculating a weighted average of the *z*-values at nearby points. Closer points are weighted more heavily than more distant points. It analyses the statistical variation in values over different distances and in different directions to determine the shape and size of the point selection area, and the set of weighting factors that will produce the minimum error in the elevation estimate.

Translocation techniques have been used for different purposes covering a wide range of disciplines and applications. Many of the applications in the forest sector deal with regionalizing point data on soils, precipitation and other topographical parameters (e.g. Saborowski and Stock 1994; Stendahl 2001). Figure 2 shows a map produced from regionalized temperature data (Growing degree days) taken in a field survey.

Other investigations concentrate on the possibilities to translocate forest characteristics from sample plots into the forest area (e.g. Biondi et al. 1994; Mandallaz 2000). Nieschulze (2003) looked into the possibilities to estimate forest stand data from forest attributes measured on sample plots using different approaches of regionalizing point data. He applied 4 different approaches including the Kriging algorithm in mixed and uneven-aged forests in central Europe. Based on ancillary data taken from aerial photographs and some ground samples he was able to estimate the stand volume and other stand attributes which are not subject of spatial neighbourhood.

In the forestry domain "Nearest Neighbour" methods have often been used in regionalizing forest stand data from sample plots. Methods such as these allow predicting the forest characteristics at a specific location based on a number of closest sample plots. Moeur and Stage (1995) proposed the "Most Similar Neighbour (MSN)" technique which combines so-called design attributes taken on sample plots with indicator attributes gathered from remotely sensed data. Another method often applied uses the k-nearest neighbours for prediction (e.g. Kilkki and Päivinen 1987; Maltamo and Kangas 1998; Franco-Lopez et al. 2001; Muionen et al. 2001; Sironen et al. 2001; Holmström 2002; Katila and Tomppo 2002; Koukal et al. 2004). It has been employed in the Finnish National Forest Inventory since 1990 which

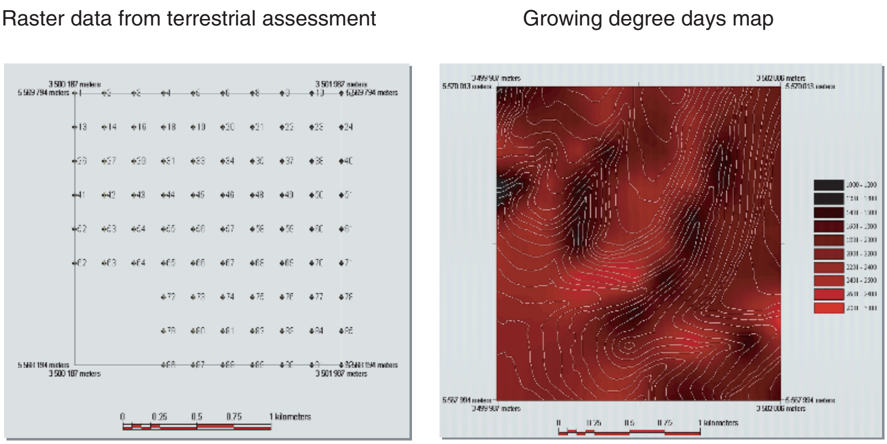

Fig. 2 Regionalization from site data using "minimum curvature interpolation"

integrates remotely sensed data with field data (Tomppo 1990, 1993, 1997; Tomppo and Katila 1991).

3.2 Initialising Phenomena and Processes as Landscape Attributes

Information about ongoing processes and phenomena, describe the dynamic element in landscapes (see Fig. 1). Changes in landscapes are basically the result of action and interaction of natural processes and human activities. Understanding these phenomena and processes is a crucial foundation for any planning concept in order to predict a future situation within the landscape.

For simulating the state of landscape attributes at a specific point in time many methods have been proposed. Traditionally the forestry sector developed a set of interesting simulation tools to predict the future situation of the forest. With the growing use of modern Geographic Information Systems (GIS) the whole methodological framework of computational modelling provided by modern Informatics gets more and more interesting in land use and forest planning.

Modelling of landscape systems with its elements; land use, land cover and land normally requires two key-components. First the landscape has to be represented in a spatial model which could be based on a cellular model. This model can be defined using spatial modelling techniques such as cellular automata, spatial diffusion models or Markov models (Berger 2001). The human decision making within the landscape is represented by the second component which can be defined as a multi-agent based model.

Cellular automata are defined as a n-dimensional space of discrete cells which interact with neighbouring cells depending on specific rules. Cellular automata have first been proposed by Stanislaw Ulam in 1940 and have later been refined by von Neumann. John Horton Conway's Game of Life presented in the 1970s was one of the first applications in artificial life programming and been successfully used in modelling biological systems since the 1990's. Cellular Automata are in particular suitable to simulate so-called self organised systems. Systems such as these are formed by evolution processes of complex structures which are mostly driven by variations of the system itself. Sprott et al. (2002) applied a 2-dimensional cellular automata with one adjustable parameter in modelling past developments of a typical landscape pattern in Wisconsin. Pommerening (2006) used a similar approach in modelling forest structures.

Agent systems consist of independent decision making entities which interact with their specific environment. The agents are driven by rules that translate both internal and external information into internal states, decisions, or actions. Agent-models are usually implemented as Multi-agent software systems. Multi-agent simulation systems were developed for very different purposes. A simple model predicts the activities of termites in collecting wooden sticks to a specific place. The system consists of a raster based simplified environment and a set of agents which represent the active termites. Other systems deal with modelling forest fires or the development of paths in more densely populated areas (Koch and Mandl 2003). More complex developments simulating changes in land use and land cover have been reported. Polhill (2001) developed the FEARLUS-Model for illuminating aspects of land change in Scotland. Aiming at predicting land use for the coming 50 years, the system models the interaction of land managers with their specific environment. A similar model was used in the Philippines (Huigen 2001). His model consists of a cellular model describing the environment which incorporates all biophysical attributes and models. The actors or agents are mainly farmers and loggers, who are usually the main actors in tropical forest land change. Berger (2001) developed a multi-agent system in the field of agro-ecological development in Chile. The model contains three basic functional types of agents which represent subsistence farmers, commercial farmers and non-agricultural landowners. More examples can be found in Parker et al. (2001) and Koch and Mandl (2003).

4 Examples from the Western Cape Province South Africa

The Western Cape is one of the nine provinces of South Africa and located at the south-western tip of Africa. It is positioned between $30^\circ\ 30'$–$34^\circ\ 45'$ South and $17^\circ\ 50' - 24^\circ\ 10'$ East and occupies an area of 129 370 km^2. It is bound by the Atlantic Ocean to the west and the Indian Ocean to the east with the L-shaped Cape Fold Mountains which separates the wetter coastal strip from a drier interior (Fig. 3).

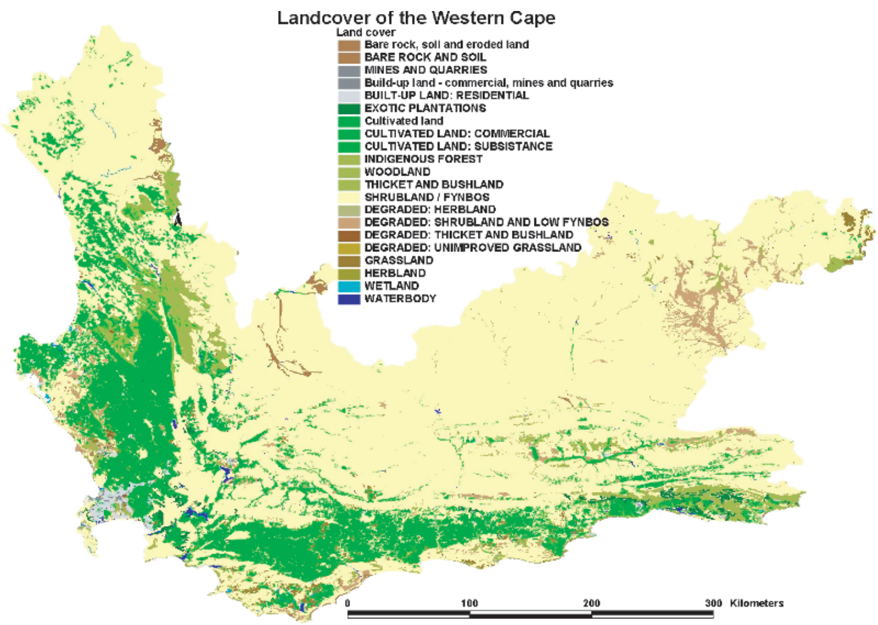

Fig. 3 Map of western cape land cover

The climate of the province differs from the rest of South Africa, most of it having a Mediterranean type climate influenced by strong coastal winds. Winter rainfall is prevalent in the south west with up to 25% of rain in summer months in the Peninsula. The summer distribution increases to the east and north east interior. The change in rainfall distribution to the east results in an area on the south coast where year-round precipitation occurs, supporting an area of indigenous forests. Precipitation varies from 100 mm/a–3000 mm/a. Coastal temperatures are less varied and milder, but temperatures range between the extremes of frost and very hot in the interior.

The physiography of the Western Cape is characterized by pronounced topographical variation with spectacular scenery. Two important land marks with international recognition is the Southern most tip of Africa, Cape Agulhas and Table Mountain. The Fold Belt consists of erosion-resistant, quartzitic sandstone underlain by softer shales, with a mantle of young silicaceous and calcareous sediments at the coast. Intrusions of the Cape Granite Suite form the undulating hills between the Atlantic Ocean and the Fold Belt. Alluvial deposits can be found in coastal areas such as the Cape Flats, Atlantis and Langebaan and in the Breede and Hex River Valleys. Soils resulting from the described underlying geology tend to be deeper at the coast, because of alluvial deposits and higher rainfall with weathering. The drier inland areas have thin or poorly formed soils, which is more clayey as a result of the weathering of underlying shales. Soils derived from colluvial material on slopes may include granite material and are generally richer in nutrients, red and yellow in colour and less acidic.

South Africa and especially the Western Cape is one of the biologically most diverse areas in the world with the following ecoregions: fynbos, forest, succulent and Nama Karoo, thicket, grassland, freshwater, estuarine and marine. The Cape Floristic Region (CFR), which is one of six floristic regions in the world, is mostly located in the Western Cape and comprises five large plant families, namely the daisy, iris, legume, protea and heather families. The CFR includes three major vegetation types, the predominating Fynbos shrubland, Renosterveld with grasslands and forest and thicket. More plant species are found in the Western Cape than in all of Europe.

Demographically the region is home to more than 4.5 million people, with 70% living in greater Cape Town. The primary economic sectors of the region are agriculture and fishing with a range of economic activities. Tourism is a growing activity with a resulting pressure on the landscape, by development of coastal and other natural areas. Studies indicate a net migration into the region mainly from the Eastern Cape which has a lack of employment opportunities and poverty, with the result of a growing informal settlement in the Western Cape region.

Due to the specific natural, cultural and economic situation in the country, landscape planning in general and inventorying landscapes in particular face problems which make it different from other parts of the globe. On the one hand side a booming economy leading to a growing demand for land along with a more or less uncontrollable spread of informal settlements causes unpredictable disturbances and fragmentation of the landscape. On the other hand the remarkable diversity in natural vegetation and rapid changes caused by natural disasters are challenges for any monitoring system able to keep information about landscape attributes up to date. Some examples mostly based on optical remote sensing techniques may give an impression about the situation.

Indigenous forests in the south eastern parts of the province give a good example on how difficult inventory concepts based on remote sensing techniques can be. Due to the enormous diversity standard image classification procedures often fail. Although the results can be significantly improved by using a hybrid approach incorporating spectral and textural image information this inventory approach remains difficult.

The following figure shows results achieved in an ongoing project in the Knysna forest (Fig. 4). The slide shows two thematic maps produced from multispectral images from the American Aster Satellite.

By using the three high resolution bands (GSD[4] 15 m) a set of 4 different classes can be found within the forest area. The second map was produced by classifying the image with the sample cluster analytical approach but with a fourth layer produced from filtering the near infrared spectral band with a WMMR-Med[5]-Filter algorithm. This algorithm is a nonlinear edge-enhancement filter that also suppresses image

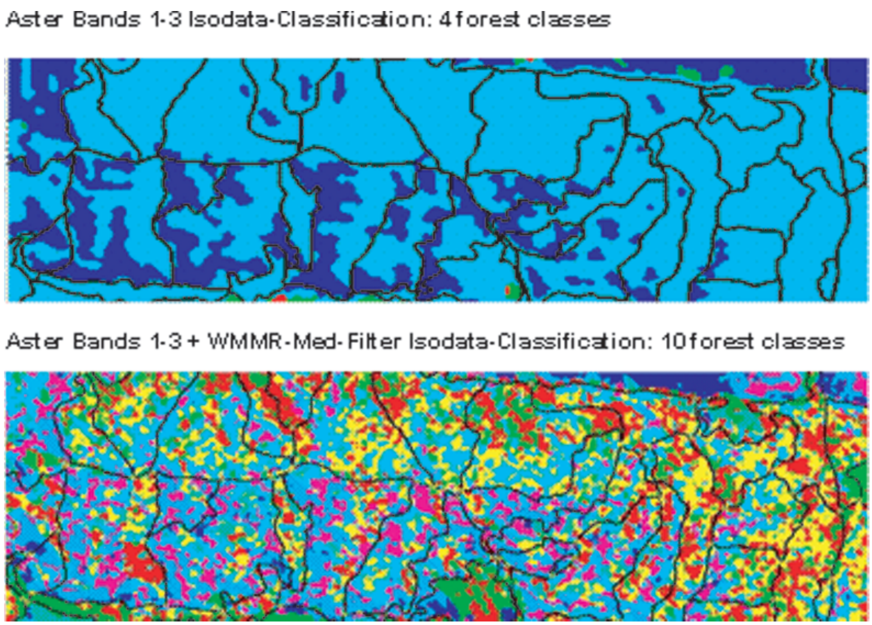

Fig. 4 Forest type classification in the Knysna forests using different image layers of the American Aster Satellite (Kätsch 2007)

[4] GSD: Ground Sample Distance (pixel size)

[5] WMMR-Med: Weighted Majority with Minimum Range – median (Anonymous 2006)

noise. It enhances changes in pattern or texture which resulted in a much more detailed classification of the forest. The classification fits well to the compartment boundaries which normally form management units related to forest types. Although the results were not verified rigorously, this example shows that modern digital image processing may allow a more detailed mapping and inventorying of forests. Nevertheless initializing forest parcel attributes in the complex uneven-aged indigenous South African forests using remote sensing techniques remain difficult and require further research.

Another issue causing severe land changes has to do with aggressive spread of alien vegetation. In the past alien plants, mainly Acacias from Australia were used to stabilise the sands of the Cape Flats and other coastal regions. This practice resulted in the uncontrolled spread of some Acacia species across a large area in the Western Cape endangering some indigenous and endemic vegetation. The spread of invasive alien plants follow disturbed and degraded areas. Some planted timber species also spread to adjacent vegetation. The natural vegetation is a fire climax vegetation and pioneering species rapidly invade the areas of slower regenerating natural vegetation. Figure 5 shows an example of using change detection to illustrate the dynamics of invasive plants in the Western Cape.

The map was produced using simple grey scale difference calculations on Landsat 5 satellite image time series (Altena 2000). The two colours represent an increase

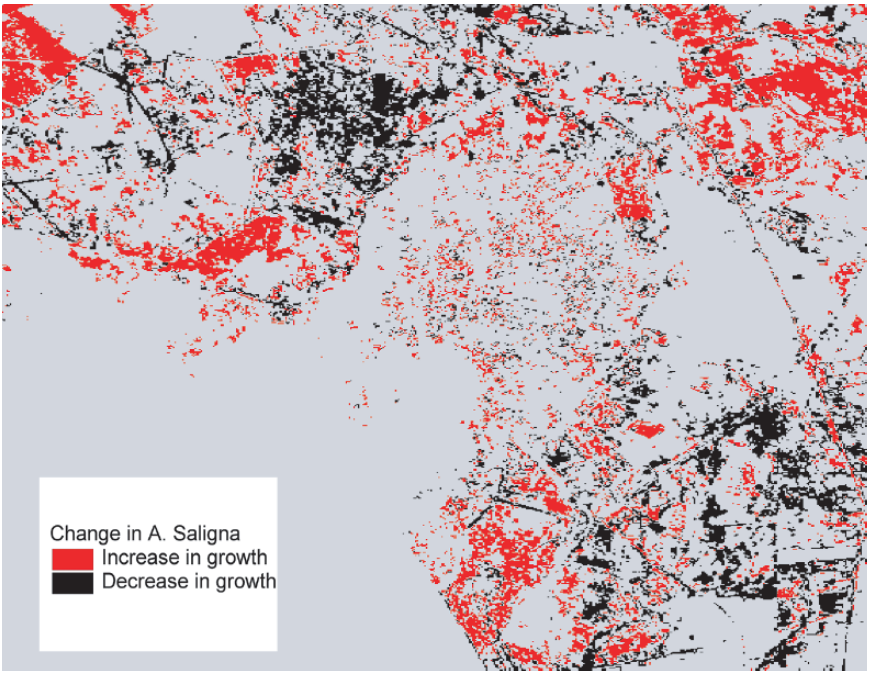

Fig. 5 Change of the Acacia saligna population in the period between 1993 and 1997 (Altena 2000)

and decrease in area of the invasive species (Acacia saligna). While fire and biological and mechanical control decreases the areas invaded by the alien species, regeneration from seed, normally after fire or other disturbance increases the area invaded.

Fire changes the landscape in both natural and man made landscapes. The indigenous fynbos vegetation is a fire climax vegetation which naturally burns in cycles of between 3 and 15 years. Forestry plantations are susceptible to fires having large above ground biomass and a litter layer which might burn in the dry summer season. Figure 6 shows an example of an area during a fire event, with burnt and unburned vegetation. In order to classify the area affected by a fire disaster, experiences in many other countries reveal that mainly the use of so-called ratio images is promising to identify more or less damaged areas (e.g. Brewer et al. 2005). Ratio images are artificial images calculated from the original images taken reflectance in several spectral bands. Two well known ratios are recommended in fire assessment. First the Normalized Burn Ratio (NBR) can be calculated combining the pixel taken in the near infrared spectral band (wavelength 0, 78 μm–0, 9 μm) and the mid-infrared band (wavelength 2, 09 μm–2, 35 μm). The other ratio known as Normalized Difference Vegetation Index (NDVI) combines the red colour pixel with the near infrared pixel. It gives an indication of the overall health status of the trees (Kätsch and Kunneke 2006).

Experiences from the United States and some South European countries show that the overall accuracy of a damage classification with 3–4 classes (no damage, slight damage, medium damage, severe damage) can be improved if some additional

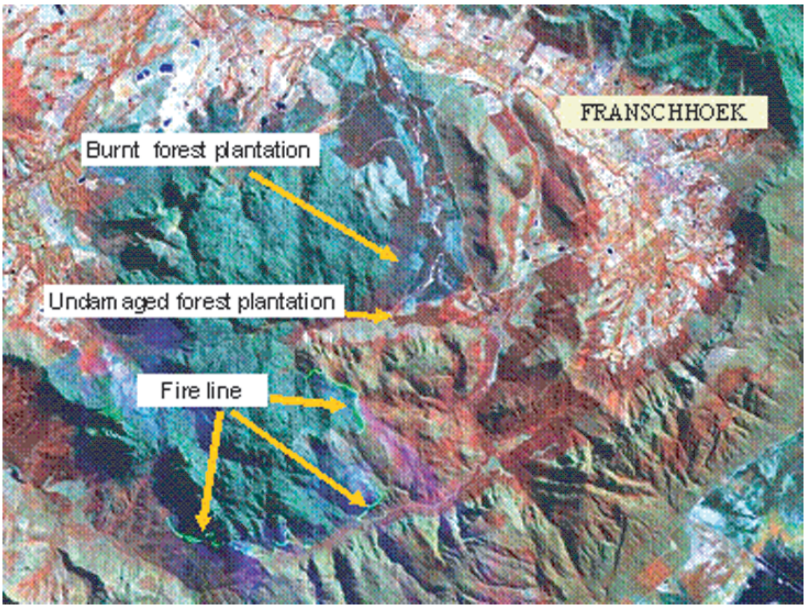

Fig. 6 Landsat TM image showing fire impact on landscape change

terrestrial assessment is carried out and linked to the image data using the Composite Burn Index (CBI) (e.g. Key and Benson 2003; Wagtendonk et al. 2004). With the help of correlation analysis and regression models a much more reliable prediction for the whole area of interest is possible.

In order to maintain South Africa's biodiversity and to minimize the risk from wild fires and other disasters forest companies in some parts of the country are currently planning forest landscapes as a network of natural vegetation, agricultural crops and forest plantations. Large patch sizes, good patch quality, reduced patch isolation and minimized patch contrast are the main principles in designing these landscapes. Initialising the attributes of landscapes such as these becomes a challenge for any inventory concept due to enormous patch heterogeneity, large number of edges and high interdependencies between land parcels.

5 Conclusions

The design or planned development of green landscapes relies on accurate information about the current and future state of the land parcels, its land cover and land use. In a multiple-path oriented planning approach incorporating a range of different land use options a wide variety of information is required, no matter what type of land cover or land use dominates the landscape. Information requirements however are not easy to define as land use planners, foresters and other stakeholders with very different background and knowledge need to learn to share a common perspective. This can be achieved by defining a "landscape model" on a systems-analytical basis. A commonly developed "Ontology" is the only way to agree upon a set of land parcel attributes to be measured during the planning process.

Besides the fact, that information needs are often controversial, problems with information quality need to be considered. Information fit for being used in a landscape planning procedure should be relevant in the planning context, should be delivered in time and is free of bias and other disturbing factors.

Data describing landscape attributes in all their complexity and diversity however are not easy to collect. Remote Sensing, a booming scientific discipline, has definitely opened up entirely new roads towards a comprehensive inventory of complete landscapes at the meso- as well as at the macro-scale. Although quite efficient and easy to use in most cases some limitations and typical characteristics hamper the use of the modern technology. Simple more organisational problems include image availability or quality at a specific point in time. Monitoring systems for example rely on periodic data capture which might fail if cloud cover prevents a clear view of the surface. More serious problems are linked to technical limitations. Limited geometric accuracy when looking from space often leads to spatial mismatch and changes in atmospheric conditions might cause a wrong thematic interpretation of the image. Factors such as these are responsible for the fact that image classification is not a robust method for data collection from images. Highest resolution images and sophisticated image classification methods have not changed this problem significantly. The integration of ancillary data from other sources, often recommended to improve classification, may be an option but should be done quite carefully. These data can be generalised or error prone bringing another source of uncertainty in the classification process.

Qualitative assessment of vegetation using remote sensors has not yet reached a full reliable, practical state. Air- and space born hyperspectral sensor systems with a high spectral resolution are available but data analysis and interpretation hold a big potential but require further research. Even under very clear conditions reflectance of electromagnetic radiation by plants is driven by many concurrent factors with synergetic or suppressive effects. Modern concepts for using hyperspectral imaging technology therefore suggest a combination of remotely sensed data with some data measured "in situ" on the ground.

Understanding of land change and predicting its consequences is a crucial issue. It is fundamental for any landscape plan or design to know how the landscape system will react and where possible uncertainties are. In particular under the conditions of a developing country with scarce resources and many more or less uncontrollable factors the need for a multiple path approach becomes obvious. In addition a timely adaptation to climate change requires a clear prediction of possible scenarios in land changes. Modelling and simulation techniques like cellular machines or agent based systems can help to predict and to understand the spatial situation at a specific point in time. The reliability of these methods however depends on how accurate the driving factors and their relationship to other landscape entities can be integrated. Many of these factors and their role in the system are still unknown making increased efforts towards a better and more complete understanding necessary.

The South African examples show how diverse and manifold problems in landscape planning can be. In this country with booming hotspots of an emerging economy on the one hand side and vast, fairly untouched areas on the other hand inventory concepts need to be flexible allowing for data capture at very different scales and with a wide thematic orientation.

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Part III
Numerical Analysis of Landscape Design

Spatial Optimisation – Computational Methods

Pete Bettinger and Young-Hwan Kim

1 Introduction

Forest Management planning can reduce the uncertainty and the likelihood of unexpected scenarios by anticipating future events in a systematic way. Management planning can also improve the probability that future developments are consistent with specified objectives. Many of these objectives, such as biodiversity, are affected by spatial structure in the landscape, i.e. the relative arrangement of patches. As a result, forest planning objectives today must include consideration to spatial relationships. In general there are two main approaches for incorporating spatial objectives into the planning process, the exogenous and the endogenous approaches (Öhman 2001; Kurttila 2001). In the exogenous approach, the optimisation does not include any spatial information but it takes into account predetermined spatial constraints. One example of a case where the exogenous approach could be useful is when set-aside areas are decided in advance, e.g. key habitats. In the endogenous approach, variables that in some way describe the spatial relationships between stands are processed by the solution algorithm, and the spatial layout is generated by the optimisation process (Kurttila 2001). Therefore, the endogenous optimisation approach can evaluate a huge number of different spatial alternatives and allow trade-off analysis between different objectives, which might be impossible with the exogenous approach (Hof and Bevers 1998).

Within the endogenous approach, the field of forest planning continues to evolve. First, there is continued interest in locating methods to optimally solve the spatial forest planning problems using traditional mathematical programming solution techniques (mixed integer programming, integer programming, etc.). The main emphasis here has been on the development of constraint structures to more efficiently handle spatial restrictions, and allow branch and bound or cutting plane algorithms to effectively solve planning problems, and if possible, solve them faster. Second, there has been an expansion on the use of heuristic techniques to solve spatial forest

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planning problems. In this area of work, researchers can effectively solve complex planning problems, but the concerns are both with the quality of the results, and with the speed in which the results can be obtained. The main reason for the exploration into heuristics is due to the combinatorial nature of today's forest management problems. Since spatial constraints typically require integer decision variables, and complex physical or biological effects models include non-linear components, one may find it difficult, if not impossible, to solve problems using traditional mathematical programming solution techniques. Therefore, heuristic programming techniques are considered viable spatial optimisation alternatives. The exogenous approach does not introduce any new requirements on the computational methods compared with traditional harvest scheduling. Linear programming, the mainstay of optimisation-based forest planning since the 60's, is in most cases a wholly functional tool.

This chapter introduces several computational methods that have been shown to be of value in the development of spatial forest plans. We begin with linear programming, which can handle spatial concerns, albeit only along exogenous lines. We then transition to integer and mixed integer programming, to illustrate how spatial constraints can be handled within traditional mathematical programming methods. Then we describe a suite of heuristics, which include simulated annealing, threshold accepting, the great deluge algorithm, tabu search, and genetic algorithms. Each of these search processes is unique and has value in developing complex forest management plans.

2 Traditional Mathematical Programming Solution Techniques

2.1 Linear Programming

Linear programming was the main planning methodology used for the long-term U.S. National Forest plans that were developed between 1970 and 1995. The solution technique is still used widely today for industrial and Provincial (Canada) forest planning, and is being used today in some of the recent U.S. National Forest plan revisions. The Simplex Method was first introduced in 1947 by George Dantzig, then a planner for the U.S. Air Force, as a way to efficiently solve a complex problem based on linear equations (Gass 2003). The term "linear programming" was suggested as a name for the solution technique he developed because linear equations are used, and because at that time, "programming" was synonymous with "planning" in the U.S. military.

Like all of the computational methods described in this chapter, linear programming requires an objective function, which helps one determine the quality of a proposed solution. Associated with an objective function is the notion that something is either being maximised or minimised. For example, assume that the intent of a forest management problem is to maximise revenue generated, and that there are three decisions (harvest stands in time periods 1, 2, and 3). Assume each hectare of stand 1 generates \$1485.16 of revenue in time period 1, \$1675.01 of revenue in time period 2, and \$1741.18 of revenue in time period 3. If we let the number of hectares of stand 1 that is harvested in period 1 be denoted by S1P1, in period 2 by S1P2, and in period 3 by S1P3, the objective function associated with stand 1 would be:

$$\text{maximise} \quad 1485.16 \quad \text{S1P1} + 1675.01 \quad \text{S1P2} + 1741.18 \quad \text{S1P3}$$

Some constraints are required to ensure that the number of units produced are consistent with the resources, labor, and productive capacity associated with the system. Otherwise the solution to this problem would be $\text{S1P1} = \infty$, $\text{S1P2} = \infty$, and $\text{S1P3} = \infty$.

Linear programming, in its most basic form, is guided by four main assumptions about the variables and the equations:

(1) Proportionality. Each decision variable has associated with it a coefficient. The contribution of each product to the objective function or a constraint is proportional to the number of units of each product produced.
(2) Additivity. Each decision variable in an objective function or a constraint contributes to the objective function or the constraint in a way that is independent of the other variables.
(3) Divisibility. The value assigned to each decision variable is assumed to be a continuous real number.
(4) Certainty. The coefficients associated with each decision variable are assumed to be known. Papps (2000) provides an example of stochastic linear programming, where this assumption is relaxed.

Accounting rows are used in linear programming models to sum values that may be needed for reporting purposes. Assume, for example, that one was interested in knowing how many units of land are being scheduled for harvest each time period. The following equation allows the summation of areas harvested from five stands during the first time period of a planning problem

$$S1P1 + S2P1 + S3P1 + S4P1 + S5P1 = AC1$$

The right-hand side must be adjusted so that all variables are in the left-hand side of each equation, so the equation becomes:

$$S1P1 + S2P1 + S3P1 + S4P1 + S5P1 - AC1 = 0$$

One could use the knowledge gained with information collected in an accounting row to constrain a management plan. For example, one could place a limit on the size of AC1.

Two types of constraints are generally used in linear programming models, resource and policy constraints. Resource constraints are employed to ensure that no more of some resource (budget, personnel, equipment, land, etc.) at one's disposal can be utilized. Policy constraints are employed to force a solution to adhere to organizational goals or regulatory restrictions. The resource constraint associated with the choices available to a single stand (assuming three planning periods) might be:

$$S1P1 + S1P2 + S1P3 \leq 12.3$$

Where 12.3 is the total area of the stand, and the choices relate to assigning a clearcut activity to the stand during one of the three periods. A policy constraint on the amount of area clearcut during a period might be:

$$AC1 \geq 300$$

The mathematical expression of an entire linear programming problem is called a *problem formulation*. The results one might obtain by solving the problem include the objective function value, decision variable values, reduced costs, slack, and dual prices. For example, when using LINDO (Lindo Systems, Inc. 2002) to solve a problem, one might obtain results that resemble these:

Objective	Function Value
1)	1589646.

Variable	Value	Reduced cost
S1P1	0.000000	256.010010
S1P2	0.000000	66.169991
S1P3	12.300000	0.000000
S2P1	0.000000	92.209984
S2P2	35.599998	0.000000
S2P3	10.000000	0.000000
....		
....		
AC1	399.799988	0.000000
AC2	345.399994	0.000000
AC3	376.299988	0.000000

One interprets the results to determine that the objective function value is $1,589,646, and that the optimal solution is to harvest 12.3 units of land of stand 1 during time period 3, and 35.6 units of land of stand 2 during time period 2, and so on. The reduced cost represents the amount that the coefficient for each decision variable in the objective function must increase before that choice becomes competitive enough, given the other choices available, to enter the solution. For example, the reduced cost for variable S1P1 is $256.01001. What this implies, is that the value of harvesting stand 1 during time period 1 must increase $256.02 before one unit of land of stand 1 is scheduled for harvest in time period 1. One also finds here the values of the variables that were introduced in the accounting rows. The variable

AC1, for instance, represents the total harvested area in time period 1 (399.8 units of land).

The remainder of the linear programming report generated by LINDO is shown below.

Row	Slack or surplus	Dual prices
2)	0.000000	1741.180054
3)	0.000000	1908.479980
4)	0.000000	2401.439941
5)	0.000000	2357.179932
6)	0.000000	1825.250000
7)	0.000000	1281.930054

The "row" column refers to each constraint (or accounting row). The slack associated with a constraint indicates how much of the right-hand side of each constraint is not being used in the optimal solution. If the slack associated with a constraint is 0, the constraint is considered binding, implying that the constraint has some influence on the optimal solution. The dual price associated with each constraint indicates how much the objective function value would increase if one more unit of the right-hand side of the constraint were available. For example, the dual price associated with constraint 2 is 1741.18 ($1,741.18). If one more unit of land of stand 1 were available (making the stand 13.3 units of land in size), the objective function would increase another $1,741.18.

2.2 Mixed Integer Programming

To control the spatial location of activities using linear programming, one would need to make sure that the entire area of each unit of land (or section of road, etc.) was treated the same way, during the same time period. Otherwise, since linear programming assumes that decision variables can be assigned values that are continuous numbers, some portion of a stand (or section of road, etc.) could be assigned one treatment, and some portion assigned another. Or, some portion could be assigned a treatment in one time period, and some portion assigned a treatment in another time period. In either case, we are not certain where the activity occurs, as in the case above, where stand 2 has 35.6 units of land assigned for treatment in time period 2, and 10 units of land assigned for treatment in time period 3.

The Big M method can be employed to overcome this limitation of linear programming. Here, new constraints are formulated, and new variables are created, to force the problem to assign all units of land to a single time period of treatment (or single management prescription). For example, in the equation below, when a single unit of land is assigned to stand 1, period 1, a second variable (Y1P1, a binary variable) is forced to become 1 (rather than 0).

$$S1P1 - 12.3 \ \mathrm{Y1P1} \leq 0$$

Since the new variable Y1P1 is defined as a binary number, the problem is now considered a mixed integer programming problem. The coefficient associated with Y1P1 is the "big M," and needs to be as large, or larger, than the largest value that could be assigned to S1P1 (12.3 units of land). More precisely, the equation can be re-formulated to ensure that when any unit of land is assigned to S1P1, Y1P1 is assigned the value 1.0, and the full size of stand 1 is assigned, forcibly, to S1P1:

$$\mathit{S1P1} - 12.3 \;\; \mathrm{Y1P1} = 0$$

The binary decision variables associated with each stand and each time period are then used to control the timing and placement of activities across the landscape. For example, if stand 1 and stand 2 were adjacent, and the unit restriction model of adjacency (Murray 1999) were used to prevent the scheduling of one stand during the same time period as the other, the following constraints could be used:

$$\mathrm{Y1P1} + \mathrm{Y2P1} \leq 1$$
$$\mathrm{Y1P2} + \mathrm{Y2P2} \leq 1$$
$$\mathrm{Y1P3} + \mathrm{Y2P3} \leq 1$$

With these constraints, one could develop a schedule that ensures that the entire area of each unit of land (or section of road, etc.) was treated the same way, during the same time period, and that two similar activities in adjacent stands are not scheduled during the same period of time.

An alternative formulation to this is to simply make the original decision variables (S1P1, for example) binary. The objective function would then be adjusted to indicate that the coefficients associated with the decision variables include not only the value of the proposed activity, but also the size of the area. From the linear programming problem we know that the intent of a forest manager is to maximise revenue generated, and that there are three decisions (harvest stands in time periods 1, 2, and 3) and that each land unit of stand 1 generates \$1485.16 of revenue in time period 1, \$1675.01 of revenue in time period 2, \$1741.18 of revenue in time period 3. Given the size of the stand, 12.3 units of land, the objective function associated with stand 1 would be:

$$\text{maximise} \quad 18267.47 \;\; \mathrm{S1P1} + 20602.62 \;\; \mathrm{S1P2} + 21416.51 \;\; \mathrm{S1P3}$$

Here, the values of S1P1, S1P2, and S1P3 are from the set {0,1} implying that you cut the whole stand or not at all. Accounting rows are also used here, yet they now include the size of each treated activity:

$$12.3\ \mathit{S1P1} + 45.6\ \mathit{S2P1} + 34.2\ \mathit{S3P1} + 10.3\ \mathit{S4P1} + 9.8\ \mathit{S5P1} - \mathit{AC1} = \mathit{0}$$

Note that the variable AC1 is not required to be integer, making the model a mixed integer model.

The resource constraints then recognize the binary nature of the decision variables:

$$S1P1 + S1P2 + S1P3 \leq 1$$

Here, the equation indicates that stand 1 can only be harvested once during the three periods of the analysis, or not at all. To control the spatial location of activities using this form of mixed integer programming, one would use equations similar to the ones we described earlier:

$$\begin{aligned} \mathrm{S1P1} + \mathrm{S2P1} &\leq 1 \\ \mathrm{S1P2} + \mathrm{S2P2} &\leq 1 \\ \mathrm{S1P3} + \mathrm{S2P3} &\leq 1 \end{aligned}$$

With these constraints, one could develop a schedule that ensures that the entire area of each unit of land (or section of road, etc.) was treated the same way, during the same time period, and that two similar activities in adjacent stands are not scheduled during the same period of time.

While the discussion above allows one to visualize how a linear model can be modified (with integer variables) to accommodate spatial constraints, the more typical, and current, methodology used to account for spatial arrangement of harvests is based on Murray (1999) and Murray and Weintraub (2002), where two basic types of adjacency restrictions are introduced. The first is the unit restriction model, which implies that when a management unit is scheduled for a clearcut, any adjacent management units (regardless of their size) are prohibited from being scheduled for clearcut harvest during the green-up period. This model was designed for situations where management units within the geographic information system database are close to the maximum clearcut size, which is rarely the case in practice. The formulation for unit restriction adjacency constraints is similar to what was described above:

$$X_{it} + X_{jt} \leq 1 \qquad \forall\, t,\ i,\ j\ \in \{N_i\} \tag{1}$$

Where:

i, j = a management unit
t = a time period
N_i = the set of all management units adjacent to unit i

The second model suggested by Murray (1999) is the area restriction model, which implies that adjacent management units can be scheduled for clearcut harvest during the green-up period. However, this can be done as long as the total size of the clearcut area does not exceed the maximum clearcut area. The formulation for the area restriction adjacency constraints is:

$$X_{it}A_i + \sum_{z \in N_i \cup S_i} X_{zt}A_z \leq \mathrm{MCA} \;\; \forall\, i,\ t \tag{2}$$

Where:

A_i = area of management unit i
MCA = maximum clearcut area
S_i = the set of all management units adjacent to those management units adjacent to management unit i
z = a management unit

The spatial constraints for the unit restriction model that we have introduced $(\mathrm{S1P1}+\mathrm{S2P1} \leq 1)$ are simple pairwise adjacency constraints, and perhaps not the most efficient for solving problems with mixed integer programming techniques. Several types of adjacency constraint formulations have been described previously for mixed integer and integer programming techniques (e.g., pairwise, Type I nondominated, new ordinary adjacency matrix). Type I nondominated constraints result in significantly faster processing time, new ordinary adjacency matrix constraints perform better in problems containing mainly immature forests, and pairwise constraints seem to perform better in forest planning problems containing overmature and old-growth forests (McDill and Braze 2000). McDill and Braze (2000) also indicate that the more mature the forest, the harder the problem is to solve using integer programming techniques. This is important when using the branch and bound algorithm generally associated with integer programming, because a smaller proportion of branches are trimmed during the search process, thus the size of the decision tree that is explored is larger than when using other types of adjacency constraints (McDill and Braze 2001).

The area restriction model is more difficult to employ, given that the relationship of management units to other management units and *their* neighbors must be recognized. However, progress is being made in addressing spatial optimisation with this model (McDill et al. 2002; Murray and Weintraub 2002; Goycoolea et al. 2005). The area restriction model typically requires more constraints than the unit restriction model, and the problem formulation will become increasingly complex as the average size of a management unit becomes smaller in relation to the maximum clearcut size. Some *a priori* limits on the potential to schedule adjacent management units could reduce the complexity, however, particularly in cases where the age of forests in adjacent management units is quite different (McDill et al. 2002). The appropriateness of the unit and area restriction models depends on the application intent and the underlying management objectives (Murray and Weintraub 2002).

3 Heuristic Solution Techniques

Linear and mixed integer programming are considered traditional mathematical programming methods that facilitate the development of the exact global optimum solution to a planning problem. The remaining computational processes are known as heuristics, and they rely on logic and rules to iteratively modify a suboptimal solution to a problem, with the goal of locating the best solution possible, usually a near-optimal solution. Many heuristic processes operate relatively quickly, as compared to traditional mathematical programming methods. Another advantage is that some of them are rather simple and require only modest programming skills, and some can also be found as share ware on the Internet. However, the location of appropriate heuristic parameters could slow the process. Pukkala and Heinonen (2006) describe a way to locate a set of appropriate parameter values that is independent of human intervention, and can lead to highly efficient solutions. One chief drawback of using heuristic algorithms is that they must be modified to the specific planning problem at hand. Each organization, it seems, is guided by a different set of objectives and constraints that make developing a universally-applicable method elusive. Another drawback is the quality of solution values. Bettinger et al. (2002) and Pukkala and Kurtilla (2005) have illustrated the quality of heuristic solutions when applied to forest management problems. If solution quality can be judged as being quite high when compared to mixed integer (exact) solutions, the speed at which a heuristic can solve a problem becomes the main factor in choosing these methods over mixed integer methods. What the future holds for these techniques is uncertain, as the efficiency of mathematical (exact) solvers continues to increase.

3.1 Monte Carlo Simulation

Monte Carlo (MC) simulation is essentially a random process that allows one to generate a forest plan rather quickly, compared to some other heuristics. Applications of Monte Carlo simulation in forest planning have been described in Clements et al. (1990), Nelson and Brodie (1990), Boston and Bettinger (1999), and Bettinger et al. (2002). A generic Monte Carlo simulation process can be described as:

Step 1. Select a random management unit and harvest period, and schedule the treatment.
Step 2. Determine the number of management units not scheduled.
Step 3. If more feasible choices are available, return to step 1.
Step 4. Calculate the objective function value, and save the solution.
Step 5. If more solutions are desired, return to the empty set and return to step 1.
Step 6. Report the best solution.

Enhancements to the process include (a) an assessment of spatial harvesting constraints (during step 1) to determine whether a particular randomly selected choice is feasible, and (b) an assessment of spatial harvesting constraints (during step 3) to determine how many feasible choices remain. In addition, the latter of these two enhancements can be used to inform the model about the set of potential choices, and bias the selection of choices based on the value of the choice being made.

3.2 Simulated Annealing

Simulated annealing was introduced as a search process by Kirkpatrick et al. (1983), based on the work of Metropolis et al. (1953). As a search process, simulated annealing is analogous to physical systems that may be in thermal equilibrium at many different temperatures, and it seeks to determine the low-temperature state of a system under a number of different scenarios. The process of annealing simply refers to the cooling of a material that was heated well above its known melting point. The annealing of metals, for example, is ideally done in such a way that defects in the final product are minimised. The probability that each state of a system occurs at thermal equilibrium, bounded between 0 and 1 is known as the Boltzmann-Gibbs distribution, or

$$P(x) = \left(\frac{e^{-E(x)\,/\,k_B T}}{Tr(e^{-E(x)\,/\,k_B T})} \right) \tag{3}$$

Here, k_b is Boltzmann's constant, T is the temperature of the annealing process, x is a given state of the system, *Tr* is the sum over all possible configurations of the particles in the system, and $E(x)$ is an energy function. Transitions from state to state are estimated by assuming that, on average, the probability of changing from state x_1 to state x_2 is the same as changing from x_2 to x_1.

$$P(x_1 \rightarrow x_2) = e^{-\Delta E\,/k_B T} \tag{4}$$

When the transition is to a lower state (i.e., cooling), the probability of the transition is 1. However, not every change in a solution improves the solution quality. When a change to a solution does not improve on the previous solution, the decision is made based on drawing a random number from a uniform distribution in the interval (0,1), and comparing it to the result of:

$$P\,(\Delta E)\, = \,\exp\left(\frac{-\Delta E}{k_B T}\right) \tag{5}$$

Here, ΔE is the difference between the current and previous solution values. If the drawn random number is less than $P(\Delta E)$, the suggested change to the solution is acceptable, even though the quality of the objective function value will decrease.

Applications of simulated annealing to forest management and planning have been described in Baskent and Jordan (2002), Bettinger et al. (2002), Boston and Bettinger (1999), Liu et al. (2006), Lockwood and Moore (1993), Öhman and Eriksson (2002) and Öhman and Lamås (2005). The parameters that are needed to enable the use of a simulated annealing search process include the initial temperature of the system, the number of iterations that are allowed at each temperature, the rate of cooling of the temperature, and the minimum temperature below which the process terminates. A basic simulated annealing search process can be described as:

Step 1. Select a random management unit and harvest period.
Step 2. Determine whether the change in the objective function value is positive (i.e., increasing for a maximization process, decreasing for a minimization process).
Step 3. If the choice is not acceptable, calculate the simulated annealing criteria. If a drawn random number is less than $P(\Delta E)$, the suggested change to the solution is acceptable. If the choice is still not acceptable, return to step 1.
Step 4. If the choice is acceptable, check constraints.
Step 5. If the constraints are violated, return to step 1.
Step 6. If acceptable, formally include the choice in the current solution.
Step 7. If iterations per temperature are equal to the predefined number, change the temperature. Iterations per temperature $= 0$.
Step 8. If temperature is greater than the minimum pre-defined level, return to step 1.
Step 9. Report the best solution.

The parameters used in a simulated annealing process are typically user-defined, and may require several trial runs of the heuristic to locate the appropriate values. In many cases, higher quality objective function values can be obtained by using a high initial temperature and a slow cooling rate. However, this is not a universal rule of thumb. The initial temperature is very influential on the final solution quality. If the initial temperature is not high enough, the search process will not be allowed to move around the solution space freely during the initial stages of the search, and quickly become trapped in an inefficient local optima. Similarly, if the cooling schedule is not fast enough, the search process can spend a significant amount of time searching inefficient solutions.

Rather than requiring the user to define the initial temperature, one enhancement to the search process includes defining the initial temperature based on the initial quality of the first feasible solution. This removes one subjective parameter from the user's control. In addition, the cooling rate, which is typically a linear function (e.g., $0.99\times$ previous temperature) can be modified (made non-linear) to allow more iterations of the model at lower temperatures. This would facilitate a more intensive search when the process has found very high quality solutions. Alternatively, rather than changing the cooling schedule, some logic can be incorporated into the process to simply require more iterations to be used at lower system temperatures.

3.3 Threshold Accepting

Threshold accepting was introduced as a computational technique by Dueck and Scheuer (1990). Applications of threshold accepting in forest management and planning have been described by Bettinger et al. (2002, 2003) and Pukkala and Heinonen (2006). The threshold accepting process is similar to the simulated annealing search process. Users are required to parameterize a problem and specify an initial threshold level, the number of iterations per threshold that will be modeled, the rate of change in the threshold, and the number of unsuccessful iterations that are attempted per threshold. As each management activity is proposed to be added to a schedule, the threshold accepting process determines (a) whether to accept or decline the proposed activity, and (b) whether to change the threshold. When the threshold is reduced to some small number, perhaps 0, the search process terminates and the best solution is reported. A basic threshold accepting search process can be described as:

Step 1. Select a random management unit and harvest period.
Step 2. Determine whether the change in the objective function value is acceptable (i.e., greater than [best solution value – threshold]) for a maximization problem.
Step 3. If the choice is not acceptable, unacceptable iterations = unacceptable iterations +1. If unacceptable iterations is greater than the maximum value, change the threshold, set unacceptable iterations to 0, and return to step 1.
Step 4. If the choice is acceptable, check constraints.
Step 5. If the constraints are violated, unacceptable iterations = unacceptable iterations +1. If unacceptable iterations is greater than the maximum value, change the threshold, set unacceptable iterations to 0, and return to step 1.
Step 6. If acceptable, formally include the choice in the current solution. Iterations per threshold = iterations per threshold +1; unacceptable iterations = 0.
Step 7. If iterations per threshold are equal to the predefined number, change the threshold. Iterations per threshold = 0.
Step 8. If threshold is greater than the minimum threshold level, return to step 1.
Step 9. Report the best solution.

During the development of a forest plan, a management unit and time period for harvest are typically selected at random and proposed as changes to the current solution. The spatial constraints may be assessed at the same time as the threshold accepting criteria (steps 2 and 4), however, if the constraints require extensive computer processing time (as in the case of some spatial constraints), these two concerns may be separated, and the constraints assessed only when one is certain that the threshold accepting criteria are satisfied. The number of unsuccessful changes to the current solution (either due to adjacency violations or failing the threshold accepting criteria test) are counted during each iteration, and when this number exceeds a number specified by the user, the threshold level is reduced. This aspect of a threshold accepting search process is employed to prevent it from becoming stalled during the search.

One enhancement to the threshold accepting process is the use of a non-linear function to change the threshold level. Typically, the threshold is changed linearly, however, preliminary tests on forest management problems involving maximizing even-flow harvest levels suggest that larger rates of change are needed early in the search process, and smaller rates of change are necessary later. The former limits the ability of the search process to wander off into undesirable areas of the solution space, and the latter allows the search process to refine the solution by spending more time searching around the best local optima. Another enhancement, to reduce the time required to parameterize a model, is to base the initial threshold level on the objective function value of the initial, randomly defined solution. Relatively little work has been done in this area to determine the usefulness of the idea, but intuitively, reducing the number of parameters required may make the process more easily implementable by a typical user.

3.4 Great Deluge Algorithm

The great deluge algorithm was introduced as a heuristic search process by Dueck (1993), and has been applied in forest management and planning by Bettinger et al. (2002) and Kim and Bettinger (2005). The great deluge algorithm is similar to threshold accepting, in that it uses a threshold value, called "water level," to determine the acceptability of solutions. Users are required to specify several other parameters as well, including the initial water level, the upper limit of the water level, the rain speed, the total number of iterations, and the number of unacceptable iterations. In the threshold accepting process, the threshold for accepting inferior solutions is dependent on the best solution value (i.e., the values between the best solution value and the best solution value minus the threshold). In the great deluge algorithm, the water level (the threshold above which inferior solutions may be acceptable, for a maximisation process) is not associated with the best solution value, and it increases by a certain rate of change, or "rain speed." Thus each solution's objective function value is directly compared to the current level of water, and it must be above the water level to be acceptable (again, for a maximisation process). The concept of great deluge algorithm is that as water level increases, the algorithm "walks around" (moves around) the solution space to seek a higher peak (higher objective function value) before getting wet or reaching the upper limit of water level. A basic great deluge process can be described as:

Step 1. Select a random management unit and harvest period.
Step 2. Determine whether the change in the objective function value is acceptable (i.e., greater than best solution value or the water level for a maximisation problem).
Step 3. If the choice is not acceptable, unacceptable iterations = unacceptable iterations +1. If unacceptable iterations is greater than the maximum value, go to step 8, otherwise return to step 1.
Step 4. If the choice is acceptable, check the constraints.
Step 5. If the constraints are violated, unacceptable iterations = unacceptable iterations +1. If unacceptable iterations is greater than the maximum value, go to step 8, otherwise return step 1.
Step 6. If the choice is acceptable, formally include the choice in the current solution. Set unacceptable iterations to 0; iterations = iterations +1. Change the water level.

Step 7. If either the total iterations or the water level is greater than the pre-defined maximum value, go to step 8, otherwise, return to step 1.
Step 8. Report the best solution.

Similar to simulated annealing or threshold accepting, the initial water level and the rain speed are critical parameters for obtaining a near-optimal solution. For example, if attempting to solve a problem with a low initial water level and slow rain speed, one can spend a considerable amount of time searching inefficient solutions. On the other hand, if attempting to solve a problem with a high initial water level or a faster rain speed, the search process can be quickly trapped in an inefficient local optimum. Therefore, in order to obtain high quality objective function values, it is recommended that one define the initial water level based on the initial value of the first feasible solution. Moreover, as suggested with simulated annealing or threshold accepting, one can enhance the search process by selecting an adequate rain speed from several trial runs, or use a non-linear function to change the water level.

3.5 Tabu Search

Tabu search was introduced as a search process by Glover (1989, 1990). Tabu search, in contrast to the other heuristics, is generally considered a deterministic process, whereby the best choices are made with each iteration of the model. Applications of tabu search have been described in a number of forest management and planning papers (Batten et al. 2005; Bettinger et al. 1997, 1998, 1999, 2002; Boston and Bettinger 1999; Brumelle et al. 1998; Caro et al. 2003; Richards and Gunn 2000). Most of the previous work on tabu search in forest management utilizes 1-opt decision choices (a change to a single decision variable represents one iteration of the model). However, Caro et al. (2003) and Bettinger et al. (1999) have demonstrated that 2-opt decision choices (a change to two decision variables at once, by switching their choices) improves the solutions that can be generated.

Two parameters are required for tabu search: the total number of iterations, and the tabu state. The tabu state is associated with each decision choice as it enters the solution. For example, should management unit 35, harvest period 3 enter the solution, it will be assigned the maximum tabu state value (e.g., 25). What this implies is that for the next 25 iterations, the choice management 35, harvest period 3 will be tabu (off limits). With each successive iteration, the tabu state associated with each decision choice is reduced by 1, until it once again is 0. At that point, the decision choice is no longer considered tabu. The point of assigning a tabu state to a choice that has been recently made is to prevent the search process from selecting the choice over and over again. However, one caveat to this process is that a choice that is considered tabu can be selected if, by doing so, the best solution ever found will be created. This is considered the aspiration criteria. A typical tabu search process can be described as:

Step 1. Develop a neighborhood of potential changes to a solution.
Step 2. Select from the neighborhood either (a) the choice that provides the greatest increase in solution value, or (b) the choice that provides the least decrease in solution value.
Step 3. Assess constraints.
Step 4. If the choice results in a feasible solution, check tabu status, otherwise return to step 2.
Step 5. If the choice is tabu (has been selected recently, or has a tabu state > 0), but does not result in the best solution found so far, return to step 2.
Step 6. Formally include the choice in the current solution. Iterations=iterations+1; tabu state for this choice=a value pre-defined by the user; tabu state for other choices=previous tabu state – 1.
Step 7. If iterations are less than the predefined maximum number, return to step 1.
Step 8. Report the best solution.

The development of the neighborhood is the most time-consuming part of the search process. Depending on the size of the neighborhood, many thousands of computations can be performed prior to the selection of a single potential change to the solution. Threshold accepting and simulation annealing, by comparison, perform very few computations prior to selecting a choice. Theoretically, the size of a full 1-opt tabu search neighborhood is

$$\text{number of choices per management unit} \times \text{number of management units}$$

This takes into account the fact that a management unit can be unscheduled, and that the calculation for the current choice is not needed. Neighborhoods for 2-opt moves are even more time-intensive, as the theoretical size of a full 2-opt tabu search neighborhood is:

$$\text{number of management units} \times \text{number of management units}$$

Some (Bettinger et al. 2007) have proposed using a region-limited neighborhood that assesses a smaller set of potential changes to a solution. This region-limited neighborhood can be randomly defined, or can systematically shift with each iteration. Either types of processes have shown promise, and effectively increase the speed of the search process.

The tabu state is perhaps the most important aspect of the search process. Selecting an inappropriate tabu state could result in a cycling of solutions. For example, if the tabu state were, say 10 iterations, a tabu search process could simply return to the same solution every 10 iterations of the model, and be ineffective at locating the best local optima possible. Therefore, some trial and error parameterization is required to determine the most appropriate tabu state that prevents recycling of solutions. Since each problem has been shown to require a different tabu state, this process should not be avoided. One enhancement to the selection of the tabu state includes randomly changing the tabu state value with each iteration, however, preliminary (and un-reported) tests have shown this to have mixed results on the final solution quality.

3.6 Genetic Algorithms

The genetic algorithm search process was initially described by Holland (1975). The search process is based on an analogy with biological reproductive processes, and is favored by many ecologists as a way to search a solution space for highly desirable near-optimal solutions to forest management problems. A number of researchers (Boston and Bettinger 2002; Falcão and Borges 2001; Lu and Eriksson 2000; Mullen and Butler 2000) have described applications of genetic algorithms in forest management and planning. The genetic algorithm approach to the development of a management plan is unique in that two solutions are used to develop two new solutions, each of which are then assessed (based on their resulting objective function values) and considered for retention.

A genetic algorithm search process uses two parents drawn from a population. They are split and recombined using a cross-over process, and two children solutions are created. The parameters that are required to utilize the search process include the size of the population and the mutation rate. The population is a set of feasible solutions contained in computer memory, from which two will be drawn and used to create two new solutions. The process for drawing two solutions from memory can be purely random, or based on the fitness of each solution in the population. Fitness is an analogy for the objective function value of each solution. The mutation rate is a value that is used to decide whether to randomly change the status of a single decision variable. This is performed after the cross-over, and before the objective function value is assessed. A generic genetic algorithm search process can be described as:

Step 1. Select two parent solutions from the population.
Step 2. Determine where to initiative the cross-over process.
Step 3. Split the parents and create the children solutions.
Step 4. Apply mutations to each allele (decision variable) of each child solution.
Step 5. Check the constraints.
Step 6. If a child solution is feasible, assess the objective function value. If neither of the child solutions are feasible, return to step 1.
Step 7. Select the higher quality of the feasible child solutions, discard the other.
Step 8. Select a parent solution from the population, and discard it.
Step 9. Place the best child solution from step 7 in the population. Iterations = Iterations+1.
Step 10. If iterations are less than the maximum number pre-defined, return to step 1.
Step 11. Report the best solution.

The cross-over process is problematic with spatial forest planning goals. Typically, a cross-over process will split two parent solutions into two pieces. Assume, for example, that parent A had a schedule (time periods of clearcut harvest) for five management units of [2,1,3,1,1], and parent B had a schedule of [1,3,0,2,2]. Assume now that a random location was chosen for the cross-over (after the second unit in the schedule). Child AB would become [2,1,0,2,2] and child BA would become [1,3,3,1,1]. The spatial juxtaposition of each of these five units is not considered in the cross-over, and tests have shown that simply choosing a single cross-over point and creating two child solutions may result in numerous adjacency constraint violations. To ease this limitation of genetic algorithms, some unpublished tests have indicated that the transfer of a small amount of genetic code could result in fewer infeasible solutions, and perhaps higher valued final solutions. In this case one would select a cross-over point at random, but only transfer a small part of each solution to the other (i.e., one or two time periods of clearcut).

The newly created solutions may be mutated randomly depending on the rate of mutation. Here, one would draw a random number for each of the alleles (decision variables), and if the random number is smaller than the mutation rate, the value of the allele (the harvest timing, in this case) would be randomly changed. From extensive tests of various mutation rates, we have found that selecting a high mutation rate results in poor solutions. Selecting a mutation rate of 0 generally leads to lower quality solutions as well. After mutating each child solution, potential constraint violations could occur as well.

The process of retention of the best child solution that is produced could also vary. Generally, the best child replaces a random parent in the population. However, the best child solution could replace the worst parent in the population, or replace a parent based on fitness (lower probability of being replaced as solution quality increases relative to the other parents). Each of these replacement processes may seem intuitively superior to simply randomly replacing a parent in the population, particularly when the parent replaced could be the highest quality solution. However, some testing of these processes in forest management planning problems has shown that simply replacing the lowest quality solution with the best child may eventually result in a population of clones (each with very similar solution values). Therefore, the retention of some variation of genetic code may be important.

4 Applications

Two example applications are provided to illustrate the use of spatial optimisation and computational techniques in forest management. The first involves the development of a forest plan for a hypothetical southern U.S. forest products company that wants to maximise the net present value of their plan while acknowledging spatial constraints related to the timing and placement of clearcut harvests. The second involves the spatial arrangement of management activities in the interior western U.S. This is a research project that examined the placement of operational management prescriptions in patterns across the landscape to determine whether the patterns of activities would have an influence on severe wildfire behavior.

4.1 Industrial Application from Southeastern U.S.

Southern U.S. forest products companies, when developing forest plans, typically have the goal of maximizing some economic or commodity production values. These goals may include the net present value, the periodic cash flow, wood flows, or product mixes that can be tracked within forest plans. We present an example of maximizing the net present value of a forest plan, while acknowledging spatial constraints related to the timing and placement of clearcut harvests. The objective function includes maximizing the value of products produced over time (discounted using a 5% interest rate), while also recognizing regeneration costs. The main activity that is modeled is clearcutting. The time horizon is 40 years, divided into 40 one-year time periods. Yields were derived from the Plantation Management Research Cooperative (1996), and represent typical yields for site 75 (base age 25) loblolly pine (*Pinus taeda*) plantations in the southern U.S. The hypothetical forest consists of 5,262 hectares of deciduous and coniferous forests. We assume that the 2,875 hectares of coniferous forests (Fig. 1) are composed of loblolly pine, and that the harvests will be scheduled in these areas. There are 175 pine management units, with an average size of 16.2 hectares.

The spatial consideration that is included in this forest plan concerns the timing and placement of clearcuts harvests. There are two aspects to the spatial consideration: the green-up period (the amount of time required between adjacent clearcuts), and the allowable size of clearcuts.

To illustrate a set of alternative forest plans, we used the tabu search model described in Batten et al. (2005), and the following parameters:

1-opt tabu state	250 (unit restriction model)
	400 (area restriction model)
2-opt tabu state	30
1-opt iterations	100
2-opt iterations	10
1–2 opt loops	7
maximum clearcut area per year	80.9 hectares

The parameters were selected based on numerous trial runs of the heuristic applied to this problem. When using the unit restriction model of adjacency, forest plans with green-up periods from 2–5 years were developed. When using the area restriction model of adjacency, a maximum clearcut size of 48.6 hectares was assumed, and forest plans with green-up periods from 2–5 years were developed. Some management units are larger than the maximum clearcut area allowed per year and the maximum clearcut size, therefore these constraints are relaxed in the Batten et al. (2005) model to allow a management a unit that is larger than the constraint to be scheduled by itself in any one year.

The results generated using this forest planning methodology, which includes spatial optimisation and computational techniques, allow managers to understand

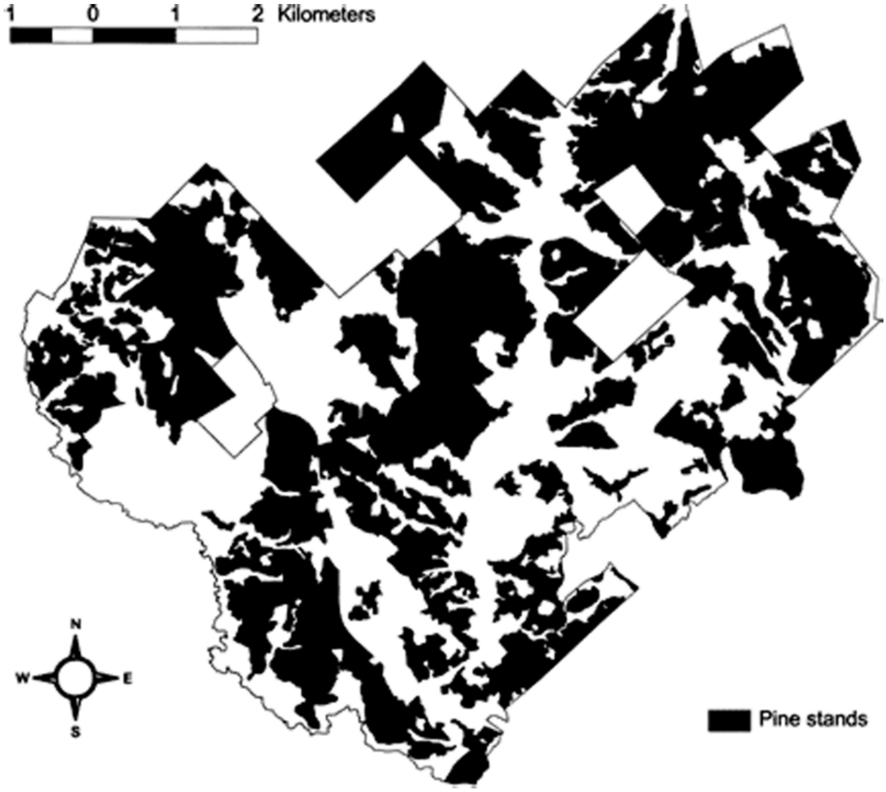


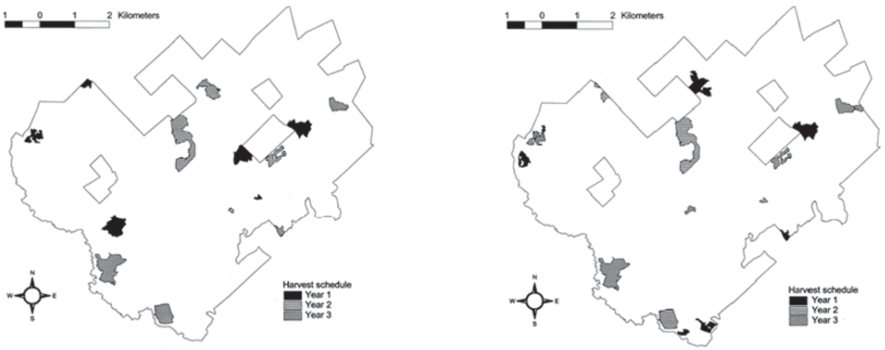
Fig. 1 Pine stands within a hypothetical 5,262 hectare southern U.S. forest

the trade-offs among planning goals. For example, if either of the adjacency models were used to guide the implementation of activities across a landscape, one could begin to understand the cost associated with increasing the green-up period from 2–5 years, for example (Table 1).

One could also get a sense for the trade-off involved in using the area restriction model of adjacency rather than the unit restriction model. The area restriction model generally allows more flexibility in the spatial and temporal arrangement of

Table 1 Net present value associated with several forest plan alternatives for a hypothetical 5,262 hectare southern U.S. forest

Green-up period (years)	Unit restriction model of adjacency	Area restriction model of adjacency
	Net present value (US dollars)	
2	6,531,104	6,566,382
3	6,484,751	6,555,483
4	6,456,008	6,510,795
5	6,429,743	6,474,831



(a) (b)

Fig. 2 Harvest schedules for a hypothetical 5,262 hectare southern U.S. forest designed using (a) the unit restriction model of adjacency and (b) the area restriction model of adjacency

activities across the landscape, therefore, objective function values are generally higher than when using the unit restriction model. Land managers can also be given a map that illustrates the first few years of a plan, so that they can begin to develop operational management plans to complement the tactical management plan (Fig. 2).

4.2 Spatial Arrangement of Fuels Management Treatments in the Western U.S.

Many western North American forests are faced with a high risk of a catastrophic wildfire, and to reduce the potential consequences, fuel management prescriptions have been extensively applied. While individual management prescriptions might alter wildfire behavior on a local scale, some have proposed arranging the treatments in patterns across the landscape to mitigate the landscape-scale effects of wildfire. In examining the influence of random patterns of fuel management activities, Finney (2003) suggested that relatively large proportions of the landscape must be treated to substantially reduce fire sizes. Regular patterns of management activities (Finney 2001) have also been shown to reduce fire spread rate in a similar fashion to parallel strips. Each of these approaches assumes that there is some limiting factor (e.g. budget) that prevents the widespread implementation of fuels reduction management prescriptions, thus the need to spread a limited amount of treatments efficiently across the landscape.

In a recent research project (Kim 2006), a planning process was developed to schedule management activities in four patterns across the landscape. Simulated wildfires were then applied to the landscape (after simulation of the first decade's activities) to determine whether the pattern of activity and subsequent changes in fuels and forest structure could mitigate severe wildfire behavior. The area considered was the Grand Ronde River Basin (approximately 178,000 hectares) in northeastern Oregon (USA). Most of the area is managed by the U.S. Forest Service. Data regarding stand structure and available harvest volume for each management unit were utilized to assess changes in stand condition and harvest volume from several management activities that were simulated over 10 ten-year periods (100 years) using a stand-level optimisation model (Bettinger et al. 2005).

Four spatial patterns of activities were tested: dispersed, random, clustered, and regular. An objective function was designed for each pattern to simultaneously optimise timber harvest volume and the pattern of activity. For example, with the dispersed pattern, management units assigned treatments were assumed to maximise the mean distance between each other. An even-flow harvest volume was a desired outcome as well, and the resulting objective function was to:

Minimise

$$\sum_{t=1}^{T}\left(\left|\sum_{i=1}^{N_t} H_{it} - TV\right|\right) - \sum_{t=1}^{T}\left(\frac{1}{N_t}\cdot\sum_{i=1}^{N_t} D_i\right) \tag{6}$$

Where:

H_{it} = Harvest volume from unit i in time period t
TV = Target timber harvest volume
D_i = Distance between centroids of unit i and its nearest neighbor
i = Index of management units scheduled for harvest
t = A time period
T = Total number of time periods $(T = 10)$
N_t = The set of management units scheduled for harvest in time period t

Two sets of management prescriptions were assessed: operational management prescriptions, and those specifically designed as fuels reduction management prescriptions. Operational management prescriptions were developed with the goal of maintaining stand density levels within a target range (35%–55% Stand Density Index), a strategy now preferred on government land in the study region. A stand-level optimisation process for developing efficient management regimes that uses dynamic programming and a region-limited search strategy was also employed (Bettinger et al. 2005). Operational considerations, such as minimum harvest levels and harvestable diameter ranges, were also used as constraints in developing the management prescriptions. Four fuels management prescriptions were designed, with the goal of maintaining a desired stand density target by thinning small-diameter trees. The first two limited the range of diameters that could be removed to two groups: (1) 2.5–17.8 cm, and (2) 2.5–25.4 cm. The management prescriptions also included a residual basal area constraint of $18.4\,\mathrm{m}^2\,\mathrm{ha}^{-1}$ and the need to maintain stand density within a specific range (35%–55%). Two additional management prescriptions were produced by assuming (1) prescribed fires would be implemented in the same period as the thinnings, and (2) all surface fuels less than 2 m in height would be assumed killed. To accomplish this, the original 2 fuels management prescriptions were modified to remove from the associated tree lists all trees less than 2 m in height.

Patterns of activities were generated with operational and fuels management activities, and an example is provided in Fig. 3.

Fig. 3 Regular spatial pattern of management activities developed for a 178,000 hectare forest in northeastern Oregon (USA)

Some of the advantages of this work were (1) one could develop a computational model to design the pattern of activity across and landscape while also accommodating commodity production goals, and (2) the activity scheduling and subsequent wildfire simulations could be performed for a large area in a reasonable amount of time. Some aspects of simulated severe wildfires can be moderately mitigated by spatially optimising the pattern of activity. However, the drawbacks of the planning methodology were (1) some of the patterns, while visually verifiable, could not be validated as the level of activity increased, and (2) simply scheduling operational treatments in a pattern across the landscape does not guarantee that simulated fire behavior could be altered, since the operational management prescriptions do not address surface or ladder fuels.

5 Summary

Computational techniques for accommodating spatial and temporal forest management goals in forest plans range from the traditional mathematical programming to heuristic techniques. One uses equations and methods for simultaneously solving those equations to develop a forest plan, and the other uses logic and rules of thumb to iteratively build a forest plan. The main limitations related to the traditional mathematical programming techniques are (1) the inability to formulate complex problems as linear equations, (2) the inability to solve some problems in a reasonable amount of time, and (3) a limit on the number of rows (constraints) or variables that can be included in a problem. These limitations become less of an issue as computer technology, both hardware and software, evolves. The main limitations related to heuristics are (1) the time required to develop a method for each specific planning problem, and (2) the inability to guarantee that the optimal solution can be located. The main advantage of using a traditional mathematical programming technique is that when a solution is generated, one has confidence that it is the optimal solution to the problem being solved (perhaps within some minor tolerance in the case of integer programming). The main advantage of using a heuristic is that one can quickly generate very good solutions to complex problems, if the heuristic is developed appropriately.

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Integrating Multiple Services in the Numerical Analysis of Landscape Design

Timo Pukkala

1 Introduction

Planning of the design of forested landscapes employs both qualitative and quantitative methods, and subjective and objective evaluations. Both types of analyses and evaluations are useful and necessary in most planning cases. This chapter concentrates on quantitative methods that may help managers to find the optimal management of future landscapes in situations where there are several simultaneous management objectives. Quantitative analyses are efficient, quick, cheap, objective and repeatable. These are benefits that should not be neglected in the planning of the management of large forest areas.

The multifunctional nature of forests has been recognised more and more clearly in forest management and forest planning. Most of the traditional quantitative techniques and models that have been developed to support management decisions and planning calculations deal only with the economic function of the forest. Other functions such as ecological, environmental and social functions are taken into account in a subjective and expert-oriented way. However, the use of expert opinions in routine planning is often too costly. Modern planning is cost-effective and based on highly computerised quantitative information systems. Such systems enable detailed calculations to be made cheaply. In situations where the forest is large, several stakeholders are involved in planning, there are several management objectives, and the planning horizon is long, the task of finding the best management for the forest is so difficult that it is far beyond the capacity of human brains. Therefore, computer-aided numerical planning may also improve the quality of plans, in addition to reducing planning costs.

If forest's functions other than timber production cannot be integrated in the quantitative planning systems, these functions are often neglected in the analyses that are conducted to get support to management decisions. This decreases the usefulness of such analyses. Another consequence of lacking quantitative methods is

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inefficient forestry, which means that the cost of producing certain benefits is higher than it would be in the optimal use of resources.

The approach that is followed in this chapter to design forest landscapes corresponds to the approach proposed in management science for solving decision problems (Fig. 1). In this approach, an important step is to prepare a situation-specific model, which describes how the services that the forest provides depend on decision variables, which are variables that the decision-maker can control. This means that forest planning is modelling. The model that is prepared in a planning effort may be called as a planning model or a management model. In addition to describing the relationships between controllable variables and forest services, the model also accommodates information about the management objectives and preferences of stakeholders. Developing the planning model means deciding the type of the model, and selecting the decision variables, objectives and constraints included in the model.

Another important step of planning is to fill the model with data. These data tell, firstly, how much the decision variables produce or consume the objective or constraining variables, and secondly, what are the target levels of objective variables and constraints, and how important different management objectives are. The first type of data usually comes from simulators that combine inventory data to the information contained in models. The models used in simulators are based of previous research, and they typically describe the relationships between different variables in a stand (for instance, growth model, volume model, mushroom yield model). The second type of data describes the preferences of the decision makers, and it comes from the decision makers.

Once the planning problem has been formulated as a model and the model filled with data, the problem can be solved with an automated technique, the result

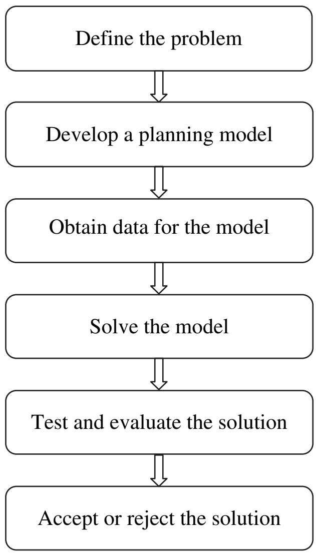


Fig. 1 Quantitative approach to solving decision problems in management science

being a solution for the planning problem, or an optimal management plan (Fig. 1). However, since numerical planning models seldom are perfect descriptions of planning problems, additional analyses are required to test the solution and to gain confidence towards the solution. These analyses may include visualisations of future landscapes, and sensitivity analyses that show how the optimal management would change if some uncertain parameters are altered.

Several questions should be considered when the above approach is used in the planning of forested landscapes which have multiple functions. This chapter concentrates on the following questions:

1. How to numerically describe the different functions of the forest?
2. Which kind of planning model should be used?
3. Which kind of optimisation technique should be used to solve the planning model?

All these questions are interlinked, especially questions 2 and 3 since the way of modelling the problem to some extent determines the optimisation technique. In addition, the types of management objectives affect the choice of suitable planning models. Relevant things are for instance whether or not the objective variables and constraints are linear functions of controllable variables, and whether or not some of the management objectives are spatial.

This chapter discusses the above questions in the planning of multifunctional management of forest landscapes. It begins with a classification of planning models suitable for multifunctional forestry (Section 2), after which alternative methods to solve the problems described by planning models are briefly discussed (Section 3). However, this discussion is succinct because the topic is dealt with in more detail in the other chapters of this book. Section 4 gives examples of how to numerically describe forest services other than timber. The chapter is not a systematic review of research conducted on multi-functional forest planning. Most of the examples given in Section 4 are taken from the author's own research.

2 Planning Models for Multifunctional Forestry

There are several alternatives to formulate a planning model that accommodates several simultaneous management objectives. Categorising planning models is difficult if one wants to avoid gaps and overlaps. This article uses the following classification to structure the discussion:

1. Monetary model
2. Mathematical programming model

    - Linear programming, integer programming and mixed integer programming models
    - Goal programming model

3. Penalty function model
4. Multi-attribute utility model
5. Compromise programming model

2.1 Monetary Model

The monetary model corresponds to the economic approach in which all the benefits and costs are converted into money and alternative solutions are evaluated using the net present value. In multifunctional forestry, the net present value of one rotation can be calculated as follows (adapted from Hartman 1876):

$$NPV = \sum_{t=0}^{T} \frac{R_t - C_t}{(1+i)^t} + \sum_{t=0}^{T} \frac{B_t + M_t + S_t}{(1+i)^t} \tag{1}$$

where T is rotation length (years), R_t, C_t, B_t, M_t and S_t are, respectively, timber production revenue, timber production cost, biodiversity benefit, mushroom benefit, and scenic beauty benefit in year t, and i is discounting rate. In this example, the non-timber benefits are biodiversity maintenance, mushroom collecting, and scenic amenities, but they could be other benefits as well. The net present value can be converted into land expectation value, which is the eventual objective function in the monetary approach:

$$LEV = NPV \left[1 - \frac{1}{(1+i)^T}\right]^{-1} \tag{2}$$

Although the monetary approach is the easiest and in a way the most exact approach, it is not used much in landscape level planning when there are non-timber management objectives. This is because monetary valuation of several benefits is difficult. Management planning usually aims at maximising the benefit of the forest owner, which means that valuation functions developed earlier for other people or society cannot be used. However, monetary valuation and net present value are commonly used in the other planning model types to describe part of the benefits (usually timber production benefits).

A practical problem in calculating the land expectation value is that planning systems do not always simulate a full rotation to get the NPV of one rotation. In many cases, only planning horizons clearly shorter than the rotation length are simulated, and the LEV is obtained as the sum of the NPV of the simulated period plus the NPV of the ending inventory (growing stock at the end of the simulation period). Especially in a multifunctional case it is difficult to calculate the value of ending inventory. Moreover, for some benefits the LEV is not equal to the NPV of the first rotation multiplied by $[1 - 1/(1+\mathrm{i})^{\mathrm{T}}]^{-1}$. This is the case especially with carbon balance, which always returns to zero when a certain type of silviculture is continued for a long time. The carbon balance is positive or negative only temporarily, when forest biomass is increased or decreased, or the set of timber assortments is altered so that the life spans of products change.

2.2 Mathematical Programming Model

When the planning problem is formulated as a linear programming model, the following objective function (or its modification) is maximised subject to linear constraints:

$$z = \sum_{j=1}^{n} \sum_{i=1}^{n_j} c_{ij} x_{ij} \tag{3}$$

where z is objective function, x_{ij} is the area of stand j treated according to schedule i, n is the number of stands (compartments), n_j is the number of treatment schedules in stand j, and c_{ij} indicates how much one area unit of alternative i of stand j produces the objective variable.

One of the goal variables is selected as the objective variable while the others are controlled trough constraints. The linear programming, integer programming and mixed integer programming models are discussed into more detail in another chapter of this book.

An alternative problem formulation, called as goal programming model, has also been developed. In this model the objective function, which is minimized, consists of the deviations (d) of objective variables from their target values:

$$z = \sum_{k=1}^{K} d_k^+ + \sum_{k=1}^{K} d_k^- \tag{4}$$

where $d_k{}^+$ indicates how much the target value of objective k is exceeded (surplus), and $d_k{}^-$ is the quantity by which target k falls short (slack). The target levels (b_k) are given through equations that are called as goal constraints:

$$\sum_{j=1}^{n} \sum_{i=1}^{n_j} a_{ijk} x_{ij} + d_k^- - d_k^+ = b_k \quad k = 1, \ldots, K \tag{5}$$

where a_{ijk} indicates how much one area unit of management schedule i of stand j produces or consumes objective variable k. These constraints together with the objective function mean that levels b_k are pursued: deviations are regrettable (they are minimized) but they are allowable (constraints are flexible). If only falling short of the target is harmful but not exceeding it, the surplus variable $(d_k{}^+)$ is left out from the objective function and goal constraint. If only exceeding is to be avoided, the slack variable $(d_k{}^-)$ is dropped. It is also possible to multiply the deviation variables in the objective function by constants that reflect the relative importance of the respective objective variables. Another improvement is that of scaling the deviation variables to the same range of variation (usually between 0 and 1) so that the units of measurement do not affect the actual weights of the goals.

2.3 Penalty Function Model

Another possibility is to add a penalty function to the objective function of linear programming to make it multi-objective:

$$z = \sum_{j=1}^{n} \sum_{i=1}^{n_j} c_{ij1} x_{ij} - \sum_{k=2}^{K} w_k \left| q_k - T_k \right|^{\alpha} \tag{6}$$

where w_k, q_k ja T_k are the weight, quantity and target value for objective k. Parameter α determines how fast the penalty increases as a function of deviation from the target level. The idea is to maximise one objective variable $(k = 1)$ but simultaneously aim at good values of other objective variables $(k = 2, \ldots, K)$.

The objective variables included in the penalty function depend on decision variables (x_{ij}), which can be expressed as follows

$$q_k = Q_k(\mathbf{x}) \tag{7}$$

where Q_k is a function that specifies how q_k depends on decision variables and $\mathbf{x}$ is a vector of decision variables (i.e., values of x_{ij}). In this formulation the penalty variables are not assumed to be linear functions of decision variables. If the objective variable is also a non-linear function of decision variables the objective function may be expressed as follows:

$$z = Q_1(\mathbf{x}) - \sum_{k=2}^{K} w_k \left| Q_k(\mathbf{x}) - T_k \right|^{\alpha} \tag{8}$$

The model can be solved with several heuristic methods.

2.4 Utility Function Model

Another way to accommodate several forest services into the planning model is to use utility function. One utility function type is the general additive model, which expresses how the utility experienced by decision maker depends on the values of objective variables

$$U = \sum_{k=1}^{K} w_k u_k(q_k) \tag{9}$$

where U is total utility, K is the number of objective variables, w_k is the weight, u_k the sub-utility function and q_k the quantity of objective k. The sub-utility functions express how much different amounts or levels of the objective variable contribute to the utility (Fig. 2). They also scale all objective variables between zero and one, i.e., remove the original units of variables. This has the benefit that weights w_k can be estimated without considering units, which makes the estimation easier. Several methods have been proposed in decision analysis literature to estimate the objective weights and derive the sub-utility functions.

Following the utility theoretic approach, the complete problem may be formulated, for instance, as follows:
Maximise

$$U = \sum_{k=1}^{K} w_k u_k(q_k) \tag{10}$$

Subject to

$$q_k = Q_k(\mathbf{x}) \quad k = 1, \ldots, K \tag{11}$$

$$\sum_{i=1}^{n_j} x_{ij} = 1 \quad j = 1, \ldots, n \tag{12}$$

$$x_{ij} = \{0, 1\} \tag{13}$$

where $\mathbf{x}$ is a vector of 0–1 variables (x_{ij}) expressing whether $(x_{ij} = 1)$ or not $(x_{ij} = 0)$ treatment alternative i of stand j is included in the solution, n_j is the number of treatment alternatives for stand j and n is the number of stands. Note that the use of utility model does not require that the decision variables are 0–1 variables. However, the model is often solved with heuristics that treat them as 0–1 variables.

The general additive utility function assumes that a poor performance in one objective can be compensated for by a good performance in the other objectives. If

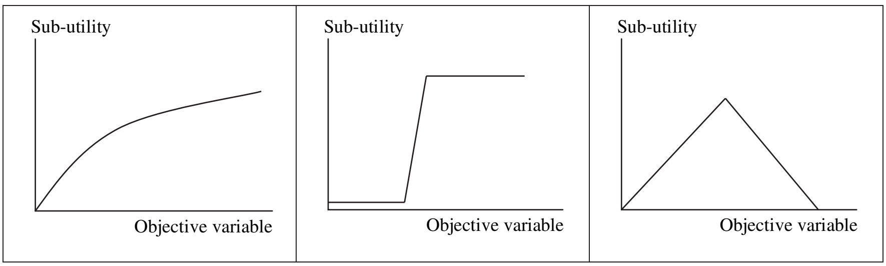


Fig. 2 Examples of sub-utility functions. The function on the left represents decreasing marginal utility. The middle function corresponds to “greater than” constraint and the function on the right “equal to” constraint

this is not in line with the true preferences, i.e., utilities through different objectives are not interchangeable, it is possible to use multiplicative utility function:

$$U = \prod_{k=1}^{K} (u_k(q_k))^{w_k} \tag{14}$$

The value of a multiplicative function is zero if any of the sub-utilities is zero. Multiplicative function is a realistic option for instance in ecological planning where the objective variables are habitat areas, habitat suitability indices, or predicted population densities of different species or species groups. To maintain biodiversity, none of the habitat types should ever be zero. If a part of the objectives are interchangeable and the other part is not, a combination of additive and multiplicative utility function can be used:

$$U = \left[\sum_{k=1}^{I} w_k u_k(q_k)\right] \prod_{k=I+1}^{K} (u_k(q_k))^{w_k} \tag{15}$$

where I is the number if interchangeable objectives and $K - I$ is the number of non-interchangeable objectives.

Utility models may be formulated in very flexible ways for different planning situations. For example, if there are several decision-makers with their own preferences, the following model can be used:

$$U_= \sum_{p=1}^{P} \left(v_p \sum_{k=1}^{K_p} w_{kp} u_{kp}(q_{kp}) \right) \tag{16}$$

where P is the number of decision-makers, v_p is the weight of decision-maker p, K_p is the number of management objectives set by decision-maker p, w_{kp} is the importance of objective k of decision-maker p and u_{kp} is a sub-utility function designed by decision-maker p for objective k and q_{kp} is the quantity of objective variable k of decision-maker p. In the case that there are several forest holdings and owners, and part of the management goals are common for the whole forest that consists of several properties, the following model can be used (Kurttila and Pukkala 2003):

$$U = v_l \sum_{j=1}^{J} w_j u_j(q_j) + \sum_{p=1}^{P} v_p \sum_{k=1}^{K_p} w_{kp} u_{kp}(q_{kp}) \tag{17}$$

where U is the total utility, v_l is the weight of the landscape level utility model, J is the number of landscape level objectives, w_j is the relative importance of landscape level management objective j, u_j is a sub-utility function for management objective j, q_j is the value of objective *j, P* is the number of forest holdings, v_p is the weight of holding p, K_p is the number of management objectives in holding p, u_{kp} is a sub-utility function in holding p for management objective k, q_{kp} is the value of objective k in holding p, and w_{kp} is the relative importance of management objective k in holding p.

If some management goals are stand-level objectives and another part are forest level objectives, the following utility function can be used (Pukkala et al. 2007):

$$U = v_f \sum_{j=1}^{J} w_j u_j(q_j) + v_s \sum_{n=1}^{N} \frac{a_n}{A} \sum_{k=1}^{K} w_k u_k(q_{kn}) \tag{18}$$

where J is the number of forest level goals, K the number of stand-level goals, N the number of stands, v_f the weight of forest level management goals, v_s the weight of stand level goals, a_n the area of stand n, and A the total area of the forest. Symbols w_j, u_j and q_j are the importance, sub-utility function and quantity of forest level objective j, and w_k, u_k and q_{kn} are the same for stand level objective k.

If risk is included in the optimisation and every decision alternative has several possible outcomes (corresponding to different states of nature), the model could be

$$U = \sum_{r=1}^{R} \left(p_r \sum_{k=1}^{K} w_k u_k(q_{kr}) \right) \tag{19}$$

where p_r is the probability of outcome r, R the number of states of nature, and q_{kr} the amount of objective variable k in outcome r. Taking into account that the above versions of utility model can be combined, it can be concluded that the utility modelling approach allows the planner to model rather easily extremely complicated planning problems.

2.5 Compromise Programming Model

Compromise programming minimises the following objective function:

$$\min H = \left[\sum_{k=1}^{K} w_k^p \left(\frac{q_k^{\max} - q_k}{q_k^{\max} - q_k^{\min}} \right)^p \right]^{1/p} \tag{20}$$

where ${q_k}^{\min}$ and ${q_k}^{\max}$ are the minimum and maximum possible value of objective variable k among the decision alternatives, w_k is the weight of objective k and p is a parameter that determines how quickly the regret increases as a function of deviation from the best possible value. The idea is to measure the loss or regret compared to such a hypothetical alternative in which every objective variable is at the best possible level. The compromise programming model can be interpreted as a utility model (or regret model), and its use therefore corresponds to that of the utility model.

The planning models corresponding to the penalty function, utility function or compromise programming approaches can also have constraints. However, especially with the penalty and utility functions they are not often needed because the constraining variables can be included in the objective function. Therefore, the only necessary constraints in these formulations are the ones that restrict the sum of the decision variables for one stand to be equal to the area of the stand.

3 Optimisation Techniques for Multifunctional Forestry

Optimisation techniques available for solving landscape level planning problems have been described in the other chapters of this book. The main categories of optimisation techniques are mathematical programming and heuristics (Borges et al. 2002). Heuristics are not always called optimisation techniques (but only search methods) because it is not guaranteed that they really find the optimal solution. As previously mentioned, the optimisation technique depends to some extent on the planning model, which in turns depends on the type of objectives considered in planning. For instance, solvers developed for linear programming (like the simplex algorithm) can be used to solve linear, integer, mixed integer and goal programming models, but not to solve the other planning model types. On the other hand, solvers suitable for the other planning model types can also solve linear programming type of problems, but not as efficiently as the mathematical programming algorithms. All planning model types described above can accommodate at least some spatial relationships. Therefore spatiality is not the criterion which alone determines whether a mathematical programming model is sufficient or if e.g. utility function should be used in combination with heuristic search methods. However, planning models other than linear programming are more flexible in accommodating non-timber forest services that are non-linear functions of decision variables.

Models other than mathematical programming models are solved with heuristic search methods. Most of the standard heuristics used in forestry (simulated annealing, tabu search, threshold accepting, and great deluge) are often used by making gradual changes (local improvements) in the current solution, e.g. by changing the treatment in one stand only. The solutions that can be obtained with one change (move or movement) are the neighbours of the current solution. In genetic algorithm new solutions are obtained by making more substantial changes to the current solution. Ant colony optimisation is another heuristic which is not a local improvement method (Zeng et al. 2007a). Some results (Palahí et al. 2004; Pukkala and Kurttila 2005) suggest that genetic algorithm performs better than the local improvement heuristics if there are complicated spatial objectives. Ant colony optimisation is also promising in spatial problems (Zeng et al. 2007a). However, making movements in which the treatment schedule is changed simultaneously in two stands has been found to improve the performance of local improvement heuristics in spatial problems (Heinonen and Pukkala 2004), narrowing the difference between local improvement methods and genetic algorithms, or between simple and more complicated heuristics (Fig. 3).

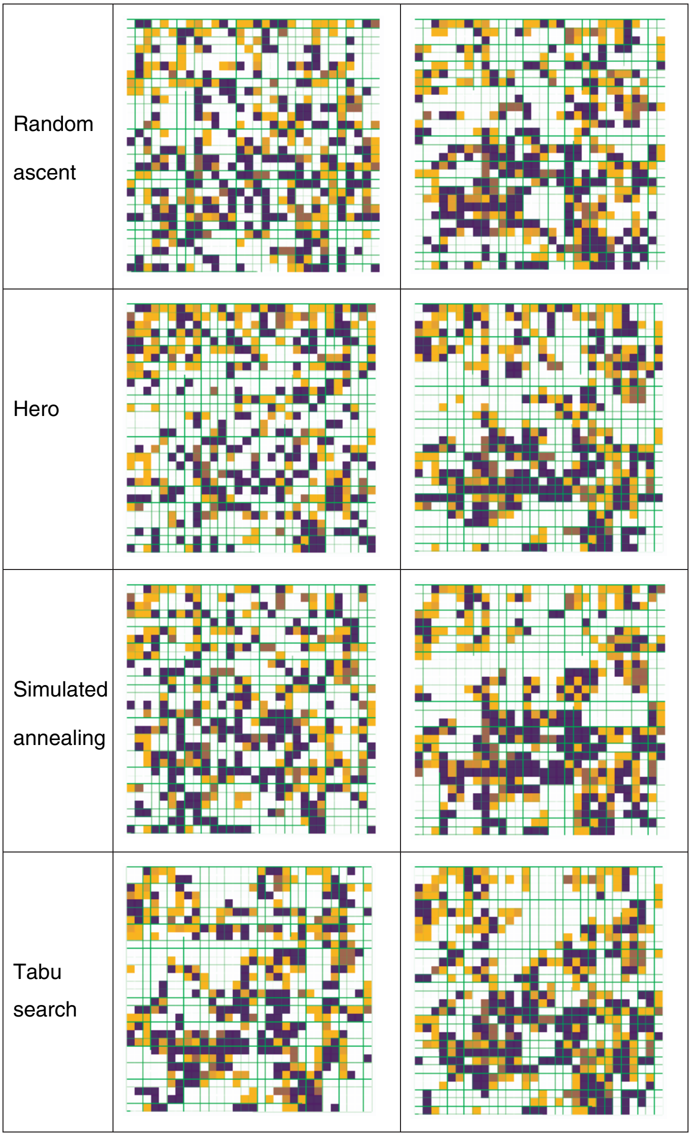


Fig. 3 Location of cuttings (shaded cells) in the optimal solutions for the same planning problem with four heuristics when 1-stand (left) or 2-stand (right) neighbourhood is used in the heuristic search. One management objective is to aggregate cuttings. Two-stand neighbourhood uses movements in which the treatment is changed simultaneously in two stands. The darker is the shading the more intensive is the cutting (black is clear-felling). The forest is imaginary and consists of 900 square-shaped stands

However, there are so few studies on the suitability of different heuristics to different planning problems that it is impossible to suggest a particular algorithm to a particular problem.

From the viewpoint of an optimisation technique the management objectives can be divided into three categories:

1. Stand level goals which are stand level variables for which the forest level value is a simple sum of stand values. Net present value, land expectation value, mushroom yield, and growing stock volume are examples of goals of this category. If the utility or priority depends on one of these goals or is a combination of several such goals, the optimisation problem reduces to finding the best management schedule separately for every stand.
2. Neighbourhood goals, which are stand level variables, the value of which depends on the value of the same variable in adjacent stands or in the proximity of the stand. These goals can be used for dispersion and aggregation purposes. For example, if a stand is an old forest patch, its value as an old forest patch increases if the neighbouring stands are also old forest patches (if aggregation is wanted).
3. Global or forest level goals, which may be non-spatial or spatial. They can be evaluated only by taking all the stands into account. Even-flow harvesting target, a certain size distribution of regeneration areas, and a certain percentage of old forest habitat are examples of global goals.

If all the goals belong to categories 1 and 2, local optimisation techniques such as cellular automata (Mathey and Nelson 2007; Heinonen and Pukkala 2007; Pukkala et al. 2007) can be used in optimisation. Linear programming can be used if all goals belong to category 1 or are non-spatial goals in category 3. The idea of local optimisation is to select the best treatment schedule separately for each stand. This makes the optimisation problem radically simpler and faster to solve as compared to problems where the best combination of stand level management options should be found. A simple example illustrates the situation. If a forest has 100 stands and each stand has 5 different management options, there are 500 different management options in total but as many as $5^{100} = 7.88861 \times 10^{69}$ combinations of stand level management alternatives. There are 1.57772×10^{67} times more combinations than there are stand level management alternatives. Local optimisation techniques only need to evaluate 500 management alternatives to find the optimal plan for the forest if all management objectives belong to the first category, but global techniques should, in principle, evaluate 7.88861×10^{69} combinations. If a part of the management goals belong to category 2, the selection of the best schedule for every stand must be repeated several times because the ranking of the alternatives of a certain stand may change depending on which treatments were selected to neighbouring stands. However, the process converges quickly, which means that even with neighbourhood goals, local optimisation is much faster than a careful global optimisation.

Unfortunately, practically all forest planning problems have goals that belong to category 3, i.e. they can be evaluated only by considering the combination of stand prescriptions. Therefore, if local optimisation is used in planning, there must be a way to tie the stand-level problems together so that the achievement of the global goals is granted. An elegant way to do this is the method of Hoganson and Rose

(1984), in which the reduced cost is maximised separately for every stand. The reduced cost is computed from:

$$RC_{ij} = c_{ij} - \sum_{f=1}^{F} a_{ijf} v_f \tag{21}$$

where $\mathrm{R}C_{ij}$ is the reduced cost of management alternative i of stand j, c_{ij} is the amount of objective variable in the alternative, a_{ijf} is the amount that alternative i of stand j produces or consumes forest level constraint variable *f, F* is the number of forest level constraints and v_f is the dual price of constraint f. For example, if the objective is to the maximize net present value with the constraint that the total harvest is at least $1000\,\mathrm{m}^3$, such a treatment is first selected for every stand, which maximises the net present value. Then the dual price (v) of the harvest constraint is derived based on the difference between the target drain and the total harvest of the selected schedules, and the stand treatments are selected again using a modified stand-level objective function:

$$RC_{ij} = NPV_{ij} - v \times H_{ij} \tag{22}$$

where NPV_{ij} is the net present value of treatment alternative i of stand j and H_{ij} is the harvested volume in this alternative. Then the total harvest is calculated, and if the target is not reached, the dual price is up-dated and the process repeated until the constraint is met.

Recently, Pukkala et al. (2007) proposed a multi-objective spatial modification of the method of Hoganson and Rose (1984). In this method, the single objective variable (c_{ij}) is replaced by a stand level priority function (or utility function), which may include stand level goals or neighbourhood goals:

$$P_{ij} = \sum_{k=1}^{K} w_k p_k(q_{ijk}) \tag{23}$$

where P_{ij} is the priority of treatment alternative i of stand *j, K* is the number of stand level and neighbourhood objectives, w_k and p_k are the weight and priority function of objective variable k, and q_{ijk} is the amount of objective variable k in treatment alternative i of stand j. The function that is maximised at the stand level is therefore:

$$RC_{ij} = P_{ij} - \sum_{f=1}^{F} a_{ijf} v_f = \sum_{k=1}^{K} w_k p_k(q_{ijk}) - \sum_{f=1}^{F} a_{ijf} v_f \tag{24}$$

where v_l are heuristically up-dated dual prices of forest level constraints.

Although local optimisation is seldom sufficient for solving forest level planning problems, it may quickly produce very good starting points for global optimisation.

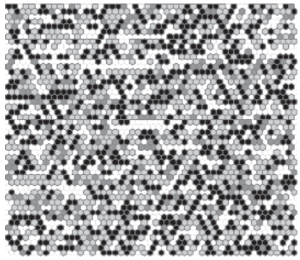
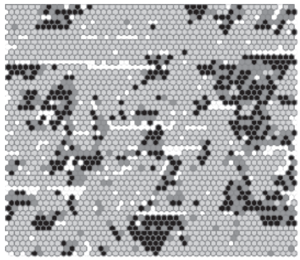
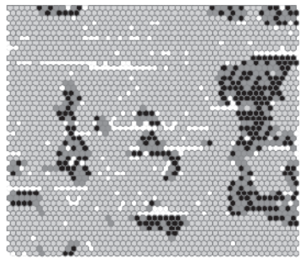
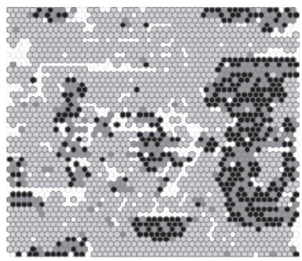
Initial solution: a random treatment schedule has been selected for every stand.	
After one round of local optimisation. The schedule that maximises the stand level objective function has been selected once for every stand. The stand level function maximises NPV with 3 adjacency goals: high number of old-forest–old-forest adjacencies, high cumber of cut–cut adjacencies and a low number of clear-cut–clear-cut adjacencies.	
End of local optimisation: The selection of locally optimal schedule has been repeated until the selected schedules no longer change.	
End of global optimisation: A forest level objective function has been added to the stand level function and the best schedules have been selected again. The weight of the global function has been gradually increased until the global objectives are met.	

Fig. 4 An example of the use of cellular automaton in spatial optimisation in a 2500-ha forest consisting of 2500 1-ha hexagons. The aim is to maximise net present value, aggregate old forests (light grey), aggregate cuttings (dark grey and black), and disperse clear-fellings (black) with the constraint that the 20-year harvest must be at least 170 000 m^3 and the ending volume must be at least 500 000 m^3. The map shows the cuttings of the 3rd 20-year period and the old forests at the end of the 3rd period

The use of local optimisation in landscape planning may notably fasten the whole optimisation process (Fig. 4).

The gain can be particularly significant if raster cells or other fine-grained data are used as calculation units instead of traditional stand compartments. In addition, the optimisation result may be better when local optimisation is used in the beginning of optimisation. This is because local optimisation with neighbourhood functions efficiently finds the natural way of the forest landscape to self-organise. The result may not be feasible in terms of global objectives, but converting it into feasible solution may only need some fine-tuning of the set of locally optimal solutions.

The neighbourhood goals used in spatial local optimisation techniques are often only surrogates of the true management goal. For example, the neighbourhood goal may be to have many neighbours that are similar in terms of treatment or habitat quality (Fig. 5). This kind of simple adjacency goal tends to aggregate resources or treatments but does not measure the "true goal", which might be the total habitat

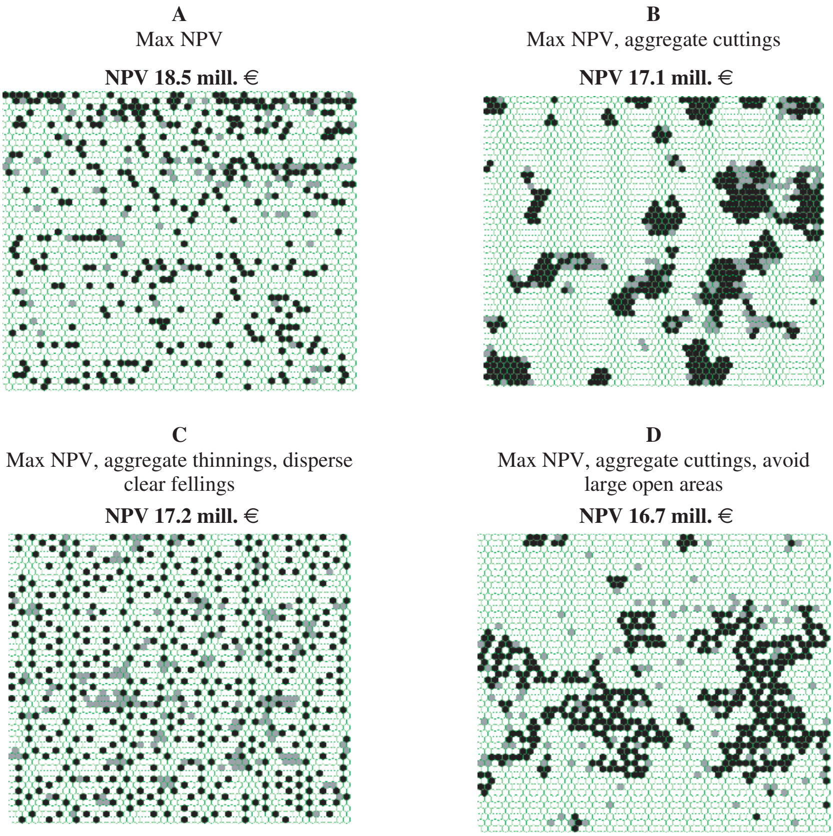


Fig. 5 Examples on the effect of adjacency goals on the spatial distribution of cuttings of a 20-year period in a 2500-ha forest consisting of 2500 1-ha hexagons. Black cells are clear-cuttings and grey cells are thinnings. In all plans the total harvest is constrained between 150 000 and 200 000 m^3

area or core area, or the size distribution of core areas. However, the simple goals based on adjacencies enable fast optimisation and they often correlate strongly with the true goal. In addition, they can be used flexibly to reach very different landscape designs with a small computational effort. Moreover, in many instances it is hard to say what the true spatial goal should be. Can we say for instance that a certain area or size distribution of old forest patches is ecologically better than a high number of old forest adjacencies? For some management objectives all feasible objective variables are surrogates of the eventual goal.

4 Numerical Measures for Non-timber Benefits

4.1 Non-timber Products

Non-timber products such as berries, mushrooms, grass, rattan, fruits, tar, latex and medicinal plants can be measured by empirical yield models. These models typically express the yield as a function of stand and site characteristics. The yields of many non-wood products greatly depend on the weather conditions of the current or previous year. However, the weather conditions of coming years cannot be predicted for the long time horizons used in forest planning. Therefore, annual weather conditions are not useful predictors in long-term forest planning, although they often would improve the predictions. The relevant thing is to relate the yield of the non-timber product to such characteristics which can be controlled in forest management, such as stand density, size distribution of trees, and tree species composition. This enables the planner to examine the effect of alternative management options on the yield of the non-wood product.

There are few empirical models for non-wood products due to lack of systematically collected data. In the absence of empirical data it is possible to construct expert models, which are based on the expertise of people. Such models enable the transfer of expert knowledge to the automated calculation systems of forest planning. An example of expert models for non-timber products are the equations of Ihalainen et al. (2002) for cowberry and bilberry, which are the two most important forest berries of Finland. The models are based on evaluations of 100 stands by 27 experts. The models relate the priority of the stand, as a berry collection place, to site and stand characteristics:

$$\begin{aligned}\ln(v_{\mathrm{B}}) = {} & 0.0062T_{\mathrm{g}} - 0.0136G + 0.0363H_{\mathrm{dom}} + 0.0014V_{\mathrm{p}} \\ & - 0.0013V_{\mathrm{d}} + 0.2393S \qquad (25)\\ \ln(v_{\mathrm{C}}) = {} & 0.0053T_{\mathrm{g}} - 0.0024V_{\mathrm{p}} - 0.0033V_{\mathrm{d}} + 1.6652S \\ & - 0.1673SD_{\mathrm{g}} + 0.005SD_{\mathrm{g}}^{2} \qquad (26)\end{aligned}$$

where v_{B} is the priority of the stand in terms of bilberry yield, v_{C} the priority in terms of cowberry yield, T_{g} the mean age of trees (years), G the stand basal area ($\mathrm{m^2ha^{-1}}$), H_{dom} the dominant height (m), V_{p} the standing volume of pine ($\mathrm{m^3ha^{-1}}$),

V_d the standing volume of deciduous trees, D_g the mean diameter of trees (cm), and S is a dummy variable which is equal to one if the forest site type is *Vaccinium* type (rather poor) or poorer, and zero otherwise.

If there are enough yield measurements in different stand conditions, an empirical model can be developed. An example is the recent model of Bonet et al. (2007) for the mushroom yield of Catalonian pine stands:

$$\ln(y) = -4.329 + 1.966 \ln(G) - 0.118G + 0.636 \cos(Asp) + 0.00331 Alt \quad (27)$$

where y is the fresh mass of edible mushrooms (kg/ha), G is stand basal area ($\mathrm{m^2 ha^{-1}}$), *Asp* is aspect (rad) and *Alt* is elevation (m a.s.l.). According to the equation, mushroom yields are the highest when stand basal area is 15–20 $\mathrm{m^2 ha^{-1}}$. There are more mushrooms at high elevations and on northern aspects.

4.2 Recreation Amenities

Many forest amenities are related to the scenic beauty of forest stands. When people spend time in the forest, within-forest beauty is important rather that distant vistas. An example of a stand-level scenic beauty model is the following equation (Pukkala et al. 1988), which is based on the evaluation of 100 stands by 122 people.

$$\begin{aligned} \mathrm{SB} = 4.471 &+ 0.06450 d_{\mathrm{g}} - 0.0001745 N \\ &+ 0.006439 V_{\mathrm{Pine}} D + 0.005733 V_{\mathrm{BirchAspen}} D \end{aligned} \quad (28)$$

where d_g is the basal-area-weighted mean diameter of trees (cm), N is the number of trees per hectare, V_{Pine} is the volume of pine ($\mathrm{m^3 ha^{-1}}$) and $V_{\mathrm{BirchAspen}}$ is the volume of birch and aspen ($\mathrm{m^3 ha^{-1}}$). D is a dummy variable, which is equal to one if the stand dominant height is 10 m or more, and zero otherwise. The function predicts that sparse stands of large trees, preferably pines and birches, are experienced as beautiful.

The scenic beauty of the whole forest can be measured with the mean scenic beauty index of stands. A more advanced way is to take the accessibility or visibility of stands into account, and use e.g. the visibility- and area-weighted mean scenic beauty index of stands as the objective variable (Fig. 6).

Visibility and accessibility analyses can be calculated in geographic information systems, and the weights can be imported to the optimisation routine. If the location-weights are constant, i.e. they only depend on the terrain model and distances to viewpoints or entry points and not on controllable stand characteristics, optimisation with location-weights is non-spatial and can be done e.g. with simple linear programming.

Several visualisation tools have been developed to evaluate the visual outlook of future forest landscapes (Fig. 7). These tools are good for evaluating candidate plans. However, they cannot be used in optimisation, which requires numerical measures for forest services.

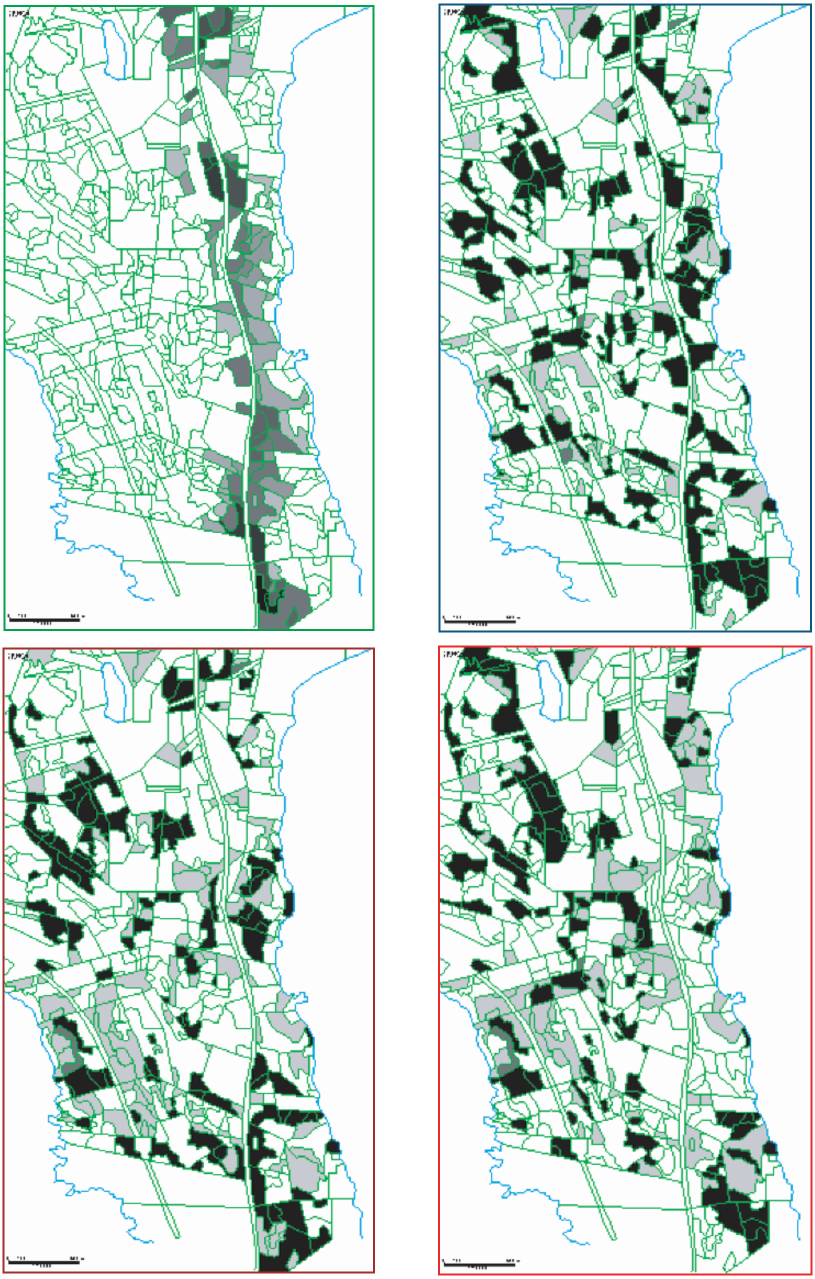


Fig. 6 Top left: Stands' visibility from the road (dark colour implies high visibility). The other maps show the cutting areas. Black is clear-felling, dark grey removal of over-story trees and light grey a thinning treatment. Top right: no scenic beauty objective; bottom left: mean scenic beauty index of stands used as an objective; bottom right: location-weighted mean scenic beauty index of stands used as an objective. The total harvest is the same in all plans (Pukkala 2002)

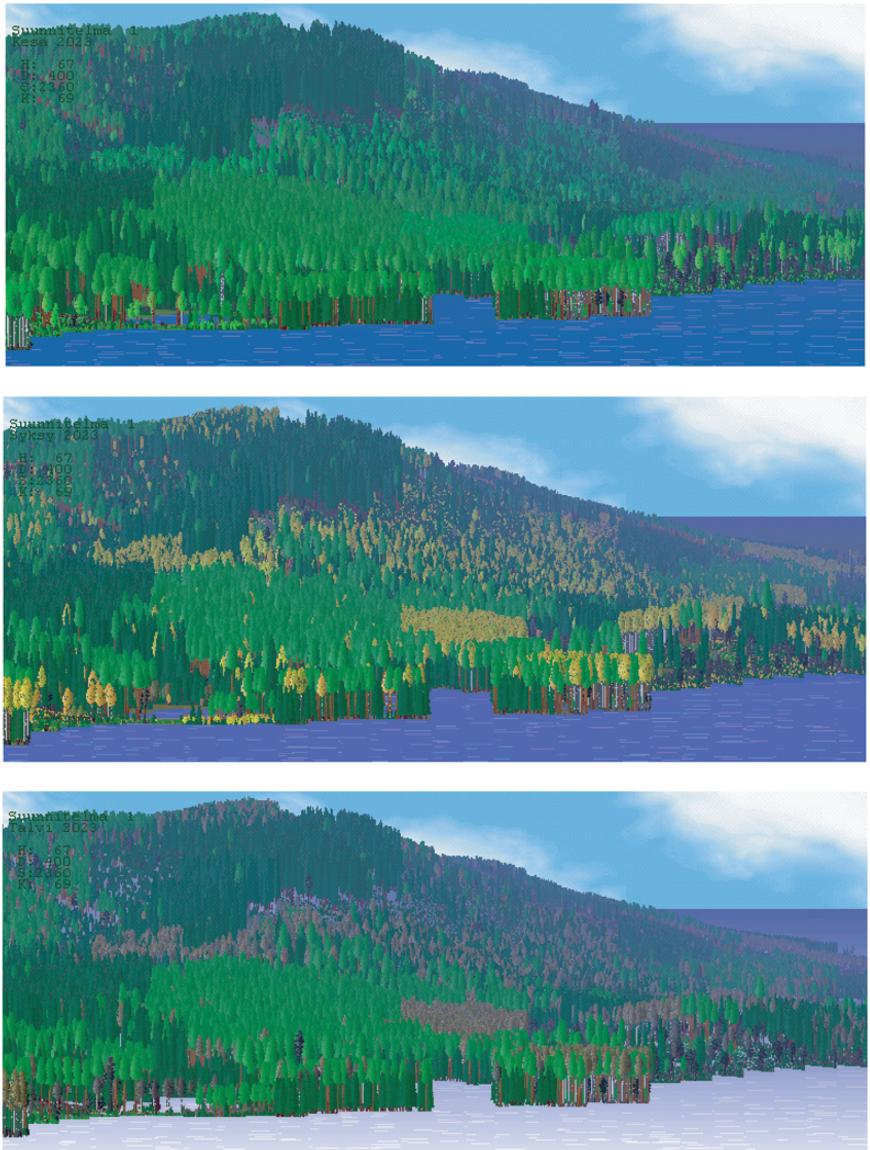


Fig. 7 Visualisation of different seasons of the year in the Monsu planning software

4.3 Hunting and Game

Hunting possibilities can be described with empirical or expert models for the population density of the game species, or suitability of the forest as a habitat for the species. The habitat suitability models may be stand level models or forest level models. In the former case, the habitat suitability indices calculated for the management alternatives of stands are used to calculate spatial or non-spatial landscape metrics that describe how the habitat is distributed in the landscape (Fig. 8).

An example of a forest level habitat suitability model is the following formula for black grouse (Kangas et al. 1993):

$$HSI_{\mathrm{BGi}} = -9.99 + 0.946 \ln(Birch) + 1.44 \ln(Height_{5\text{–}15}) + 0.023 \ln(Pine) \tag{29}$$

where *Birch* is the proportion of birch of the total growing stock volume, $Height_{5\text{–}15}$ is the share of stands (proportion of area) with mean tree height between 5 and 15 m, and *Pine* is the share of stands in which the proportion of pine is as least 40% of growing stock volume. According to the equation, black grouse favours forests in which there is much birch, but on the other side plenty of pine-dominated stands

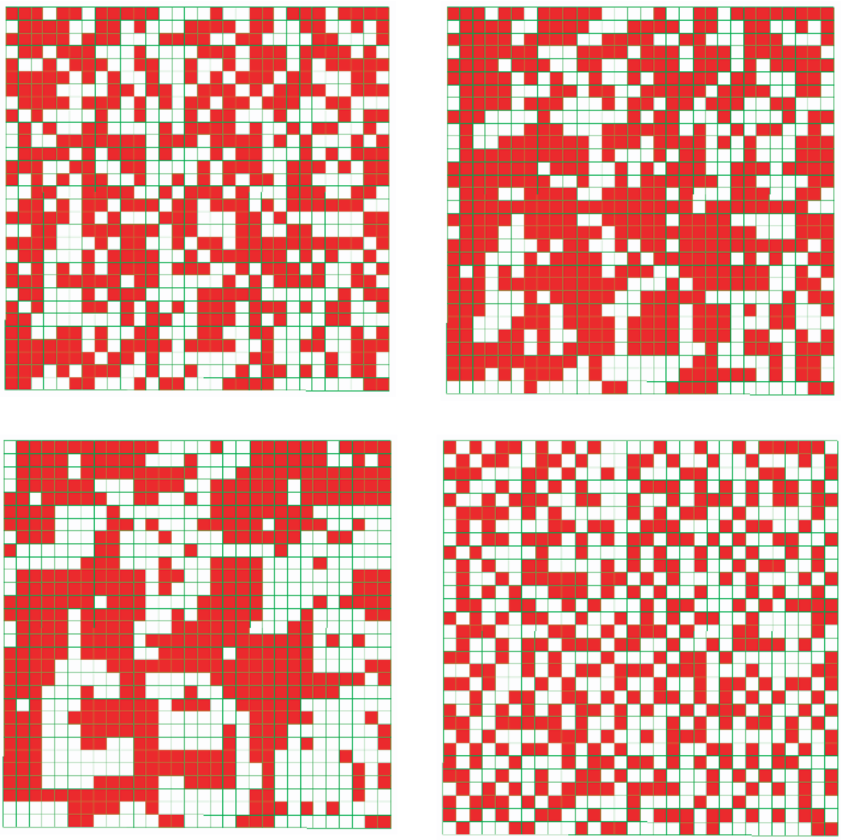


Fig. 8 Effect of the landscape metric on the spatial distribution of habitat patches (dark cells, HSI > 0.5) of capercaillie at the end of a 60-year planning period in an imaginary forest landscape consisting of 900 square-shaped stands. The maximised landscape metric is: top left, mean habitat suitability index (HSI); top right, habitat–habitat boundary; bottom left, spatial autocorrelation of HSI; bottom right, habitat–non-habitat boundary (Palahí et al. 2004)

and young stands. The model is based on the evaluations of 10 forested landscaped by 15 experts.

4.4 Biodiversity

The principal ecological service of the forest is to maintain biological diversity. Two main lines have emerged for evaluating the quality of forest in terms of biodiversity maintenance: the ecosystem approach and the species approach. The idea of the ecosystem approach is to measure the number of different structural elements present in the forest. This can be done at the stand level or the forest level. At the stand level, different tree species, deadwood components, canopy layers etc., are different structural elements. At the forest level, age classes, stand structures, species compositions, stand densities etc. can be taken as structural elements. Several indices can be calculated for both levels that measure the structural diversity with a single variable. One way to calculate such an index is the following formula

$$D = \sum_{k=1}^{K} w_k p_k(q_k) \tag{30}$$

where w_k is the weight, p_k a priority function and q_k the quantity of element k. In forest level measurement the quantity may be the area of stand type k, and in stand level measurement it could be the volume or biomass of species, size class or deadwood component k. Functions p_k allow one to take into account the decreasing marginal contribution of an element. For example, one cubic meter of spruce down wood may increase structural diversity fast when spruce down wood is scanty, but much slower when spruce down wood is abundant. The rationale behind the ecosystem approach is that different structural elements are habitats for different species, i.e., many structural elements imply the presence of many species.

The species approach evaluates the suitability of the forest as a habitat for individual species or species groups. Kurttila et al. (2002) proposed the following habitat suitability index for flying squirrel:

$$HSI = \prod_{k=1}^{K} [p_k(q_k)]^{1/K} \tag{31}$$

where K is the number of variables that affect the index value, p_k is a priority function for variable k and q_k is the quantity of variable k. Functions p_k express how much different amounts of the variables contribute to the index. The variables of the model are growing stock volume, proportion of spruce, proportion of deciduous trees, mean diameter of deciduous trees and volume of standing dead trees. A good habitat has a medium or high volume, at least 50 % of spruce, at least 30% of deciduous trees of large size, and at least $3\,\mathrm{m}^3/\mathrm{ha}$ of standing dead trees. Figure 9

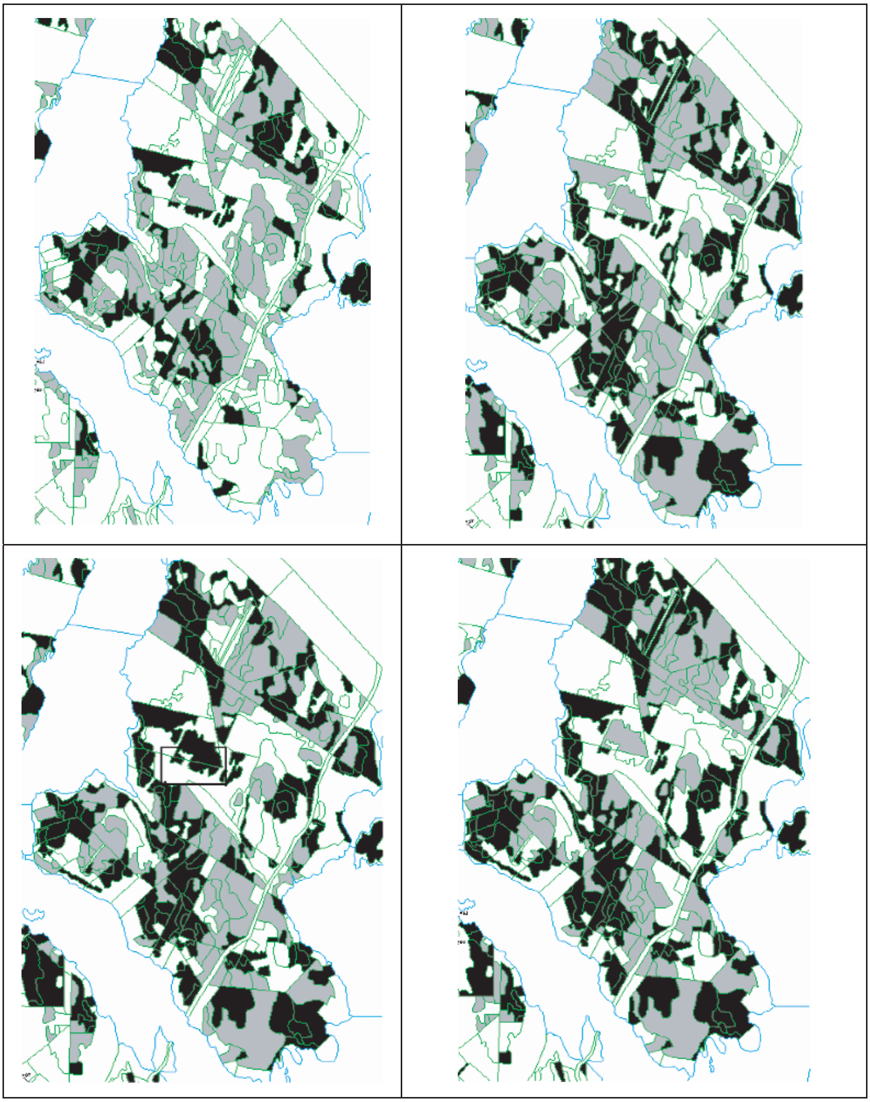


Fig. 9 Habitat suitability indices (HSI) of flying squirrel initially and in three different forest plans at the end of a 60-year planning period. Black indicates a HSI value of 0.4 or more, grey HSI values between 0.01 and 0.4 and white a HSI value of 0.0 or a non-forest area. Top left: initial situation; top right: combination of holding plans (objective function corresponds to Equation 16); Bottom left: integrated holding and area plan (objective function corresponds to Equation 17); Bottom right: area plan (objective function corresponds to Equation 9). The rectangle drawn by thick black line in the integrated plan shows an example of across-holding-coordinated location of flying squirrel habitat patches

shows examples of the use of this formula and utility functions that correspond to Equations 16 (Holding plans), 17 (Integrated plan) and 8 (Area plan).

A good rule of thumb for calculating the habitat suitability index (*HSI*) for a certain species is to divide the relevant features into environment (e.g. shade provided by trees) and resource factors (e.g. large stems of dead aspen) and use these features to calculate sub-indices for the two factors.

The *HSI* is then computed from:

$$HSI = (Environmentsub\text{-}index \times Resourcesub\text{-}index)^{1/2} \tag{32}$$

If temporal continuity (habitat maintained in the same place) is important, the index may be computed from:

$$HSI_t = (Environment\text{-}sub\text{-}index_t \times Resource\text{-}sub\text{-}index_t \times HSI_{t-1})^{1/3} \tag{33}$$

Figure 10 shows an example in which the HSIs of five different threatened species are used in the planning model. The aim was to have at least 500 hectares of suitable habitat (HSI > threshold) for each species by the end of the 3rd 10-year time period. This goal was achieved for four species.

The habitat suitability indices calculated for the stands can be used in many ways to form forest-level variables for planning models. Various landscape metrics may be computed from the stand-level indices. Landscape metrics are variables that measure the sizes, shapes, relative arrangement and connectivity of habitat patches (or just stands with certain features) as well as their total area (McGarical and Marks 1995). The use of landscape metrics sometimes requires that the stands be classified into suitable or unsuitable ones (or good and bad stands). Some landscape metrics, like the mean or spatial autocorrelation, do not require bisecting stands into good and bad ones. Some examples of landscape metrics, which can be calculated reasonably easily during optimisation, are the following ones:

- Non-spatial
  - Area or percentage of suitable stands (index > threshold)
  - Mean index value
  - Location-weighted mean of the index
- Spatial
  - Spatial autocorrelation of the index (Moran's I)
  - Core area (habitat area minus the area of edge zone)
  - Percentage of good–good boundary (e.g. habitat–habitat boundary)
  - Percentage of good–bad (e.g. habitat–non-habitat boundary)
  - Mean difference in the index value between neighbouring stands.

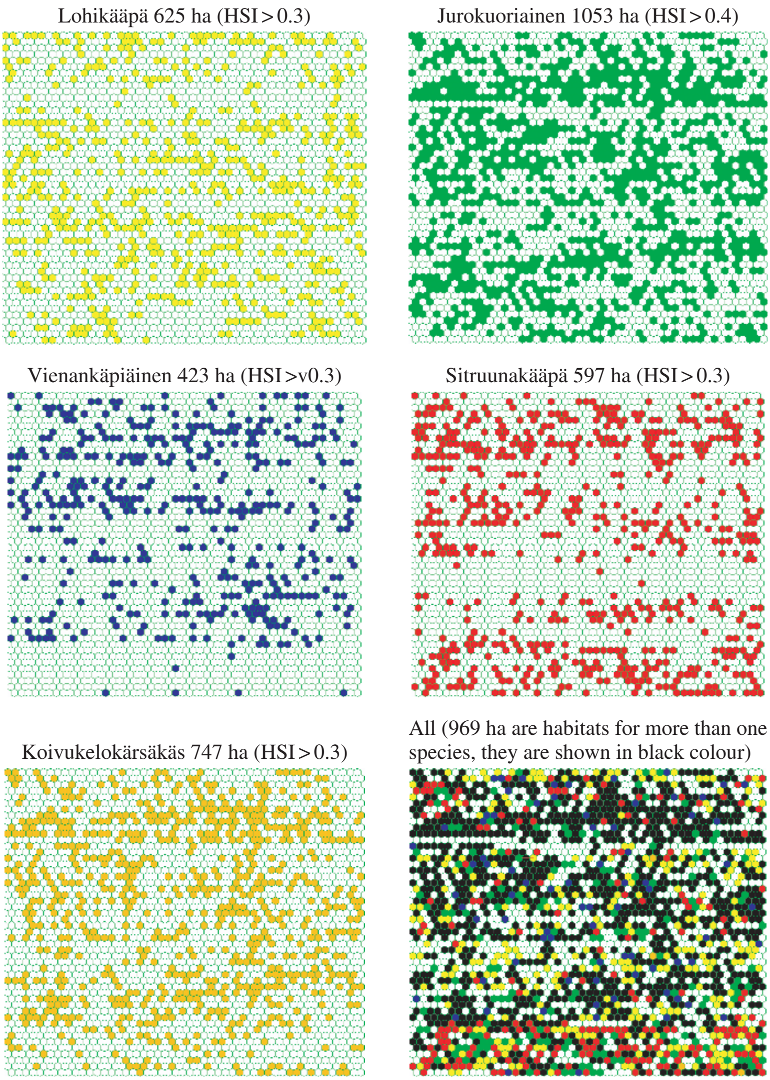


Fig. 10 Habitats (non-white cells) of five different threatened species in a plan where the aim is to have at least 500 hectares of habitat for every species (the goal was not achieved for Vienankäpiäinen). The map on bottom right shows that 969 out of 2500 1-ha stands are suitable habitats for more than one species

4.5 Environment

Examples of environmental services of forest are carbon sequestration, erosion control, and regulation of water resources. Carbon sequestration by forests has recently become topical because it has been realised that forestry is a potentially powerful way to slow-down climate change. The importance of this function of the forest is increasing due to international agreements and the fact that carbon emissions are becoming more and more tradable. Therefore, a good forest planning system of today should be able to calculate the carbon balance of a candidate plan so that its environmental impacts can be evaluated before implementing the plan.

The carbon balance of a forest is the sum of carbon balances of stands. The carbon balance of the stand during a certain time period can be calculated from the following components: (1) change of biomass, (2) biomass of trees harvested during the period, (3) biomass of trees died during the period, (4) decomposition of dead trees present in the stand, and (5) decomposition of trees harvested from the stand (including roots, cutting residues and removed timber assortments). Although sub-process 5 partly happens outside the forest (cutting residues decompose in the forest but products outside the forest) it can be included in the carbon balance of the forest. If this is done, it is possible to take into account also the decomposition of products that have been prepared before the start of the planning period. In the same way, if decomposition of dead wood is included in the carbon balance, the deadwood and its state of decomposition present in the stand in the beginning of the planning period should be included in the balance.

The carbon balance of period $t1$–$t2$ can be calculated from the following formula (Diaz-Balteiro and Romero 2003)

$$C_{t1,t2} = a(B_{t2} - B_{t1} + H_{t1,t2}) - P_{t1,t2} \quad (34)$$

where $C_{t1,t2}$ is the carbon balance of period $t1$–$t2$, a is the proportion of carbon in biomass (usually around 0.5), B is biomass, $H_{t1,t2}$ biomass of trees harvested or dead during period $t1$–$t2$ and $P_{t1,t2}$ is the amount of carbon released in the decomposition of dead biomass and products during period $t1$–$t2$.

A careful calculation of the carbon balance needs information about the deadwood present in the stand in the beginning of the planning period, as well as on the decomposition of products that were prepared from timber harvested from the forest before the planning period. However, there are also simpler options. The following alternatives may be considered:

1. Only the change of living biomass is included in the carbon balance.
2. Change of living biomass and natural mortality are included. This requires the estimation of the initial mass of deadwood originating from natural mortality.
3. Change in living biomass, natural mortality and cutting residues are included. This requires the estimation of the initial deadwood originating from natural mortality and cuttings.
4. Change in living biomass, natural mortality, cutting residues and products are included. This requires the estimation of the initial deadwood originating from natural mortality and cuttings. In addition, the amount and stage of decomposition of products prepared before the planning period should be predicted.

Table 1 is and example of these balances in a 2500-ha forest in Finland, calculated using the Monsu software (Pukkala 2004). The harvested volumes of the three 10-year periods are equal ($98000\,\mathrm{m}^3/10$ years) and such that the ending growing

Table 1 Carbon balances of three successive 10-year periods for a 2500-ha forest for a plan in which the harvested volumes of 10-year periods are equal, and ending growing stock volume is equal to initial volume. The amounts of initial deadwood and cutting residuals (including stumps and roots) have been predicted with models. Decomposition of products does not include products prepared before the planning period

Component of carbon balance	10-year period		
	I	II	III
Balance 1: biomass change, tn	3600	1055	−795
+ New dead trees, tn	4501	3606	3518
− Decomposition of dead trees, tn	−3215	−3536	−3219
= **Balance 2: biomass + mortality, tn**	4886	1125	−496
+ New cutting residues, tn	13685	13124	13469
− Decomposition of cutting residues, tn	−24432	−19547	−18943
= **Balance 3: biomass + mortality + residues, tn**	−5861	−5298	−5970
+ New timber, tn	24305	24581	24585
− Decomposition of products, tn	−12950	−19601	−21038
= **Balance 4: biomass + mortality + residues + products, tn**	5495	−317	−2423

stock volume is equal to the initial volume. All balances, except the fourth one are non-biased in the sense that no relevant components of the balance are ignored. Balance 4 is overestimate, most for the first periods, because the decomposition of products prepared before the planning period is ignored. In balances 2, 3 and 4 the initial amounts of deadwood and/or cutting residues (including stumps and roots) are calculated with models, and these balances may therefore contain prediction error.

The balances of Table 1 are shown as per hectare and year values in Fig. 11. The figure shows that the carbon balances in an ordinary management of previously forested landscapes do not differ much from zero. The best ways to have clearly positive balances are to increase biomass and to produce timber for products that have longer life spans than previously produced (e.g., saw logs instead of pulpwood).

4.6 Risks

Various environmental hazards may also be relevant in forest planning. For instance, reducing the forest's vulnerability to fire may be an important management goal. Two approaches are common to solving problems with risk. The first one is to use stochastic programming, which however may be complicated in forest level planning. The other approach is to treat the risks as known quantities (indices) which are treated in the same (deterministic) way as the other objective and constraining variables. González et al. (2006) developed the following empirical model for the 12-year probability of fire occurrence in the stand:

$$P_{fire} = \frac{1}{1 + e^{\left(-1.925 - 2.256\ln(\max\{Ele-7,\ 0\}+1) - 0.015Dg + 0.012G - 1.763P_{hard} + 2.081\left(\frac{s_d}{Dg+0.01}\right)\right)}} \tag{35}$$

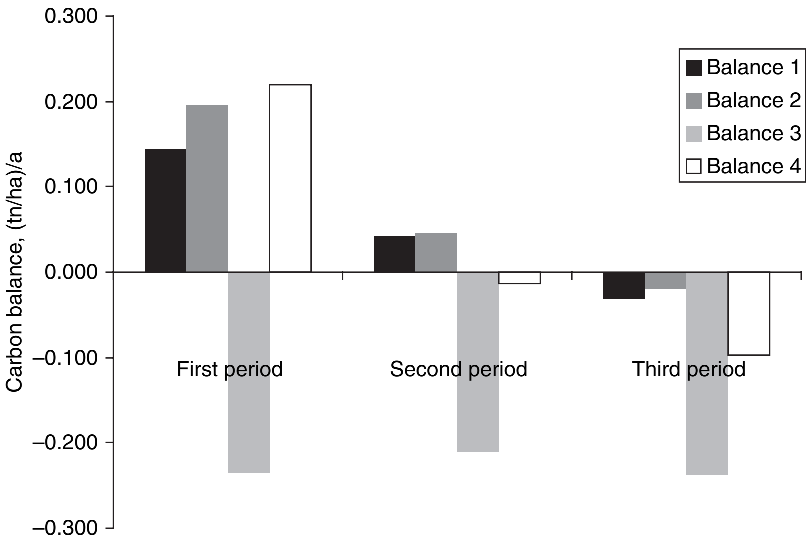


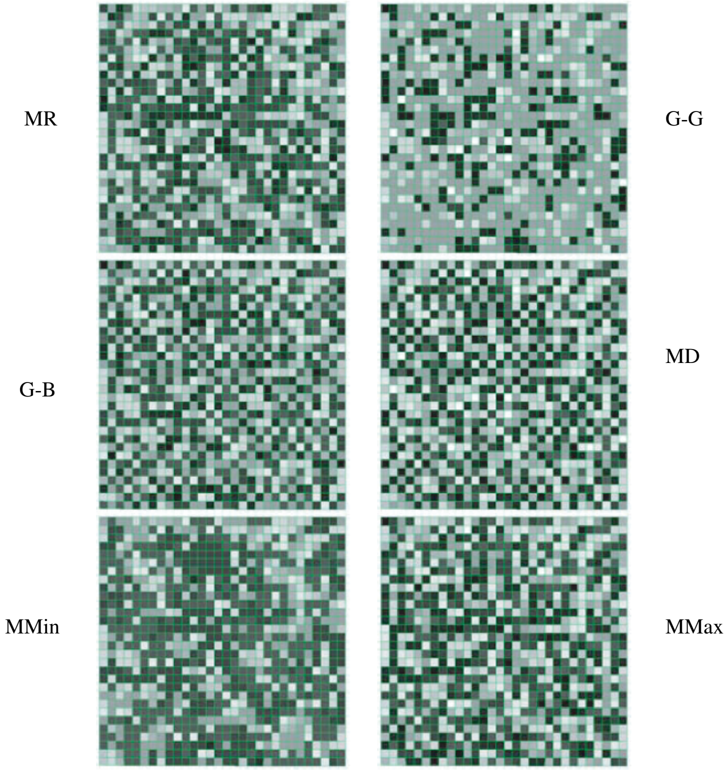
Fig. 11 Carbon balance calculated in four different ways for three 10-year periods in a 2500-forest when cuttings are equal to volume growth (see Table 1)

where P_{fire} is the 12-year probability of fire occurrence, *Ele* is elevation (in hundreds of meters), *Dg* is the basal-area-weighted mean diameter (cm) of trees, *G* is the total basal area $(\mathrm{m^2ha^{-1}})$, P_{hard} is the proportion of hardwood species of the number of trees, and s_d is the standard deviation of dbh (cm). The last predictor expresses the relative variability of tree diameters. The model is based on systematic fire records and the plots of the national forest inventory of Catalonia. According to the model, the higher the elevation the lower is the probability of fire. Furthermore, forest stands with high values of *G* and s_d have a higher probability of fire while stands with high values of P_{hard} and *Dg* have a lower probability of fire.

Another model was developed by González et al. (2007) which predicts the degree of damage if a fire passes the stand:

$$\begin{aligned} y = & -6.131 - 0.329G + 0.060Slope + 2.266Pine \\ & + 4.319\left(\frac{G}{D_q + 0.01}\right) + 6.718\left(\frac{s_d}{D_q + 0.01}\right) \end{aligned} \tag{36}$$

P_{dead} is the proportion of dead trees, *G* is the stand basal area $(\mathrm{m^2ha^{-1}})$, *Slope* is the percentage of altitude change per distance (%), *Pine* is a dummy variable which equals 1 if the stand is dominated by pines (>50% of basal area is pine) and 0 otherwise, s_d is the standard deviation of dbh (cm) and D_q is the quadratic mean diameter (cm) of trees $(D_q = \sqrt{(40000/\pi \times G/N})$. According to the model, the relative damage decreases when stand basal area increases. Higher values of



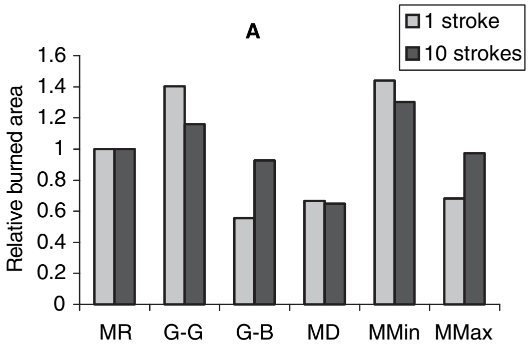


Fig. 12 Effect of the use of landscape metrics as objective variables on the spatial distribution fire-risky (dark shades) and fire-resistant (light shades) stands when the mean fire resistance index is the same in all plans. The landscape metrics have been calculated from fire risk indices of stands. The lower sub-figure shows the mean burned area in landscapes obtained with different metrics when 1 or 10 strokes of lightning hit random stands. The burned area is based on 1000 repeated simulations with a cellular automaton that simulates the spread of fire in the landscape. Landscape metrics that create discontinuity in fire sensitivity (G-B and MD) decrease the expected burned area by about 40% as compared to the landscape that was obtained without spatial goals (MR)

$G/(D_q + 0.01)$ and $s_d/(D_q + 0.01)$ increase the damage. The other two factors that contribute to a high fire damage are steep slopes and pine dominance.

These two models may be used separately (Fig. 12) or together ("loss index" = predicted fire probability × predicted degree of damage) in forest planning calculations when the aim is to manage the forest so as to minimise losses due to fire.

Most wind throws occur at newly created stand edges. If a stand is adjacent to a gap, for instance a clear-felling area, trees near the stand edge are easily wind thrown. Different stand types are differently sensitive to wind damage. For example, broadleaf trees, young stands or stands consisting of short and quickly tapering trees are usually less sensitive than conifer stands of tall and slender trees (Fig. 13).

There are models than can be used to calculate the so-called critical wind speed for the stand (Peltola et al. 1999). The critical speed is the smallest wind velocity that is likely to fell or break edge trees of the stand. Those edges for which the critical

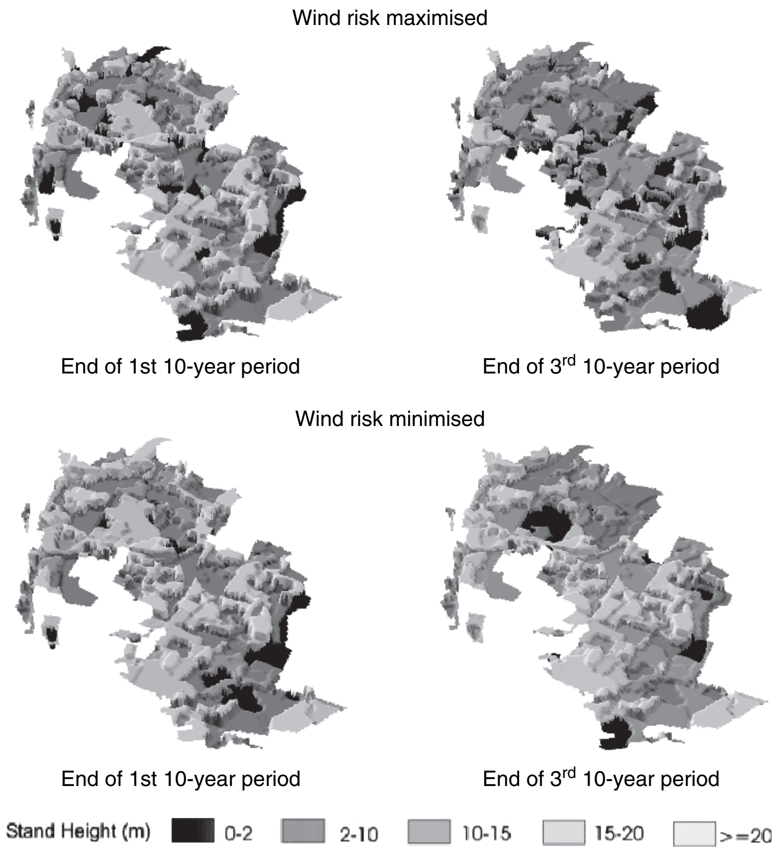


Fig. 13 Tree height of the forest when the risk of wind damage (percentage of vulnerable edge) is maximised or minimised. When wind risk is maximized the length of vulnerable edge (critical wind speed < 20 m) is 6 (end of 1st period) to 20 (end of 3rd period) times greater than when wind risk is minimised (Zeng et al. 2007b)

wind speed is “low” (wind speed that occurs commonly) can be termed as vulnerable edges. Minimising the length of vulnerable edge is a simple neighbourhood related management objective. The objective variable, which is minimised in planning, can be the length of such a stand boundary in which the critical wind speed for one neighbour is less than (say) 20 m/s, and the tree height of the other stand is less than (say) 2 m. Minimising this variable leads to forest landscapes in which common wind speeds do not cause much loss (Fig. 13).

5 Conclusions

This review shows that many methods exist for the numerical planning of multi-functional forestry. The discussed methods are immediately available for practical use. Therefore, lack of methods is not the bottleneck for not integrating non-timber benefits into forest planning calculations.

However, there is lack of empirical data and even expert knowledge on the relationships between controllable growing stock characteristics and the non-timber features of stands and forests. For instance, although mushroom collection is a very important leisure activity in many countries, there is little information on the effect of growing stock characteristics and management actions on the mushroom yields. Also the habitat requirements of many forest-dwelling species are poorly known.

The measures and indices discussed in this article may be based–besides empirical measurements – on expert knowledge. Modelling expert knowledge is a way to make expert knowledge cheap and portable. This will greatly enhance its use in forestry practice. However, models are only partial descriptions of complicated relationships, which means that subjective evaluation, by forest owners, planning consultants or experts, is still required. Quantitative calculations offer partial but valuable information for decision-making, which is always subjective and eventually based on subjective choices.

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Decentralized Forest Planning Models – a Cellular Automata Framework

Anne-Hélène Mathey and John Nelson

1 Introduction

Natural resources in general, and forest resources in particular, are complex systems with environmental, social and economic roles, and one can conceptualize them as the aggregation of natural and human capitals. These categories are not meant to be restrictive and they overlap; for example, natural resources constitute both natural capital with regard to ecosystem functioning and human capital with regard the production and extraction of goods and services. The realization that natural resources are complex systems with overlaps among their components has two main implications: first, all components of natural resources are interconnected. If the forest soil is depleted, it is very unlikely that tree growth will be satisfactory for timber exploitation purposes. This inference has brought about such paradigms as integrated forest management and ecosystem management. Second, because the current forest management paradigm strives to include a range of environmental, social and economic services beyond timber products, decision support tools are needed to provide the decision maker with the information required to assess the sustainability of any given forest management strategy with respect to these services. The main requirements for planning tools are:

(1) Planning tools must address the management requirements for conservation and protected areas at multiple temporal and spatial scales.
(2) Planning tools must be amenable to modifications of value systems and localized alterations resulting from interactions with various stakeholders in the planning process.
(3) Planning tools must assess economic costs and benefits from a wider perspective than those provided by assuming fixed prices, fixed markets and a focus on timber products.

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The practical repercussion of these three requirements is that planning tools need to be able to accommodate spatial objectives and assess the forest under different management strategies and changing conditions over the planning horizon (e.g. natural processes such as succession, natural disturbance or climate change).

The objective of this chapter is to describe and demonstrate a decentralized approach to forest planning problems that have spatial-temporal objectives at multiple scales. Specifically, we use a planning frame work based on co-evolutionary cellular automata to demonstrate the potential of this method to simultaneously meet local, stand neighborhood objectives and global forest-level objectives. We begin by describing the advantages and disadvantages of the traditional centralized planning approach that is based on metaheursitics. We then introduce the decentralized approach based on cellular automata as an alternative, and describe the general procedures of this method in the context of forest planning. Next, the method is applied to a case study and results are presented that demonstrate its effectiveness in meeting both local and global objectives. Finally, we present conclusions and recommendations for further research in decentralized planning.

2 The Centralized Approach to Forest Planning

Considerable progress has been made in spatial forest planning since the introduction of metaheuristics, such as simulated annealing, tabu search, genetic algorithms and various combinations of these evolution-based programs (Lockwood and Moore, 1993; Baskent and Keles, 2005; Bettinger and Chung, 2004; Ohman, 2000). Application of these methods has taken a “top-down” or centralized approach, where random perturbations are made and assessed according to their contribution to global objectives. Common examples of global objectives are bounds on harvest flows and net revenue, lower bounds on the amount of old forest, upper bounds on disturbance rates and patch size distribution targets. In large forest estate models, variations of these objectives are often specified for sub-compartments within the forest, leading to numerous objectives that need to be incorporated into the objective function. Each objective needs a target, a penalty function to measure deviation from the target and a corresponding weight in order to compute the objective function (Nelson, 2003).

While the metaheuristics are very good at generating strong solutions to these global objectives, they are not particularly good at simultaneously generating strong solutions for the local objectives. Common examples of local objectives are clustering, connectivity and continuity of specific stand types or attributes that result from application of specific treatments. Another example is where certain treatments (e.g. partial cutting) are only appropriate based on the proximity of the stand to roads or perhaps waterways. We may be fortunate and achieve objectives at both scales, but there is no explicit link in the centralized approach that promotes meeting both the local objectives and the global objectives. Therefore, even though we may achieve a desirable solution, it is difficult to identify a strategy that managers can implement in hopes of reaching the desired outcome. If local objectives are not satisfied (but global objectives are) then an iterative process is needed whereby the solution is locally constrained (e.g. a cluster of stands is reserved from harvest) and the model re-run. As more objectives are added to the centralized model, it becomes less responsive to meeting all the objectives simultaneously, and considerable manipulation of the objective weights is needed to provide direction.

3 Decentralized Planning as an Alternative Framework

The essential decision in forest management is which management regime to apply to a stand. One of the principal drivers of this decision is the suitability (and feasibility) of the stand for different silvicultural alternatives. Suitability is generally evaluated on the basis of stand productivity (volume-based or revenue-based) and is therefore a function of the stand characteristics alone. The operability of the stand, its age, density and species composition as well as site quality or riparian and wetland features will determine what type management can be applied to the stand, making it possible to forecast stand attributes into the future.

In the forest level context, the decision concerning a single stand seeks to identify the course of action that will maximize the stand's contribution to the overall management strategy. When the management goals are non-spatial such as maximizing long-term timber extraction, the solution is given by a particular combination of treatments through time for each stand on the management unit. The potential contribution of each stand to management objectives is therefore a function of its attributes at the time of decision making. Centralized planning approaches are particularly efficient at solving this type of problem.

However, part of the contribution of stands to the management objectives is born from their spatial relationship to other stands. For instance, the choice of protecting a stand in old growth is potentially more valuable for habitat conservation if the stand then becomes part of a larger cluster of protected stands than if the stand is a small isolated area. Similarly, a stand scheduled to be harvested simultaneously with other adjacent stands can lead to economies of scale. With the introduction of spatial objectives in forest management, decisions regarding two nearby stands can affect each other's contribution to the global objective more than decisions taken at distant locations (Strange et al., 2001; Hoganson and Borges, 1998; Hoganson et al., 1998). In such cases, decentralized planning is very effective at evaluating the contribution of stand management to global objectives because a large component of the contribution can be assessed at the neighborhood level. Such an approach builds information from the lower levels (i.e., individual stands and their neighborhoods) and has the advantage of directly addressing local conditions and constraints (stand dynamics, adjacency requirements, topology, etc.).

A dichotomy between stand- and forest-wide improvements arises when global constraints are present. Such is the case when a minimum amount of conserved area is to be maintained in all periods of the planning horizon or when the timber harvest flow is required to fall within minimum and maximum bounds. Global constraints require some stands to choose management regimes which are suboptimal from their local perspective but necessary to meet forest-wide objectives. In order to achieve an acceptable level of the global objective in a decentralized planning framework, the stands, which are akin to independent decision makers are forced to cooperate to achieve common or independent goals. In decentralized models, the incorporation of guiding rules is needed to ensure that not only local (stand-level) objectives, but also global (landscape-level) goals are met.

4 Case Study on Decentralized Planning Applied to Forest Management

In this section we present a decentralized planning framework for forest management. This framework is based on Cellular automata (CA) modeling, a tool capable of representing discrete dynamic systems whose behaviour is specified in terms of local relations. Essentially, a cellular automaton is an abstract machine with a state selected from a finite set of states and a set of transition rules. Applied at discrete time intervals, they determine the automaton's subsequent states based on the cell's own state and the states of its neighbors (Toffoli and Margolus, 1987). CA models are generally constructed as a collection of cells that form a lattice. The decentralized transition rules and the spatial interdependence of the lattice cells account for CA's self-organization and scale-integration capabilities. CA models have been used extensively for process simulation in the exploration of complex systems in physics, geography and biology (Hogeweg, 1988; Green, 1989) and have further been used to generate the spatial arrangement of outputs from aspatial planning models (White and Engelen, 1997, 2000).

In a forest planning context; the forest is defined as a grid of cells that represent individual CA stands; CA states represent management schedules and the CA transition rules are decision-making models (e.g. optimization functions). The neighborhood influencing a single stand decision consists of the eight adjacent stands. Each stand makes the decision concerning its own management based on its characteristics and on those of its neighbors. A plan that fulfills global objectives will be developed eventually if individual stand decisions benefit the forest as a whole.

4.1 Computation: Reconciling Local Decisions with Global Objectives

4.1.1 Generate Local Objective Value

Each iteration of CA typically represents a time step and the transition rules allow a cell to move from its current state to the next state. However, a single call to the cell-based decision-making model (the stand management schedule optimization function) does not ensure the best course of action from the forest perspective because it will unlikely result in the ideal spatial arrangement of management activities for the global objectives. Therefore, an iterative co-evolution of stand decisions is needed to provide the opportunity to arrange management activities so they best fulfill spatial objectives. Mathey et al. (2007) designed a CA-based algorithm to foster co-evolution among local decisions. It is based on spatial evolutionary game theory where the decisions of individual players are most influenced by players in close vicinity. An asynchronous updating procedure is applied to allow stands to make decisions successively and to react to previous decisions made by their neighbors. In the course of the iterative process, the cells which have an obvious "best" state will identify it quickly and will influence the surrounding cells in choosing their own management state.

4.1.2 Comply with Global Constraints

The forest plan resulting from the co-evolutionary decision process of cells may still digress from an optimum forest-wide configuration when the planning problem is subject to global constraints. This situation arises if all stands were to co-evolve a solution that seeks to generate harvest revenues and large old growth core area and there exists bounds on the timber harvest flow. The challenge in a decentralized framework is to identify which stands should modify their management regime in order for the final plan to meet minimum and maximum harvest levels.

There is a need to define "penalty functions" that are efficient surrogates for the *global* constraints within the *local* optimization process. In the CA model, violations of global level constraints are incorporated as penalties or incentives that influence cell level choices. These measures are used to modify the stand level objective function values thus encouraging or discouraging local level decisions so as to better meet the global constraints. The incentive and penalty rates augment the value of a management regime/schedule in proportion to the amount that it decreases a constraint violation. The inclusion of the incentive/penalty parameters can be seen to be equivalent to establishing period-dependent weights for each objective. From this perspective, a specific set of penalty rates can be considered to represent a global policy choice and represent a new expression of the relative values of each objective. Although the cells make local level decisions, their decisions influence their neighbors' decisions and in turn their neighbors' neighbors. The CA requires a number of iterations to co-evolve a configuration of treatment schedules that represent an effective global solution for any given set of penalty parameters. If penalty parameters change too often, stands do not have the opportunity to properly co-evolve a forest plan. On the other hand, if penalty parameters change rarely, it is hard to find a feasible solution. Incentive/penalty multipliers are therefore reevaluated periodically throughout the iterative process to progressively guide stands towards transitions that minimize global constraint violations.

Following this scheme, all stands interact and co-evolve into a final solution plan. The pseudo code of the algorithm is as follows:

Step 1 **Initialize** parameters
-Input forest grid
-Define local objective function z(neighborhood(), local attributes)
-Define global constraints
-Define function to define frequency of reevaluation of penalty multipliers (i.e. dynamic/adaptive function based on the size of the problem)
-Initialize iteration number iter = 0 and maximum number of iterations max_iter
Step 2 **Generate** random initial solution
Assign a locally feasible management regime x schedule state to each forested cell in the forest grid
Step 3 **BEGIN** Iterative decision process
DO
{
Step 4 **Retrieve** penalty multipliers
(recalculate at iteration interval frequency as defined in Step 1)
Step 5 **Generate** a random-order list of all decision units (forested cells)
Step 6 **Select** first forested cell *g* in the list
Step 7 **Calculate** neighborhood functions
Step 8 **Choose** the candidate state that maximizes *z(local neighborhood(), local attributes, penalty multipliers)*
Step 9 **If** new "best state" different from previous state
{
iter^{++}
go to step 5
}
Else go to next forested cell in the random-order list (step 6)
Step 10 }**WHILE**(g $\neq$ end of the random-order list AND iter <= max_iter)
END

4.2 Implementation: Object-oriented Programming for Flexible Use

Data support is a large part of what makes a planning system effective and efficient. Harvest scheduling models have evolved to accommodate spatial issues with the integration of geographic information systems and database tools. These relational database structures can handle a number of spatial objectives and constraints involving topology (e.g., adjacency issues) or location (e.g., riparian management). However, they are not efficient at addressing complex, local spatial relationships (e.g., size and shape of harvest blocks and conservation reserves) or at representing temporal data.

An object-oriented implementation of data, where each data element is represented by an object enclosing its own data attributes and associated methods, is useful for forming the basis of a platform capable of integrating prescriptive and spatial models. For a decentralized forest application, the fundamental objects are forest stand objects whose data elements include current age, site quality, forest type and treatment. In addition, an object-based implementation of the co-evolutionary CA model entails that each forest stand object holds its CA state (management schedule) as part of its data structure and is capable of changing this state upon execution of internal transition rules. When executing transition rules, a stand only needs to process its own information and that of its immediate neighborhood to achieve a decision.

Although all stand objects have an identical data structure, the actual data they contain is specific to their forest ecosystem type and to the specifics of the stand. Object classes or blueprints of stand objects are defined for each ecosystem type. All stand objects that are derived from the same class share the same growth and yield information and potential management information.

At the next level of abstraction is the cell surface or the grid object. The grid object can be construed as the actual surface of the forest management area, or as a two-dimensional array of stand objects. The grid object includes the solving algorithm, i.e. the function which calls the forest stand objects to execute their transition rules. It is therefore relatively easy to modify the solving algorithm without any changes to the forest stand objects. When applying the co-evolutionary algorithm, the grid sends a request to a forest stand at each iteration to execute its transition function. The forest stand does the actual state transition, but the grid coordinates the transition activity and provides the global parameters of the optimization function. These parameters include the weights associated with the respective objectives, the length of the planning horizon, the current time information and the incentive scheme to satisfy constraints.

The information contained in the forest stand objects and grid objects can be obtained from GIS raster thematic data layers (e.g. forest cover type, stand age, cell location, site quality or topology) and from other data sources (e.g., growth and yield curves, wood prices, and planning horizon). Most of the attributes and some state variables for the stands are directly extracted from GIS (e.g. site quality, stand type, age, topology etc.). However, some information requires pre-treatment in the GIS (e.g. distance to road, distance to mill, resolution).

Figure 1 depicts the conceptualized object-oriented CA framework with the forest stand objects, their blueprints, the forest stand classes and the grid object. Figure 1 also illustrates how both spatial and non-spatial information is used to fill the data attributes and functions of the objects. The object-oriented framework has the advantages of being flexible and reusable, easily linked to GIS and other databases, and is portable to other problems, other resolutions and other planning algorithms.

4.3 Conserving Old Growth and Generating Timber Revenues

To illustrate the feasibility of CA-based planning tools, a problem consisting of maximizing the weighted sum of timber revenues and old growth conservation measures over a 100 year (10 periods) planning horizon is defined and solved with the CA algorithm. The timber revenues are measured by the net present value of the management activities. The old growth conservation measure is based on both the old growth status of the stand and the proportion of the stand's neighborhood also in old growth in any given time period. Basically clusters of old growth are more valuable than scattered old growth. A stand is defined as old-growth if it has reached the particular age most commonly associated with the onset of old-growth

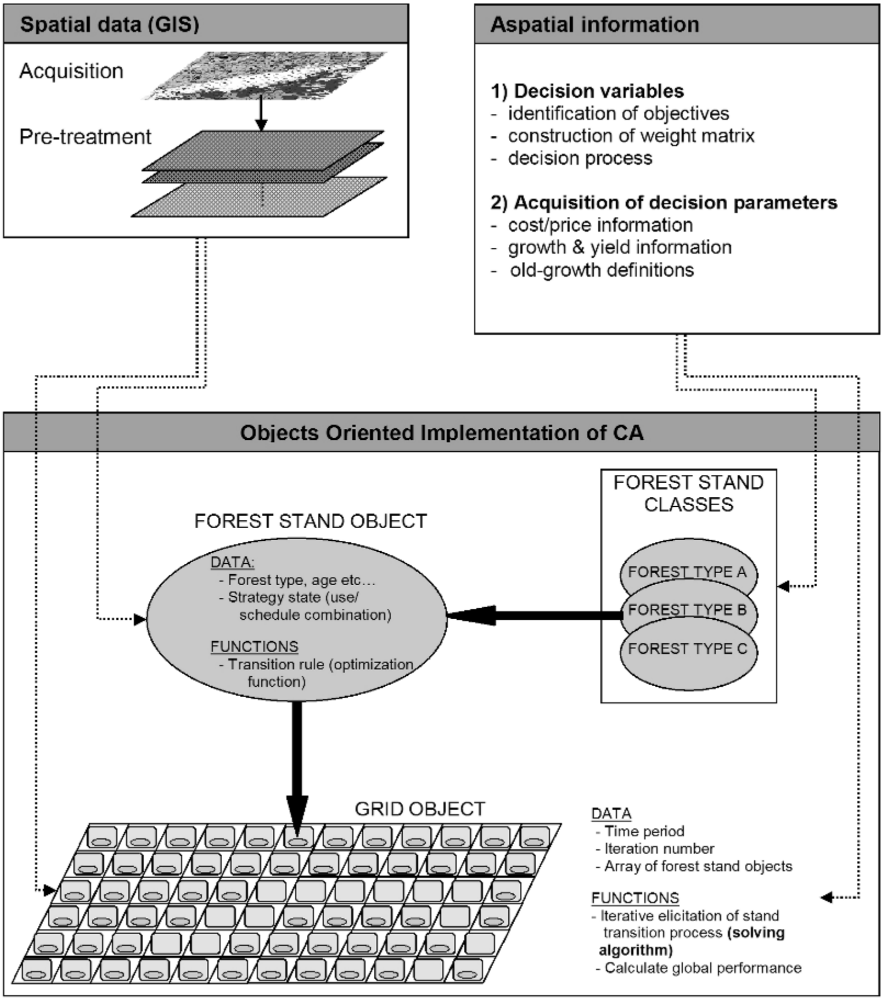

Fig. 1 Object-oriented CA framework for forest planning

characteristics in the ecosystem type of the stand (Uhlig et al., 2001). In this problem, managed stands of certain ecosystems can therefore become old growth if they have been left standing long enough.

The model is applied to a case study of about 40,000 ha of boreal forest divided into 9ha-cells. The forest is composed of sixteen different forest ecosystem types mainly dominated by black spruce. The transition rule of each forested cell at iteration i consists of maximizing a linearized two objective function involving old growth conservation and net present value. The decision variable is a combination of a silvicultural regime and harvest schedule for the forest cell. The silvicultural regimes considered range from "no-management" to extensive, basic, and intensive even-aged regimes and have predefined treatment sequences. Based on site quality and initial inventory, not all silvicultural options were available to all stands. Also, the proximity to sensitive areas such as riparian buffers, recreational and residential areas and wildlife habitat excluded intensive management regimes from the list of options. Local constraints on the type of management that can be undertaken in these sensitive areas were internalized in each forest cell.

The global constraints include stable harvest flows between 100,000 and 110,000 m^3/period and a minimum 12% retention of contiguous areas of old forest. All old growth forest on the management unit can contribute to meeting this constraint including reserved areas and otherwise unmanageable areas (e.g. economically inoperable). Global constraints are implemented using penalty and incentive multipliers, $\alpha_i^t, -\beta_i^t$ for volume and γ_i^t for old growth, in the local decision making function. The penalties and incentives applied to the stand decision function are periodically recalculated to reflect the landscape level of satisfaction for global constraints:

$$\alpha_t = \begin{cases} {v^t_{\mathrm{min}}}/{v_t(P)} & \text{if } v_t \le v^t_{\mathrm{min}}, \text{ the minimum harvest volume target} \\ 0 & \text{otherwise,} \end{cases}$$

$$\beta_t = \begin{cases} {v_t(P)}/{v^t_{\mathrm{max}}} & \text{if } v_t \ \ge v^{\mathrm{t}}_{\mathrm{max}}, \text{ the maximum harvest volume target} \\ 0 & \text{otherwise,} \end{cases}$$

$$\gamma_t = \begin{cases} {og^t_{\mathrm{min}}}/{og_t(P)} & \text{if } og_t(P) \le og^t_{\mathrm{min}}, \text{ the minimum old growth area target} \\ 0 & \text{otherwise.} \end{cases}$$

Where $v_t(P)$ is the volume in period t of the plan and $og_t(P)$ is the old growth in period t of the plan.

The stand transition rule is a local objective function formulated as follows:

$$\max_s z\,(s(f)) = \lambda \times \frac{NPV\ (s\ (f))}{NPV_f} + \sum_{t=1}^{T} \left[(\alpha_i^t - \beta_i^t) \times \frac{HV_t\,(s\ (f))}{HV_f} + \left(\frac{1-\lambda}{T} + \gamma_i^t \right) \times \frac{OG_t\ (s\ (f))}{OG_f} \right]$$

Here $NPV\ (s(f))$ is the discounted net revenue of management and harvest over the planning horizon when schedule $s()$ is applied to stand f; $HV_t\,(s\ (f))$ is the harvest volume in time period t, and $OG_t\ (s\ (f))$ is the old growth value in time period t. The revenue, $NPV\ (s(f))$, depends on of different wood grades, harvest volume, silvicultural costs and harvest costs, which include falling, skidding and constructing spur roads. The cost of building spur roads decreases if adjacent stands are harvested in the same time period. The old growth value $OG_t\ (s\ (f))$ increases if adjacent stand are in old growth in the same time period. $NPV_f,\ HV_t\,(s\ (f))$ and OG_f are normalizing factors. The weight associated with the net present value objective is λ.

To illustrate the behavior of the CA algorithm, the planning problem is solved with and without global constraints. The CA algorithm is also run with different levels of λ to test its sensitivity and to construct a tradeoff curve between NPV and old growth. The object-oriented forest planning CA was coded in C + + and executed on a Personal Computer (PC) with an Intel(R) Core^{TM} processor with 2.4 GHz CPU speed. Each program run took about 15 seconds to run for 100,000 iterations.

5 Results

The output of the forest planning CA is examined in terms of the iterative progression of the solution and the impact of constraints on the tradeoff curve between NPV and old growth. We also include age-class maps of selected solutions to demonstrate how the model clusters both harvests and old growth.

5.1 Progression of the Solution Objective Function

Figure 2 illustrates the evolution of the objective function when the weight associated with the net present value objective is $\lambda = 0.7$ and the weight associated with old growth conservation is $1 - \lambda = 0.3$. For the first 20,000 iterations, the problem has no global constraints, after which it is constrained for harvest levels between 100,000 and $110,000\,\mathrm{m}^3$/period and for a minimum of 12% old growth. The reason

Fig. 2 Progression of the objective function when $\lambda = 0.7$. For the first 20,000 iterations, no global constraints are applied, after which harvest level and old growth conservation constraints are applied

why the model was run for 20,000 iterations without the global constraints is to allow it to find polarized solutions for NPV and old growth. Since this solution will be far from the best constrained solution, it represents a good starting point from which to test how the CA algorithm performs in meeting global constraints.

Over the first 20,000 iterations, there is only a slight increase in the objective function value when there are no constraints present. When the constraints are applied after 20,000 iterations, the objective function drops dramatically and then increases rapidly. Around 40,000–50,000 iterations, there is considerable variation in the objective function value, however, it becomes smoother as the iterations increase. The penalty/incentive parameters are recalculated periodically but more frequently as the iterations progress. As the iterations progress, there is less discrepancy between the penalties/incentives from one recalculation to the next which decreases the fluctuations of the objective function value. The plan gradually satisfies the constraints while increasing the objective function value.

5.2 The Impact of Constraints

The tradeoff curves between net present value and old growth conservation value with and without the application of constraints are depicted in Fig. 3. The unconstrained solutions (Fig. 3a) were obtained by running the model under different levels of λ and saving the unconstrained solution generated at 20,000 iterations, while the constrained solutions (Fig. 3b) were those generated at 100,000 iterations. Values of λ range from 0–1.0 in increments of 0.1

In the unconstrained tradeoff curve (Fig. 3a) increasing net present value is associated with a decreasing area of clustered old growth. As the NPV level increases, the rate at which clustered old growth decreases becomes more pronounced. At $\lambda <= 0.4$, no management takes place and no harvest occurs: the net present value equals zero. Unless the weight associated with net present value reaches a relatively high level, it is almost always easier for stands to derive value from opting for old growth conservation rather than management and harvest.

When constraints for minimum old growth and stable harvest flow are applied, a different trade-off pattern is observed: increasing net present value is associated with increasing old growth conservation (Fig. 3b). This reverse trend is at first surprising, but the result can be linked to the net present value measure, which partly depends on the spatial arrangement of the harvest since clustered harvest blocks generate economies of scale by sharing the cost of road building. What is likely happening in the constrained tradeoff curve is that for higher weights associated with net present value, the model is more careful as to where to allocate management and harvest activities. This results in clustered harvests, lower costs and as a consequence, a less fragmented landscape with a higher old growth value. On the other hand, for lower weights associated with net present value, it is solely the constraints on volume flow that force the model to harvest some stands regardless of their spatial arrangement. This results in more scattered and lesser value old growth and in smaller net

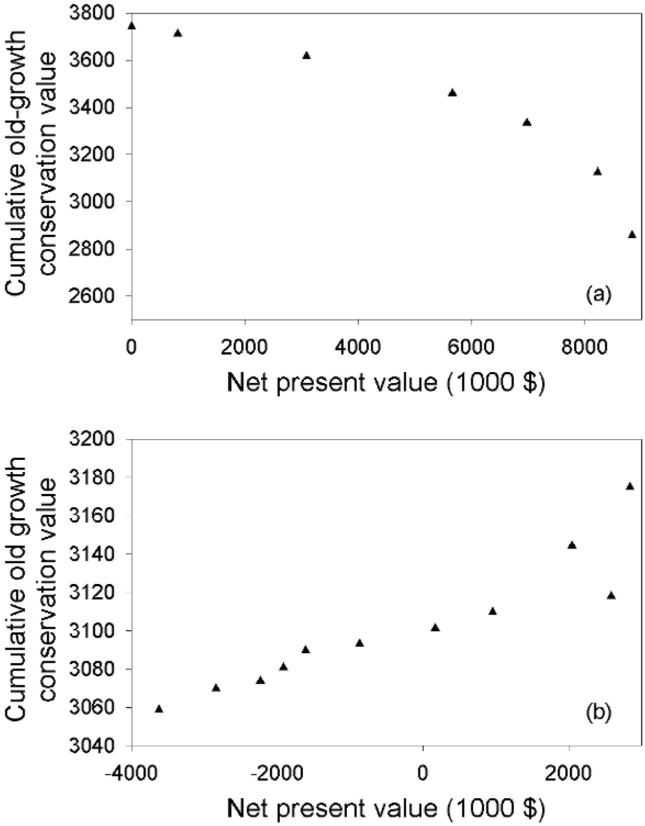

Fig. 3 Tradeoff curves between net present value and the cumulative old growth conservation measure for (a) the unconstrained problem and (b) the constrained problem ($0 < \lambda < 1$ in increments of 0.1). In (a) $\lambda <= 0.4$ all result in $NPV = 0$

revenues. If the harvest upper bound was progressively increased, then the tradeoff curve would eventually show the same trend as the unconstrained case.

The volume harvested in each period for both constrained and unconstrained scenarios further supports this explanation. Fig. 4 illustrates the amount of harvest volume over time for both the constrained and unconstrained cases under different levels of λ.

When no constraints are applied, increasing the weight on net present value results in increasingly greedy NPV solutions where stands are harvested as soon as possible (Fig. 4a). When constraints are applied, lower NPV weighted scenarios tend to hit the lower bound on the harvest level whereas higher NPV weighted scenarios hit the corresponding upper bound (Fig. 4b). In all scenarios, constrained and unconstrained, the old growth area is always satisfied well beyond the 12% requirement in each time period and is not shown. The 12% old growth constraint is not binding these problems.

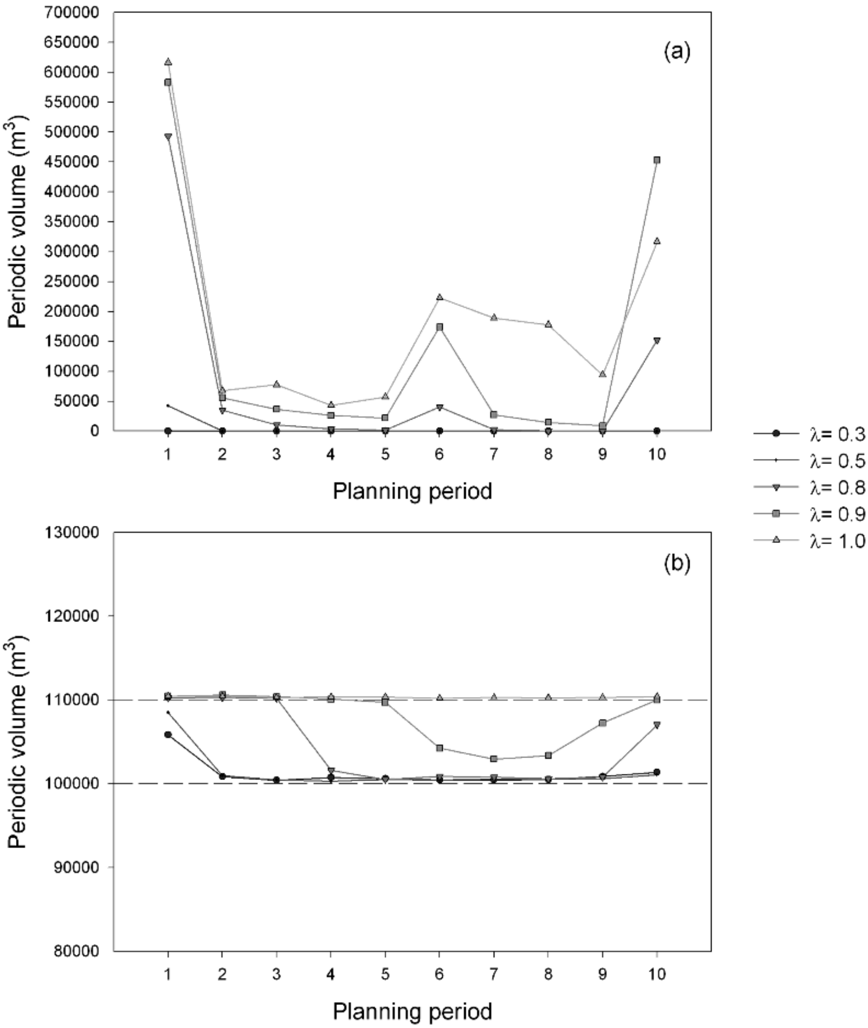

Fig. 4 Periodic harvest volumes when $0 < \lambda < 1$ for (a) unconstrained solutions and (b) constrained solutions (dashed lines represent the $100,000m^3$ lower and $110,000m^3$ upper bounds on harvest levels)

5.3 The Spatial Arrangement of Harvest and Old Growth Clusters

In the absence of constraints and with the application of a high NPV weight ($\lambda = 0.9$), there is a clear clustering of harvesting in specific periods (Fig. 5a), which also results in a high level of clustering of old growth. Figure 5 illustrates the spatial arrangement of old growth, harvested stands and "young forest" (managed or unmanaged forest that has not reached old growth status) for two sample runs. Even with the application of volume constraints, the solution still demonstrates a

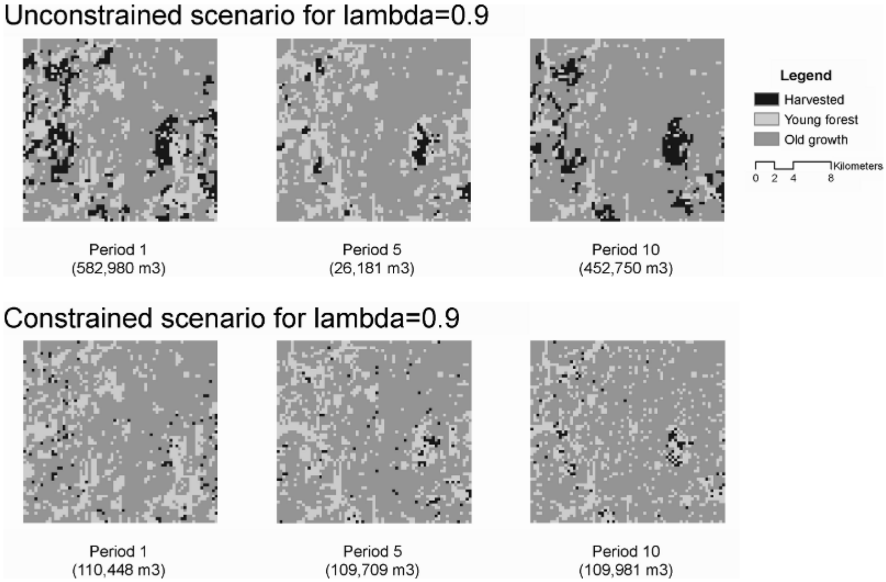

Fig. 5 Location of harvests, young forest and old growth in period 1, 5 and 10 when $\lambda = 0.9$ for (a) unconstrained solution and (b) constrained solution

high level of clustering, although it is less obvious in Fig. 5b because the level of harvest in the selected periods is lower compared to the unconstrained case.

There is a relatively large amount of contiguous old growth left in the tenth period of the unconstrained scenario for two reasons. First, accessing and harvesting timber in low-productivity stands often results in costs exceeding revenue, hence in the absence of harvest constraints, more stands are left grow into the old growth class. Second, in the unconstrained case, very high, concentrated harvests occur in only a few periods and this results in less fragmentation of old growth relative to the constrained case that forces near uniform harvests in all periods.

The decentralized planning approach successfully satisfied global constraints in each planning period despite locally made decisions and further achieved old growth clusters rather than fragmented old growth within each planning period.

6 Conclusion

6.1 Cellular Automata Case Study

The cellular automata-based planning model is capable of handling traditional forest management objectives such as harvest volume, net present value and old growth retention. In addition, this approach successfully clustered conservation areas and harvest activities, which further enhance the value of the final plan by generating economies of scale and limiting forest fragmentation. Local constraints were dealt with locally and global constraints were included in the form of parameters to the local decision making function. These parameters periodically evolved to reflect the level of satisfaction of the whole plan to the global constraints. In this model, all constraints and objectives, spatial and aspatial, are traded off not only within one planning period but across all periods of the planning horizon. The introduction of constraints on the problem significantly modified the tradeoffs between old-growth conservation and net present value. It should be noted that the CA-based decision process presented here does not lead to an exact solution because it involves several random features (e.g. initial random solution and random order list of forest stands). Although it has proven to be consistent across multiple runs (Mathey et al., 2007), it is important that replications be conducted to identify the range of model outputs for a particular management scenario.

The object-oriented data support for the implementation of the planning model has the advantage of avoiding the need to access information not directly relevant to the procedure being called. The re-usable property of the object-oriented framework provides easy transfer of the planning model to different planning problems and geographic conditions (inventory, spatial and temporal resolutions, etc.).

6.2 Decentralized Planning

The case study and its results draw attention to several key characteristics of decentralized planning methods. In a decentralized planning framework, the whole system to be optimized (forest management) is split into subsystems which operate as independent decision units (multiple stand management units). The subsystem decisions are not independent because adjacent stand management can increase or reduce the old growth value and operational costs of a particular stand. This requires a mechanism whereby each subsystem decision must account for other subsystem decisions. This mechanism allows the gradual emergence of a decision pattern and insures that decisions converge towards an overall optimum.

There are two main implications of the co-evolutionary mechanism used to direct subsystem decisions. First, a limited set of dependencies such as the eight adjacent cells of the case study, drastically reduces the solution space as compared to centralized heuristics. This also limits the amount of information that needs to be accessed when evaluating subsystem decisions. In large combinatorial problems this makes it possible to reach a solution faster and with less computer memory relative to the centralized approach. Second, the co-evolutionary mechanism ensures that good spatial layout is achieved in the final solution. Local constraints are easily satisfied by the subsystem itself. Since no "central coordinator" nor other subsystems need explicit knowledge of the feasible solution set of a subsystem, the inclusion of local constraints and their computations is not as cumbersome as in centralized methods. Furthermore, satisfactory spatial arrangements are more likely to develop in decentralized planning methods because within a fixed computing time, all local alternatives and their topology can be inspected many times, while within the equivalent time in a centralized model only a fraction of the solution space can be evaluated.

6.3 Further Research

Although the application presented here is limited to evaluating net present value, old growth, volume, and stand specific management options, it could be modified to include other processes such as succession. Further modifications could consist of varying the importance of each objective in different planning periods, or changing timber prices and management costs along the planning horizon. These additions would be a significant step toward acknowledging and planning for the dynamic environment in which forest management operates. The object-oriented framework could also support the design of harvest blocks or reserves that fit some shape requirements or some specific distribution (e.g., natural disturbance emulation in harvest openings).

Including natural disturbances and designing management strategies that are robust in the face of unforeseen events is a further area of improvement for a decentralized framework. The evolution of the forest landscape can be simulated by cycling through time periods, aging the landscape, implementing succession and coupling this evolution with a natural disturbance model. CA platforms have been successfully used to simulate fire and its spread; they have also been used to simulate the progression of insect outbreaks. These models could be integrated here to provide more realistic pictures of future forests.

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Coordinating Management Decisions of Neighboring Stands with Dynamic Programming

Howard Hoganson, Jose G. Borges and Yu Wei

1 Introduction

Stands are typically thought of as the basic analysis area (AA) or building block of a forest. Historically, forest management planning has focused on how stand management activities can be coordinated across the forest to provide a sustainable flow of products over time. With increasing concerns about sustaining environmental conditions, spatial facets of the management situation also become a concern. Often, how a stand is managed impacts not only the condition of the stand, but also the condition of a "neighborhood" surrounding the stand. These impacts can have a lasting impact over time, thus adding a temporal dimension to spatial concerns. Spatial concerns add substantial complexity for analyzing management options because forests include many stands, each with its own potential unique neighborhood. Generally, it is too simplified to separate the forest into independent neighborhoods, ignoring conditions along neighborhood boundaries.

In recent years adjacency constraints have received considerable attention as one policy tool for addressing spatial management objectives. Adjacency constraints set maximum limits on the size of harvest blocks. Specific management limitations for adjacency constraints vary, without any one accepted best set of rules. Typically, the rules involve only regeneration harvests associated with even-aged management. Rules may limit harvesting of adjacent stands within a given time period or set a maximum limit on the size (total area) of any contiguous harvest block. Rules involve an "exclusion period" during which harvesting is prohibited for adjacent stands around a harvest block. Definitions of exclusion periods also vary by user. Some assume a fixed time length for the exclusion period (e.g. 10 years) or tie its definition to the condition of the harvest block (e.g. not until the trees regenerated in the harvest block reach 20% of their height at rotation). Methods for incorporating adjacency constraints into forest planning models have received considerable

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attention in the forestry literature. Good reviews are provided by Murray (1999), McDill and Braze (2000) and McDill et al. (2002) and Weintraub and Murray (2006).

The use of adjacency constraints can be too simplified for describing spatial management objectives. Adjacency constraints tend to distribute harvests into small blocks that can fragment the remaining blocks of older forest on the landscape. Forest management planning often needs to be concerned more about the size of patches of older forest remaining on the landscape after harvests rather than on the size of the harvest blocks.

Core area (interior area) of older forest is one spatial measure that has received more attention in forest planning in the last decade. Core area is the area of the forest that is free of edge effects from surrounding habitats (Zipper 1993; Baskent and Jordan 1995). Harvesting a single stand not only eliminates the core area of older forest contained in the stand, but it can also reduce, for many years to come, the amount of core area of older forest in neighboring stands. Thus managing for core area of older forest attempts to focus harvests in areas where impacts of harvesting on neighboring stands do not reduce core area production. Harvests tend to be in larger blocks, but the emphasis is on the forest left on the landscape rather than on the size of the harvest blocks. Managing for core area of older forest is thus somewhat the opposite of managing with adjacency constraints that focus on harvest areas and limit the size of those areas.

Öhmann and Eriksson (1998), Öhmann (2000) and Öhmann and Eriksson (2002) have used core area as a spatial explicit management objective in forest management scheduling models based on simulated annealing. Hoganson et al. (2003) and Hoganson et al. (2005) have tracked and valued core area in forest management scheduling models based on dynamic programming (DP). Applications of both simulated annealing and dynamic programming to schedule core area production have considered thousands of stands. Rebain and McDill (2003) presented a mixed integer programming (MIP) formulation for developing large patch areas. Few if any applications of a MIP approach to large areas have been reported in the literature.

Dynamic programming (DP) is a useful tool from operations research that allows one to address a series of interrelated decisions by structuring analyses to decompose the problem into a series of smaller, inter-related problems. It helps avoid the problem of needing to enumerate and evaluate all possible management options. DP has been used extensively to address management thinning options for individual stands. Corresponding with a DP formulation is a decision tree that illustrates the many possible management paths for the stand in terms of timing and thinning intensity. DP analyses can be complicated substantially if stand regeneration options are considered that can change tree species composition after a regeneration harvest. DP formulations can be solved both forward and backward through time with each solution approach offering different advantages.

In this chapter we will introduce dynamic programming and then review how it has been applied to address stand-level management decisions in forestry. Then, we will expand the problem to consider how dynamic programming can also help coordinate management spatially. We'll then consider how this approach can be used in a wider forest-wide planning context and look at some recent applications involving real world situations for large planning areas.

2 Introduction to Dynamic Programming: The Stagecoach Problem

The Stagecoach problem is often used to introduce dynamic programming because of its play on words – parts of the story also use terms that have a specific meaning for describing a dynamic programming problem. It is a stretch to call the stagecoach problem, a natural resources problem, even though it involves travel through remote natural areas. It is set in the wild west of the USA, where a person wishes to travel cross-country via a series of stagecoaches (Fig. 1). We'll assume the person wishes to travel home safely from California to Minnesota with his new fortune from the 1849 gold rush. Each leg of the journey involves a separate stagecoach ride that starts and stops in a different state (Fig. 1).

This traveler's main objective is to complete the journey safely. The traveler has obtained information on the cost of insurance policies for each potential leg of the journey, realizing that this information is valuable because the safest route is likely the route with the lowest cost of insurance. He realizes that the optimal path home from California to Minnesota is not necessarily the one where he takes the safest stage out of the current state. In other words, he realizes that it is likely important to plan ahead to consider the safety of the entire multi-stage trip.

Our example can be described as a 4-stage problem because the traveler will travel via 4 stagecoaches. Each stage will start and stop with the traveler ending in a

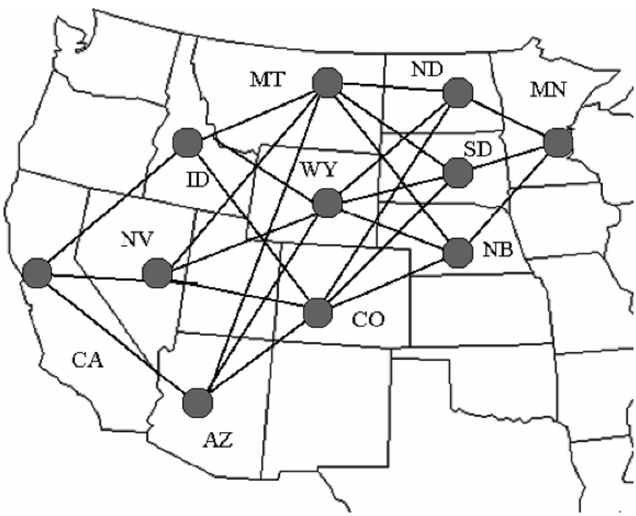


Fig. 1 A stagecoach problem for traveling from California to Minnesota

different state. When in any state at the start of a stage, a policy decision is involved regarding which route to take. The policy decision, with its leg of the journey, will transform the traveler from the state at the start of one stage to the state at the start of the next stage.

The network shown in Fig. 1 represents a dynamic programming (DP) problem because the problem can be separated and solved in distinct stages, The basic idea is to break the problem into a series of linked and smaller problems. For our example, the problem may seem complicated because there are many possible routes from California to Minnesota. But let's examine how one can solve this problem in stages. The problem of traveling from any of the states for the last stagecoach ride into Minnesota is clear, as there is only one choice from the three possible states at the start of this last stage of the problem: North Dakota, South Dakota, and Nebraska. We can easily "label" each of the nodes corresponding with these states with the total cost of insurance for traveling from that state to Minnesota. And using these node labels and the insurance costs associated with the immediate travel options for the states with two stagecoaches left to ride (Montana, Wyoming and Colorado), it would be easy to identify the low cost insurance route from traveling from any of those states to Minnesota. For example, to find the low-cost solution from Colorado to Minnesota, one just compares the total insurance costs of three options: (1) Colorado to North Dakota plus the low-cost value from North Dakota to Minnesota, (2) Colorado to South Dakota plus the low-cost value from South Dakota to Minnesota, and (3) Colorado to Nebraska plus the low-cost value from Nebraska to Minnesota. Once the optimal solutions are determined for all stages at the start of this 2-stage problem, the value of their optimal solution can be used to determine the optimal solutions for the three possible states with three stagecoaches left to ride. As an example, determining the low-cost solution from Nevada to Minnesota requires comparing only three insurance cost options: (1) Nevada to Montana plus the low-cost value from Montana to Minnesota, (2) Nevada to Wyoming plus the low-cost value from Wyoming to Minnesota and (3) Nevada to Colorado plus the low-cost value from Colorado to Minnesota. With the low-cost solutions known for all the states possible at the start of the three-stage problem, then the values of these solutions are used to find the low-cost solutions for all states possible at the start of the 4-stage problem. For this example, only one state (California) is possible at the start of stage 4 – the stage and state of interest for the problem described.

Overall, dynamic programming problems are defined by this recursive nature where the solutions for possible states at stage n are used to find the optimal decision for possible states at stage $n + 1$. Figure 2 shows the details of this solution process for our example. Arc values represent the cost of insurance for travel between the starting and ending states associated with the arc. During the solution process arcs were marked with a star$(^*)$ if they correspond with the optimal route to take from the state where the arc originates. Tracing the optimal route from California to Minnesota involves following the path of starred decisions starting with the decision for the California node and then traveling to the east (forward) through the network. Note that for this example, the DP network was solved backwards, compared to the direction of the journey. Alternatively, the network could have been solved forwards,

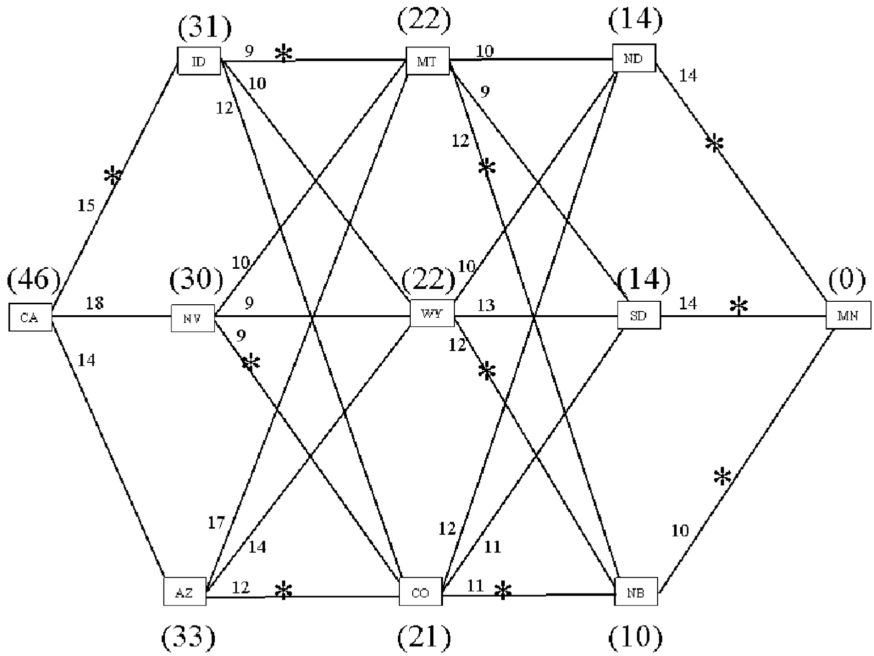


Fig. 2 Solving the stagecoach problem by labeling the DP nodes for each stage starting with the Minnesota node. Node labels are in () with each * indicating the optimal route from the starting state. Arc values represent the cost of an insurance policy for traveling between the states associated with the end of each arc

by starting from the California node, and labeling nodes successively forward, stage by stage, through the DP network. And if solved forward, then the optimal solution is traced backward through the network, starting from the Minnesota node.

The stagecoach problem highlights how DP problems can be divided into *stages* with a *policy decision* required at each stage to identify the optimal decision. Each *stage* has a number of *states* describing conditions at the start of the stage. The effect of a *policy_decision* is to transform the current *state* into a *state* associated with the start of the next stage of the problem. The *policy decision* is simply the best option (path) to follow at the corresponding *stage* and *state*. For any stage and state, it does not matter how the stage and state was achieved. This is referred to as the principle *of optimality, the memoryless_principle*, or a_*Markovian process. A recursive relationship* identifies the value of a possible decision in terms of (1) the value of the decision and (2) the value of the next stage and state associated with that decision. In a graphical representation of the problem, *Nodes* refer to stage and state combinations and *arcs* refer to the decisions available at each node. Notation varies between applications in defining stages and the recursive relationship between stages. For example, with n stages, one can say either we are: (1) searching for the value of stage n and state s where the value of decisions at stage n depend on the values of states at stage $n - 1$, or (2) searching for the value of stage n and state s where the value of decisions at stage n depend on the values of states at stage $n + 1$.

Recursive formulas of DP often seem confusing to students initially. A graphical representation often helps understand the formulation. Formulating the problem is usually the biggest challenge. There are not any hard and fast rules, but a few points to remember can be helpful:

1. The basic idea is to divide a large problem into a series of smaller problems. It is important to recognize how the stages are related. What are the smaller problems and why are they easier?
2. Stages are often defined in terms of: (1) the number of decisions (allocations) remaining or (2) time (or age).
3. States define the "possible states of the world" for a given stage. Some examples might be "dollars remaining to allocate" or "possible forest stand characteristics at the associated stage of development."
4. States should not be confused with the decision options associated with the current stage. The possible decisions move one from one stage and state to the next stage. The decision made will dictate the state associated with the next stage.
5. What does one want to max or min? The values associated with the arcs should be associated with this objective.
6. Constraints (requirements) might actually simplify the problem by reducing the number of states possible for a given stage or reduce the number of possible options at specific nodes.
7. Constraints will likely complicate the problem if the constraints impact the memoryless principle. For example, if we were allocating resources to Projects A, B and C, and a constraint requires that at least X resource units be allocated to either Projects A or B, then we need to remember how many resources are assigned to Project A when we consider how many to assign to Project B. (or vise versa).
8. The memoryless principle must apply! This should be a primary concern when determining whether DP applies and when identifying the stages and states of the problem.
9. The objective function need not be additive across stages. For example, minimizing the probability of failure might require multiplying the probability of success across all the arcs selected through the DP network.

3 Dynamic Programming for Stand-level Decisions

Dynamic Programming (DP) is useful for evaluating decisions related to the best combination of timing and intensity of thinning operations. Pioneering works on how to apply dynamic programming to the stand-level thinning problem include Hool (1966), Amidon and Akin (1968), Kilkki and Vaisanen (1969), Naslund (1969), Schreuder (1971). Applications followed focusing on detail of specific species (Martin and Ek 1981; Brodie et al. 1978; Brodie and Kao 1979; Haight et al. 1985; Arthaud and Klemperer 1988). Applications often involve starting conditions where the land is considered bare, and optimal solutions provide management schedules for full rotations under constant timber prices and costs. We'll start by using those simplifying assumptions and later discuss modifications to address situations where the initial stand condition could involve a stand of any age and stocking. The basic concepts of forest economics apply, where consideration is given to both the value of the existing rotation and the value of future rotations. In effect, discounted net returns from the first rotation may be greater with a longer first rotation, but delaying the start of the next rotation has a cost, as returns from future rotations are then delayed.

Most applications of DP to the stand-level thinning problems have involved even-aged management and have used time to define the *stages* of the problem. Points in time when stand treatments are possible are each considered as a separate stage. With age at the start of the planning horizon equal to zero, then stand age and time are the same. Each *stage* refers to a specific time interval, for example, from age 30 to age 40. The length of each time interval need not be the same. Stand treatments are assumed to occur at the *end* of this time interval stage. This assumption helps reduce concerns with rounding error. Rounding error will be discussed in more detail after the basic structure of the DP formulation for the stand thinning problem is described. Using shorter stage intervals offers at least two advantages: (a) more possible times of stand entry possible – for example ages 15, 20, 25, 30, 35 rather than just ages 20 and 30, and (b) more possible intervals between thins – 5, 10, 15, 20, 25 years rather than just 10 and 20.

States describe the condition of the stand at the start and end of each stage. States can be combinations of one or more stand descriptors such as volume, basal area, number of trees, average diameter, current stand value, etc. Figure 3 shows a simple example using a stage interval of 10 years and stand basal area as a single state descriptor, with each node having a state interval of 10 square feet of basal area. Each state condition is referenced by the midpoint of its associated state condition. Of concern is selecting state descriptors that can be used to predict both future stand growth, and current stand volume and value. Using multiple state dimensions and smaller states helps to recognize more intensities of thinning possible. It can also help reduce rounding errors caused by the discrete nature of the problem. Using multiple state descriptors helps to better describe future growth potential and average tree size. This relates to the memoryless principle of dynamic programming.

Arcs from each state (Fig. 3) represent possible treatment options. One option for each state in each stage is typically a "grow only" or "do nothing" arc. Thinning options bring the stand to the midpoint of a state in the next stage. Midpoints of state intervals thus represent possible states immediately after thinning Rounding error is not associated with thins, if it can be assumed that thinnings can be implemented to achieve the associated midpoint conditions of a given state.

Objective function values are associated with the "flows" the discounted benefits and costs of management options represented by the arcs. Many arcs are likely to be "grow only" arcs and thus have no direct objective function value associated with it. The "grow only" arcs have an associated rounding error in their projection of

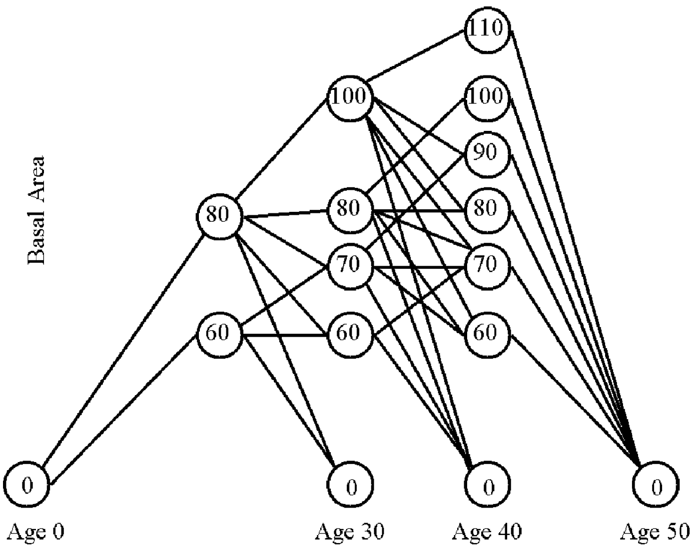


Fig. 3 A simple stand thinning problem

growth, as these arcs connect to the state that most closely reflects the projected growth condition.

Rounding error associated with stand growth is a potential concern. Rounding error is not necessarily random for the optimal solution found using DP. In solving the model, maybe the process is good at finding rounding errors that increase the estimated value of optimal solution. The optimal solution found by the model can be checked outside the model, where each associated arc need not be rounded to a nearest state at each stage (age) represented in the DP formulation. With more stages used in the formulation (more ages of possible treatment), more arcs will be associated with the optimal path. Each arc of the optimal solution potentially has rounding error, and thus the total rounding error is potentially larger when more stages are used. Using more states usually reduces rounding error. With many stages, the time interval between stages is smaller and thus the growth of the stand will be less. To model this growth accurately, the interval for defining states likely needs to be less. Matching stage interval with state interval is thus important.

Formulations of the stand management-thinning problem have been solved either forward or backward through time. Each solution direction has potential strengths and potential weaknesses.

3.1 Forward Through Time Approach

This approach generally only applies to situations involving a full rotation. It starts by labeling the node at the start of the planning horizon (time = 0) as having a zero value and then solves forward through time, stage-by-stage. Nodes values (node labels) represent the maximum PNV from time zero to the time associated with the corresponding stage. As an example and in terms of DP network shown Fig. 3, the forward solution approach would use the value of the age 30 nodes to find the value of the age 40 nodes. More specifically, for the Age 40 node with a basal area of 90, the node value would be the maximum value of the two options connecting this node with the age 30 nodes: (1) the value of the age 30 node with basal area 70 plus the value of the arc connecting that node with the age 40 node with basal area 90, and (2) the value of the age 30 node with basal area 100 plus the value of the arc connecting that node with the age 40 node with basal area 90. The second of these options represents a thin at age 40 as the basal area at age 40 is less than the basal area at age 30. With the forward solution approach, the node values for all of the bare land nodes at the end of a full rotation are converted to a soil expectation value (SEV) estimate. These SEV estimates represent the estimated value of an infinite series of rotations. Values for alternative rotation lengths are then compared to find the rotation age that maximizes SEV. For the rotation length with the greatest SEV, the optimal management schedule is found by tracing the optimal path backwards through the DP network, starting with the bare land node corresponding with the greatest SEV.

3.2 Backward Through Time Approach

An estimate of the bare land value at the end of the rotation is needed to start this approach for solving the DP formulation. This estimate is used to value all the bare land nodes associated with each of the possible rotation ages. Each of these nodes in the DP network is valued (labeled) as the estimated bare land value discounted the number of years corresponding with the point in time associated with the stage. For example, the bare land node in year 60 would be labeled by discounting the bare land value estimate using the appropriate discount factor for 60 years. This solution approach solves the network backward through time, with the last node labeled being the bare land node at the start of the planning horizon. Nodes values (node labels) represent estimates of the maximum net present worth that can be achieved from that node to infinity. Although solved backward through time, the optimal solution is identified by tracing the optimal path forward through time starting with the node at the start of the planning horizon. The optimal solution will pass through a bare land node at the end of the first rotation. The difference between the value of the bare land node at the start of the planning horizon and the value of the bare land node at the end of the rotation represents the estimated value of the first rotation. The estimated value of the first rotation can be converted to an SEV estimate to re-estimate the land value.

If the initial land value estimate used with the backward approach is a poor one, the solution found may not be the optimal solution. The solution process can be repeated using the re-estimated bare land value estimate described above. Through repeated iterations of re-estimating the land value by resolving the problem, the process will converge on the optimal solution. This typically happens in just 2–3 iterations.

Any new land value estimate based on a solution found cannot be greater than the true optimal value. Increasing a land value estimate can lead to a shorter rotation length. Similarly, decreasing the land value estimate can lead to a longer rotation.

3.3 Comparing the Solution Approaches

An advantage of the backward solution approach is that each of its node values, at any stage, represents the value of the stand in that condition from that point to infinity. The backward approach thus shows how to treat a number of possible future stand conditions. This is potentially valuable information suggesting what should be done if the stand ends up in a condition not on the optimal path. Another advantage of the backward approach is that the one network can include a number of initial starting conditions, and thus the solution process can solve for all initial states at the same time.

An advantage of the forward solution approach is that one might solve the problem while one is drawing the network. Then, nodes that are not feasible need not be addressed. With the forward approach, it also may not be necessary to analyze long rotations, if it becomes clear that SEV values are dropping as the rotation length is increasing. The forward solution approach also does not need an initial estimate of the bare land value, nor does it need to be applied in an iterative process involving re-estimates of the bare land value. However, if it is to be used to estimate the value of an existing condition other than bare land, it cannot convert first rotation NPV estimates to SEV estimates. Instead, it must add an estimate of the bare land to develop an estimate of future rotations. That value can be developed through a separate DP analysis. Also, with the forward approach, separate DP networks need to be solved for each initial stand condition.

A potential advantage of the forward approach is that it might help reduce rounding errors associated with stand growth by using the optimal path into a node to represent the conditions of the node rather than the midpoint of the node. It is still important to consider whether state intervals are small enough so that all possible conditions for the node represent a condition of equal value. One should question whether the low-cost option for achieving a node reaches the node at a condition that has substantially less potential future value than other options for arriving at the node. For example, if the low-cost path has an associated stand basal area of $9.8\,\mathrm{m}^2$, it may be more desirable to arrive at that same node with a basal area of $10.2\,\mathrm{m}^2$ if the additional cost of doing so is small.

Whether using a forward or backward approach through time, it is important to understand the potential considerations for selecting the state dimensions. Wide state intervals can lead to solutions that are substantially less than optimal (Pelkki 1997). Figure 3 uses only 1 state dimension. Basal area is often used as one dimension, as most growth models use basal area for projecting stand-growth. But alone, basal area is not likely a good indicator of stand volume, timber value per volume unit, or future stand growth. Using average tree diameter as an additional state dimension can help, as then product size class and volume estimates can be improved substantially. State dimensions need not use the same descriptors at every stage. Stand volume or even stand value might be used at older ages near rotation age where accurate estimates of stand volume may be more important than estimates of future basal area growth. In a DP analysis of the stand thinning problem, it is also likely desirable and possible to recognize, the stochastic nature of the problem (Norstrom 1975, Gunn 2005). This complicates the analysis and is beyond the scope of our presentation here. As one might expect, applications of these models show that optimal management actions depend on outcomes associated with an uncertain future (Haight and Smith 1991). Overall, with an uncertain future, it is desirable to have management plans that consider the flexibility of decisions – the recourse possible once uncertainty about the future unfolds.

3.4 Core Area and Influence Zones

Ecological management objectives often involve important spatial detail related to forest fragmentation. Forest fragmentation refers to increasing spatial isolation and decreasing patch size of forest habitats (Harris 1988). A consequence of forest fragmentation is the loss of core area of mature forest. The plant community of core area is likely to be quite different from the plant community of the forest edge zone (Honnay and Verheyen 2002). Core area provides important habitat for wildlife species associated with forest interiors (interior species). Both human and natural disturbances often cause the amount of core area of mature forest to decline (McGarigal et al. 2001). Less core area generally produces a higher population of edge species and less interior species (Brand and George 2001).

A map of "influence zones" can be used to define the spatial interrelationships that impact the production of core area. Any location in the forest that can potentially produce core area, either currently or in the future, is assumed to be part of an influence zone. An influence zone identifies those forest AA's (stands) that influence all of the area in the zone in terms of its potential to produce core area. Whether or not core area is produced in the zone depends on the condition of *all* of the stands influencing the zone. These conditions are controlled (influenced) by the management decisions for the associated stands. Each influence zone is defined by a unique combination of stands that influence the zone. Each influence zone may or may not be contiguous. For large stands there will be an influence zone in the center of the stand that is influenced only by that stand. This zone is far enough within the stand to be buffered by the stand for producing core area. Other areas in the same stand are influenced by additional stands because they are closer to the stand boundary. Portions of the stand may not be in any influence zone, if close to major roads or other landscape features that cannot provide adequate buffer for core area. Generally for a forest there are many more influence zones than stands. Each influence zone can be labeled by the stands that influence it, and the number of stands influencing it is considered the influence zone's dimension. For example an influence zone ABC is influenced by stand A stand B, and stand C because all of that influence zone is within the core area buffer distance from each of those stands. It has a dimension of three because it is influenced by three stands.

The specific area boundary for each influence zone depends on the buffer width assumptions used to define core area. The buffer width represents the depth of edge effects, which potentially varies by landscape features and the specific ecological concerns associated with core area. With wider buffer widths, specific locations that can potentially produce core area are more likely to be influenced by more stands nearby.

To help understand the concept of influence zones, consider a forest of four stands (A, B, C and D) with irregular shapes (Fig. 4a).

By buffering outward from the boundaries of each stand, a set of influence zones {A, B, C, AB, BC, BD, CD, and BCD} can be identified (Fig. 4b). The 1-dimensional influence zone defined by stand B is an example of an influence zone that is not contiguous (Fig. 4b). The portion of each stand within an influence zone is referred to as a subzone (sub influence zone). Spatially, each subzone is inside the boundary of only one stand but influenced (for core area production) by all other stands defining the influence zone. By recognizing subzones, one can differentiate between requirements for core area and requirements for the buffer surrounding core area. Specifically, a subzone may produce core area if its associated stand meets the condition requirements for core area and of the other stands influencing that zone meet the requirements for core area buffer. The area of each influence zone is split between its subzones. Some stands influencing a zone may not contain any of the area of that zone so the number of subzones can be less than the dimension of the influence zone (Wei and Hoganson 2006).

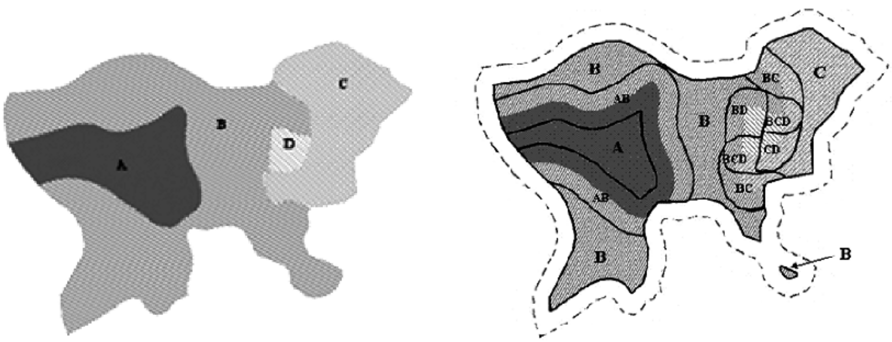


Fig. 4 Left: Stands in a simple 4-stand forest and right: corresponding influence zones. Note that the forest boundary is shown as a dashed line

4 Dynamic Programming for Coordinating Management Spatially

As noted earlier, mixed integer programming formulations have been studied extensively for addressing adjacency constraints. Often, if a mixed integer mathematical programming problem fits a DP structure, it can be solved more efficiently using DP solution techniques. DP offers flexibility for addressing various types of spatial considerations involving stand neighborhoods. Specifically, we will examine how influence zone information, as described earlier, can be used in a DP model to schedule core area production over time. We will also show how DP formulations can also be used to satisfy adjacency constraints. In all of these formulations, timber production will also be considered to be an important objective.

For the DP formulation presented here, we assume the forest has already been divided into a number of AA's or stands where each AA must be assigned a single management alternative. We will use the term stand, recognizing that AA's may actually be smaller than stands as defined by the forest inventory. For example, parts of stands that are located within a riparian corridor may be identified as separate AA's, as treatment options in riparian areas are typically more limited. Smaller AA's as the basic building blocks also offer greater flexibility for creating patch shapes more efficient for core area production. Generally, more efficient shapes would be compact – more circular in shape – with a greater ratio of patch area to patch edge. A basic assumption is that the forest is composed of many individual AA's (stands) where the spatial interdependencies of these AA's are important. To address adjacency constraints we will describe a unit restriction model (URM) where adjacent AA's cannot be harvested within a specified time period – the exclusion period. A DP formulation of an area restriction model (ARM) could also be developed, where sets of stands that violate adjacency size limits, as described by McDill et al. (2002) could be used to define spatial interdependencies involving adjacency violations.

Both timber production and core area production are considered in the objective function. As a result, both will be addressed through values assigned to arcs in the DP formulation. Adjacency constraints can be included in at least two ways. One approach is to not include treatments (arcs) in the DP formulation that would result in a violation of an adjacency constraint. Another approach is to place a high penalty cost on all arcs that violate adjacency constraints so that they do not become part of the optimal solution. In our presentation described below, we use this second approach (Hillier and Lieberman 1990, pp. 81–84). The definition of adjacent stands can be fairly flexible. Generally, adjacent stands share a common boundary. Definitions can be adjusted to include stands that are within a specified distance to each other, or to exclude adjacencies that involve only a short common boundary.

The DP formulation breaks the problem into stages with one stage for each stand. Each stage corresponds directly with the management decision for its associated stand. At each stage, state variables describe the condition of the forest in terms of decisions assumed to have been made about stands represented in earlier stages of the network. For each stage, each stand represented by an earlier stage is a potential state dimension. Once a stand is included in the network, its associated management decision must be carried as a state dimension until all decisions about stands it directly interacts with spatially have also been addressed. Otherwise, the formulation would violate the memoryless principle of dynamic programming. From a practical standpoint it is this requirement that makes the sequencing of stands (stages) in the network important because the sequence will influence model size.

In defining states for each stage of the problem, it is important to differentiate between stand management alternatives and stand management alternative types. Fortunately, the size of the DP formulation will depend on the number of alternative types and not on the number of alternatives. An alternative type is a group of one or more management alternatives that are identical in terms of spatial considerations. For example, one type might be all alternatives that have only one final harvest (harvest at end of rotation) and it occurs in planning period 6. This group might be relatively large with alternatives varying in terms of the timing and intensity of intermediate harvests (thinnings). Depending on the length of the planning horizon, even the number of alternative types can be large. Definitions of alternative types assume a Model I structure (Johnson and Scheurman 1977) where each type represents a sequence of activities over the entire planning horizon.

At the start of each stage, a "state" (node) exists for every unique combination of alternative types for all stands used as state variables. If each stand is spatially interdependent with all other stands in the forest, then the DP network would simply be a decision tree that grows in size as more stands are introduced. The DP network differs from a decision tree in that branches grow back together after stands associated with earlier stages are dropped as state variables. As described earlier, a stand is dropped as a state variable after all stands associated with it spatially have been included in the network. Each arc in the network represents a unique alternative type for the stand corresponding with the stage. Each arc represents the best alternative for that alternative type where best is defined as the alternative with the highest net present value. Decisions made in earlier stages of the network are carried as state descriptors so that spatial interdependencies can be evaluated in later stages of the network.

The influence zone information described earlier is used to help track and value core area. Each influence zone is addressed in the objective function value for the arcs associated with last stage in the formulation that corresponds with one of the stands defining the zone. For example, if an influence zone is defined by stand #3, stand #6 and stand #8, and these stands correspond with stage #14, stage #18 and stage #16, then this influence zone would be addressed in the arc values associated with stage 18. Each state (node) at the start of stage #18 would have an associated alternative type for stand #3 and stand #8 and the arcs for stage #18 each have an associated alternative type for stand #6. Similarly, a specific adjacency constraint can be evaluated at the last stage in the network that corresponds with the management decision for one of the stands associated with that constraint. Note that neither an influence zone nor an adjacency constraint can be evaluated at any earlier stage in the network because information is needed concerning the management decision for all of the stands associated with this specific spatial interdependency.

Either forward or backward recursion can be used to solve the DP formulation. Similar to DP formulations of the stand-level thinning problem the graphical construction of the problem is easier to conceptualize using a forward approach, as the set of stands represented in earlier stages impact the set of possible states in later stages. However, solving the mathematical formulation is easier to describe using a backward approach, as then the decisions associated with each node correspond directly to the set of management options associated with the stand corresponding with the stand (stage).

A detailed mathematical description of the DP formulation would be complicated as the number of state dimensions varies between stages. Below is a general representation with the definition of the state variables (s_i) kept simple.
Find:

$$f_1^*(1) \tag{1}$$

where

$$f_i^*(s_i) = \underset{(d_i)}{Max} \left[r_i(d_i) - p_i(s_i, d_i) + c_i(s_i, d_i) + f_{i+1}^*(s_{i+1}) \right] \tag{2}$$

$$s_{i+1} = s_i \otimes d_i \tag{3}$$

$$f_{N+1}^*(s_{N+1}) = 0 \tag{4}$$

and where

$d_i =$	the alternative type decision options for the stand associated with stage i.
$N =$	the number of stands.
$s_i =$	the state – a unique combination of alternative types for all stands addressed earlier in the network that are adjacent to either the stand associated with the current stage (stage i) or to a stand associated with a later stage (stage $i+1$ to N).
$r_i(d_i) =$	the net present value of stand i from timber production if decision d_i is made regarding its management.
$c_i(s_i, d_i) =$	the benefit, in terms of core area production, associated with making management decision d_i for stand i if the stands used as state variables for stage i are managed according to state s_i.
$p_i(s_i, d_i) =$	the adjacency penalty associated with making management decision d_i for stand i if the stands used as state variables for stage i are managed according to state s_i.

The objective is to maximize the sum of returns from timber production and core area production over all stands as measured by $f_1^*(1)$. Equation (2) is the recursive relationship that describes the series of interrelated stand-level decisions. The value of $p_i(s_i, d_i)$ in equation (2) is either zero or Big M, depending on whether d_i violates any adjacency constraints associated with state s_i. The value of $\mathrm{c}_i(s_i, d_i)$ in equation (2) represents the value of core area production from all influence zones associated with stage i. Each influence zone is associated with the last stage in the network that represents one of the stands defining the influence zone. Equation (3), the DP transition function, is just general notation indicating that s_{i+1}, the specific state in stage $i+1$, depends on both the state in stage i and the decision made in stage i. When determining $f_n^*(s_i)$ there are decisions yet to be evaluated for n stands. Equation (4), indicates that before any decisions are made for any stands, the sum of returns is zero. Once the backward recursive relationships have been solved for successive stages to the point where $f_1^*(1)$ is to be determined, decisions have been addressed for all but the one stand associated with stage 1. At the beginning of stage 1 there is only one state. Multiple states are needed in the stages only to remember how other stands represented in earlier stages are assumed to be managed so that adjacency constraints and core area production can be addressed through $p_i(s_i, d_i)$ and $c_i(s_i, d_i)$ as a component of the arc values.

Notation in DP formulations can be somewhat confusing as the set of nodes associated with the end of one stage are also the same nodes associated the beginning of the next stage. With N stages there exist $N+1$ sets of nodes. The problem is thus similar to a typical fence where there are N sections of fence (stages) with $N+1$ fence posts (sets of nodes). In our notation, stage i is supported by posts i and $i+1$.

Figure 5 illustrates a portion of the DP network for an application to a small "checkerboard" forest. It is assumed that: (1) stands are adjacent if they share a common edge or point (diagonal stands are adjacent), (2) the buffer width for core area production is less than one-half the width of the stands, and (3) each stand has only two management type options, option a and option b. Stage numbers in this formulation correspond directly with stand numbers. Names for the nodes show both the stand numbers used as state variables and the assumed management decisions. For example, node 1a2b implies stand 1 is assigned to alternative type a, and stand 2 is assigned to type b. From the start of stage 1 until the start of stage 4 the problem grows like a decision tree. At the end of stage 4 stand 1 and stand 2 are dropped as a state variable, as stands that are adjacent to stand 1 and stand 2 have been addressed, and all influence zones defined by stand 1 and stand 2 are also not influenced by stands remaining to include in the network. With a wider buffer distance for core area, such influence zones may exist and then stand 1 and stand 2 would not be dropped as state descriptors for the start of stage 5.

In formulating the DP, the order in which the stands are sequenced has potentially a large impact on model size. Clearly, there is potential for the curse of dimensionality if many stands must be used as state variables. At the start of any stage of the formulation one might consider a "front" or boundary which separates the stands represented in an earlier stage of the network from those represented in a later stage. All stands along this front must be used as state variables. For the sequencing used for the example (Fig. 5) the front never exceeds 3 stands. Generally, the number of states will grow exponentially with the number of stands along this front. For this example, the front is either a vertical line cutting across the width of the forest (Fig. 5) or a simple staircase across the width of the forest. Sequencing the stands

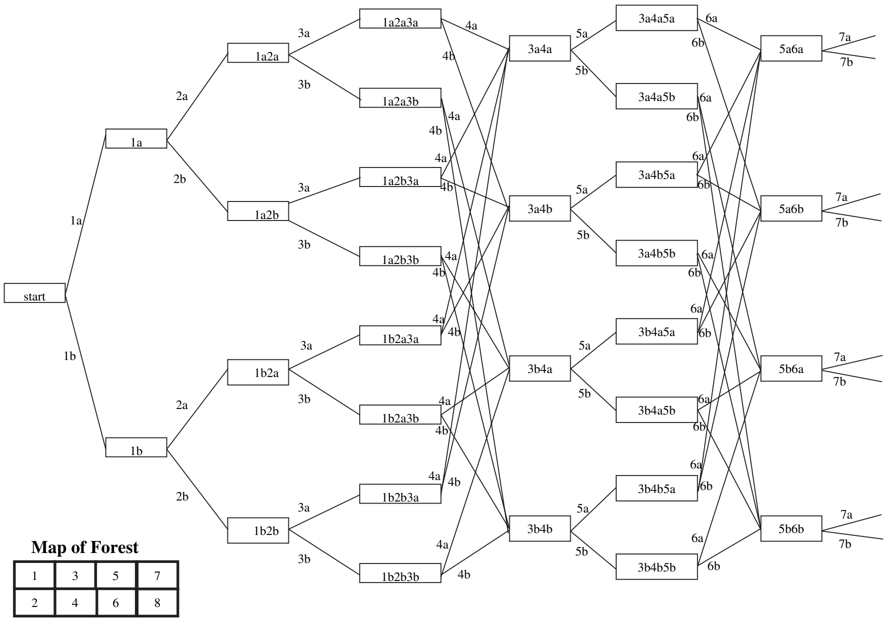


Fig. 5 An example of a portion of the DP network for a simple 8-stand forest where each stand has only 2 treatment options. Nodes are named to represent the implied treatment option for each stand number used as a state dimension

on the order presented for the example kept the front running across the width of the forest, and thus helped keep the DP formulation smaller.

It is the need to remember decisions made for stands considered in prior stages of the DP network that limits the size of problems that can be addressed in a single DP formulation. For example, in Fig. 5, there are no more than 8 nodes (2^3) at any stage because each stand was assumed to have only 2 management options. If each stand had 20 management options, then this example would have had 8,000 nodes (20^3) at the start of some stages. With currently computer technology, a problem with 8000 nodes per stage would still require minimal computer power, but seldom does one have forests that are only 2 stands wide. A forest with only 10 stands along the front with 10 options per stand would have potentially 10 billion (10^{10}) nodes at the start of the corresponding stage. Fortunately, the formulation can be combined with some simple heuristics to address large problems.

4.1 Overlapping Subproblems for Analyzing Large Areas

As described above, the size of the DP formulation increases substantially as the "width" of the forest is increased. Here we describe a method of decomposing a large problem into overlapping subproblems and solving the subproblems sequentially using DP. By overlapping subproblems, impacts of subproblem boundaries are not a critical concern. The approach is similar to the simple GIS concept of "moving windows" to analyze large areas. Windows are moved systematically across the forest with each window representing a subproblem that is solved with the DP formulation described earlier. Management decisions corresponding with the optimal solution for the DP formulation are not accepted for those stands that are close to other stands in the forest that are yet to have been analyzed. These stands will be analyzed in later windows. These stands represent the overlap between subproblems, as they remain in the widow as the window is moved. Figure 6 shows this basic moving windows concept for a simple "checkerboard forest" for four successive windows.

Tests using large forests have shown that the process finds near-optimal solution to adjacency problems with windows that are only two to three stands wide

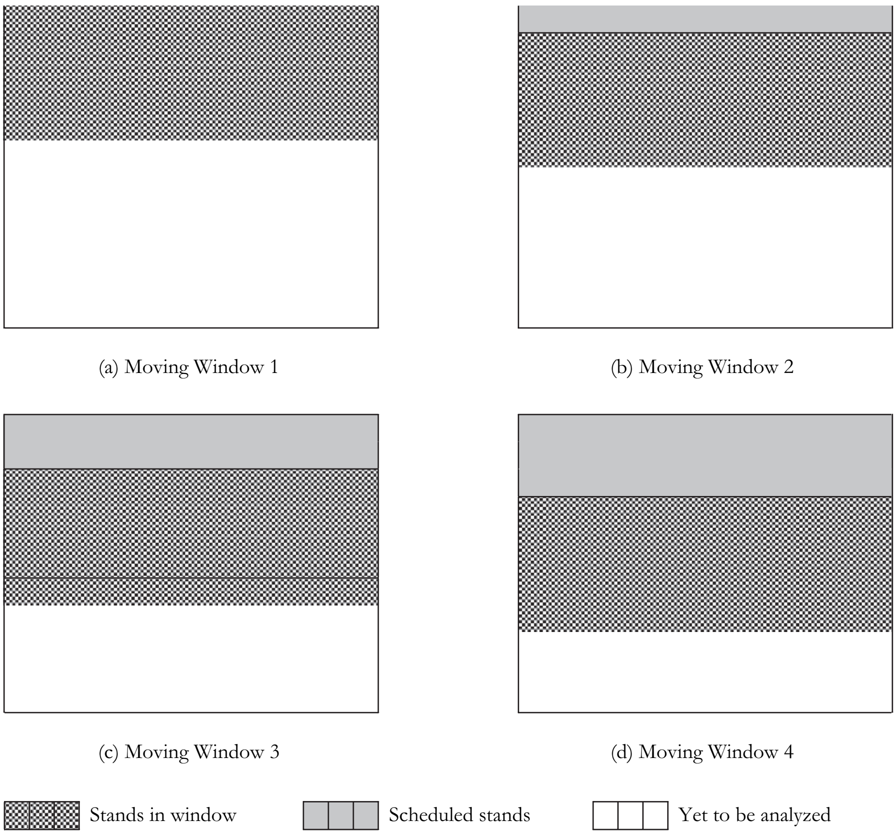


Fig. 6 Example of four overlapping subproblems for a simplified forest. Each window is a separate subproblem that can be scheduled using a DP formulation. Formulations are solved sequentially with the optimal management schedule accepted only for those stands not to be also analyzed within the next window

(Hoganson and Borges 1998). Windows widths of 6 or more stands are likely feasible with current personal computers. Designing the windows efficiently is an operations research problem in itself (Borges et al. 1999, Borges et al. 2002). Experience has shown only small gains from using windows more than 4 or 5 stands wide. As seen through the DP formulation, model size is sensitive to the number of treatment options considered for each stand. If model size is a problem, treatment options can be analyzed prior to applying the model to eliminate or trim out options for each stand that clearly will not be selected by the model (Hoganson et al. 2003). Including wider moving windows with large overlap between windows helps overcome problems associated with the sequential nature of this solution strategy.

5 Integrating the DP Model with Models to Address Forest-wide Constraints

A shortcoming of the DP-based spatial model is its inability to also directly address forest-wide constraints as are addressed in forest management scheduling models based on linear or mixed integer programming. The DP spatial model is based on the assumption that the aspatial value of each stand-level treatment option is known – the $r_i(d_i)$ values in formulation as presented earlier. These model input values need not be based strictly on timber returns. They provide an opportunity for a direct linkage with a forest-wide model to recognize forest-wide constraints.

Forest-wide scheduling models based on linear programming provide estimates of the marginal value of each stand type and the value of specific treatment options. This information is provided through the values of the dual prices associated with the constraints and reduced costs associated with each decision variable for each treatment option. Dual prices are estimates of the marginal rate of improvement in the objective function if the level of the associated forest-wide constraint were be relaxed by one unit. From an economics standpoint, this value can serve as the basis for adjusting the net present value estimate of an individual treatment option to recognize the impact of the option on the forest-wide constraints (Hoganson and Rose 1984; Paredes and Brodie 1989). For example, for a resource constraint that sets a maximum limit on the timber volume harvested in a period, the associated dual price acts like a tax or penalty on harvesting in the valuing system, recognizing that any harvesting at the stand level consumes some of the allowable forest-wide harvest. In contrast, for a forest-wide constraint that requires a minimum area of old forest, then the associated dual price acts like a subsidy or bonus by valuing stand-level treatment options that contribute to the old forest condition constraint. Forest-wide optimization models determine estimates of the dual prices that are necessary to apply in stand-level analyses so that the forest-wide constraints are achieved when results of optimal schedules from stand-level analyses are summed across the forest (Hoganson and Rose 1984).

Several factors make the problem challenging in a spatial context. First, the DP spatial model may be impractical to use for large problems if most stands require a large number of treatment options to be considered directly in the spatial model. The spatial model uses a model I format of linear programming where each treatment option for each stand potentially represents a series of rotations over the planning horizon (Johnson and Scheurman 1977). A model I format often results in many treatment options per stand. Second, if spatial aspects of the problem impact management decisions substantially, then the aspatial characteristics of the management schedule may also be impacted substantially, causing violations of aspatial forest-wide constraints. In other words, the dual price and reduced cost estimates from forest-wide aspatial model, if taken alone, are not likely accurate estimates of the marginal costs of the aspatial forest-wide constraints. Therefore, the estimates need to also take into account the spatial aspects of the problem on the forest-wide constraints.

To help keep the number of treatment options small for the spatial model without eliminating potentially optimal treatment options, a treatment trimming model can be used that utilizes both the results from the broader aspatial model and a detailed analysis of the influence zones that define the potential for each stand to contribute to the production of core area. Two basic factors can be considered for dropping a treatment option in this treatment option trimming process. First, treatments are dropped if crediting the treatment option for spatial benefits from all stands it influences spatially cannot raise its total value above the maximum aspatial value of the stand based on its best aspatial treatment option. Second, a treatment option can be dropped if there exists another treatment option that has a greater aspatial net present value and this other option also meets core area condition requirements in all time periods that the treatment option in question meets core area condition requirements.

To recognize the impact of spatial considerations on achieving aspatial forest-wide constraints, the spatial DP model and a forest-wide aspatial can be linked in an iterative process, essentially where the results of the spatial model are used by the aspatial model within its solution process for scheduling stands and estimating the dual prices of the forest-wide aspatial constraints. The iterative process is repeated until schedules are found that are "near-feasible" in terms of the forest-wide constraints. For each repetition (iteration), stand-level treatment options are re-evaluated and trimmed for use in the spatial model based on the updated forest-wide dual price estimates from the forest-wide model. Management schedules developed by the spatial model are easy to link with GIS systems for further detailed analysis (Wei and Hoganson 2005).

6 Applications for National Forest Planning in Minnesota

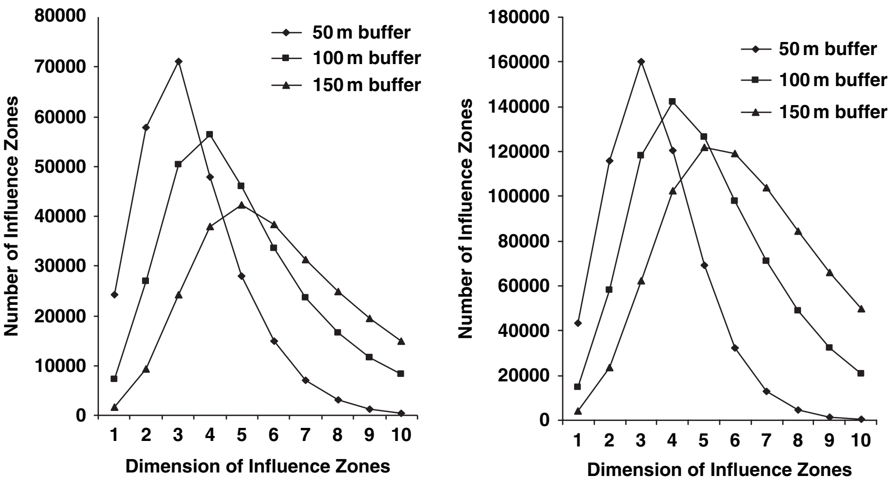
The DP spatial model has been used to analyze forest-wide alternatives for the two USDA Forest Service National Forests in Minnesota. These alternatives have focused on sustaining timber harvest levels over time while moving the forest closer to desired future conditions. The DP model was linked with Dualplan, (Hoganson and

Rose 1984) an aspatial forest-wide harvest scheduling model that decomposes the problem into parts, using the dual prices for the forest-wide constraints to link the subproblems. For the National Forest analyses, dual price estimates for the forest-wide constraints from the Dualplan model were used by the DP spatial model to value the impact of stand-level activities on forest-wide constraints. Dual price estimates were updated iteratively by linking summaries of the DP solutions with the Dualplan model. Forest-wide aspatial constraints emphasized desired future stand age class distributions for selected forest cover types in each of seven landscape ecosystems in the Chippewa National Forest and each of eight landscape ecosystems in the Superior National Forest. Multiple model formulations were considered to examine alternative desired future conditions, alternative values for core area of older forest and the rate at which the forest is moved towards the assumed desired future conditions. How the aspatial values are determined or re-estimated is beyond the scope of this chapter. Of emphasis here is an overview of the type of detail that can be recognized in a large-area real world application of the DP model that uses linked and overlapping subproblems.

For both national forests the planning horizon consists of ten ten-year periods. Analysis areas (AA's) are spatially explicit with over 67,000 AA's for the 549,000 acres modeled for the Chippewa National Forest, and with over 101,000 AA's for the 1,212,000 acres modeled for the Superior National Forest. Many AA's are small, representing substands in riparian areas. Seventeen silvicultural treatment types were recognized, representing various types of even-aged and uneven-aged management. For each treatment type, a range of harvest timings was considered. Twenty forest cover types were recognized for each forest with five timber-productivity site-quality classes for most forest cover types. Natural succession was modeled, recognizing that some stands change forest cover type over time without treatment. Estimates of pre-settlement age distributions and stand replacement fire intervals served as a benchmark for defining desired future conditions and the rate at which harvesting is applied to mimic stand replacement fires.

Forest cover type conversion (restoration) options were an important consideration for stand-level decisions because forest-wide constraints defined desired future forest-wide conditions in terms of a desired forest cover type mix. This added substantially to the number of treatment options modeled for most stands. Five key map layers influenced the set of treatment options considered for each stand. Besides the landscape ecosystem and riparian map layers described earlier, other layers included a map of management emphasis areas, a map of scenic classes and a map identifying sensitive areas for a number of plant and animal species. The map of management emphasis areas included 18 classifications used by the Forest Service to help define the theme of each alternative. For some forest-wide alternatives that emphasize older forest conditions, management emphasis area classifications limited treatment options substantially.

Frequency distributions (Fig. 7) showing the number of influence zones by influence zone dimension show a large number of influence zones, with most involving two or more stands. Not surprising, wider buffer distances for core area results in influence zones with greater dimensions (Fig. 7). In total, there are substantially



Chippewa National Forest Superior National Forest

Fig. 7 The number of influence zones by influence zone dimension for alternative buffer distances

more influence zones than AA's, averaging from about 3.5 to 7 influence zones per AA.

As one would expect, influence zones with higher dimensions, are smaller in area. Table 1 shows an area distribution of influence zones by influence zone

Table 1 Area distribution of influence zones by influence zone dimension for alternative core area buffer distances for the Chippewa and Superior National Forests

Dimension of influence zones	Area of influence zones in Chippewa National Forest (ha)				Area of influence zones in Superior National Forest (ha)			
	50 m buffer	100 m buffer	150 m buffer	100 m buffer W/L*	50 m buffer	100 m buffer	150 m buffer	100 m buffer W/L*
1	59853	11834	2716	8712	138419	34567	9976	34067
2	73195	35291	11383	28288	165023	92157	36447	91368
3	26566	36494	19879	30256	68602	90166	56740	86960
4	8157	21801	20653	18184	24074	59469	58072	54963
5	2884	10595	15277	8570	7931	32641	45541	29192
6	1232	5421	9477	4025	2787	18147	31714	15536
7	413	2981	5911	2024	781	10011	21553	8178
8	140	1687	3716	1043	229	5554	14363	4362
9	49	970	2413	573	55	3015	9371	2350
10	16	579	1597	340	11	1646	6051	1280
Sum	172506	127653	93023	102013	407913	347372	289828	328256
Percentage of forestland	77%	57%	42%	46%	87%	74%	62%	70%

* assuming core area requires a buffer form lakes and rivers.

dimension. Based on the number of influence zones, the most common influence zone dimension is 3–5 depending on the buffer width for core area (Fig. 7). In terms of the area of influence zones, the most common dimension is 2–3, varying by assumed buffer width. The Superior National Forest is more contiguous in terms of ownership and has a higher percentage of its area that can produce core area (Table 1). This difference is more pronounced for larger buffer widths, because with larger buffer widths the larger blocks are even more efficient than smaller blocks in producing core area.

A strength of a modeling approach based on influence zone data is the ability to process spatial information in great detail just once and then use that information repeatedly throughout the modeling process. Table 2 summarizes the distribution of influence zones in terms of influence zone size. Many of the very small zones are a result of the detail considered in recognizing riparian area detail.

Initial applications of the spatial model for the National Forest planning focused on a forest–wide alternative that was identified in the draft environmental impact statement (USDA Forest Service 2003) as the forest-wide alternative with the most

Table 2 Distribution of influence zone and sub-influence zone sizes for alternative core area buffer distances for the Chippewa and Superior National Forests

Category class range (ha)	Influence zone size distribution (count)			Sub-influence zone size distribution (count)		
	Buffer distance (m)			Buffer distance (m)		
	50	100	150	50	100	150
			Chippewa National Forest			
0–0.04	44025	55911	48922	92712	101909	82644
0.04–0.1	57146	71154	64608	116149	122166	105909
0.1–0.2	34174	40244	37181	62974	65570	58049
0.2–0.4	35351	40009	36802	73838	61998	53443
0.4–0.8	32832	31513	27083	37024	45012	35177
0.8–4	44620	38086	27726	43933	37010	25854
4–16	7325	3425	1988	4747	1763	814
16–80	396	124	42	386	79	17
> 80	4	1	0	4	1	0
Total	255873	280467	244352	431767	435508	361907
			Superior National Forest			
0–0.04	89646	139947	143837	194840	269654	252386
0.04–0.1	124232	184180	195601	267550	334274	329780
0.1–0.2	77792	108859	115233	151286	179148	183575
0.2–0.4	81479	108969	114013	159910	165716	165801
0.4–0.8	73818	82611	81299	81627	113547	103936
0.8–4	93781	94538	80019	95853	92214	75134
4–16	18110	10756	7267	11745	6199	3392
16–80	1396	555	276	1291	332	123
> 80	28	9	4	28	9	4
Total	560282	730424	737549	964130	1161093	1114131

fragmentation of mature forest over time. That alternative would also likely be the most complicated alternative to model because, on average, it is the least restrictive and thus involves more possible treatment options for each AA. Multiple model runs of the integrated modeling system were applied, varying the value of mature forest core area between applications. For each application, multiple iterations of the integrated model were needed to account for the impact of valuing core area on the marginal cost estimates for the many aspatial constraints that define desired future conditions for each landscape ecosystem and sustained timber harvest levels over time. For all the values of mature forest core area examined, the aspatial constraints were satisfied after multiple iterations of the integrated modeling system. Metric units were not used for the USDA Forest Service planning process. Core area was valued in terms of dollars per acre per decade with valuing based on conditions of the influence zones at the end of each decade. Scheduled harvests were assumed to occur at the midpoint of planning periods with spatial conditions thus measured at midpoints between times of potential harvest.

Results of the test applications were similar for both National Forests. Results are presented here only for Chippewa National Forest. Two types of core area were valued and tracked for both forests: upland forest core area and lowland forest core area. Both types had core area minimum stand age requirements that varied by forest cover types with a minimum age of at least 40 years for all forest cover types. Core area value assumptions impacted core area production levels similarly for both uplands (Fig. 8) Results for the $0 per acre per decade value are equivalent to applying the aspatial model alone without recognizing value of core area. Without valuing core area, core area levels decline substantially over the first two decades (Fig. 8). It should not be surprising that the draft environmental impact statement for the Minnesota Forest Plan (USDA Forest Service 2003) raises concern about potential declines in core area of mature forest. Raising the core area values to $100 per acre per decade raises the amount of mature area for both uplands and lowlands in the later decades, but there is still a decline in the early decades from the starting condition. A core area price of approximately $300 per acre per decade is needed before such a drop is not present in the early decades. For the core area values modeled, higher values for core area led to higher core area output levels for both uplands and lowlands, with levels substantially higher in the long-term than in the short-term. Increases are less in the short-term because of the time required to produce larger blocks of mature forest that are relatively well suited for producing core area. In past planning efforts producing core area of mature forest was not a primary objective, so it is not surprising that the intermediate-aged stands are not arranged spatially such that they can produce large quantities of core area when they soon reach the minimum age requirements for core area of mature forest. With more lead-time and planning, substantially more core area can be produced in the longer-term (Fig. 8).

Recognizing higher core area values tended to shift optimal management strategies from even-aged management to uneven-aged management. However, even-aged management was still predominant even at the highest core area values examined. Higher core area values also tended to lengthen rotation ages. As one might expect, higher values for core area resulted in larger patches of mature forest.

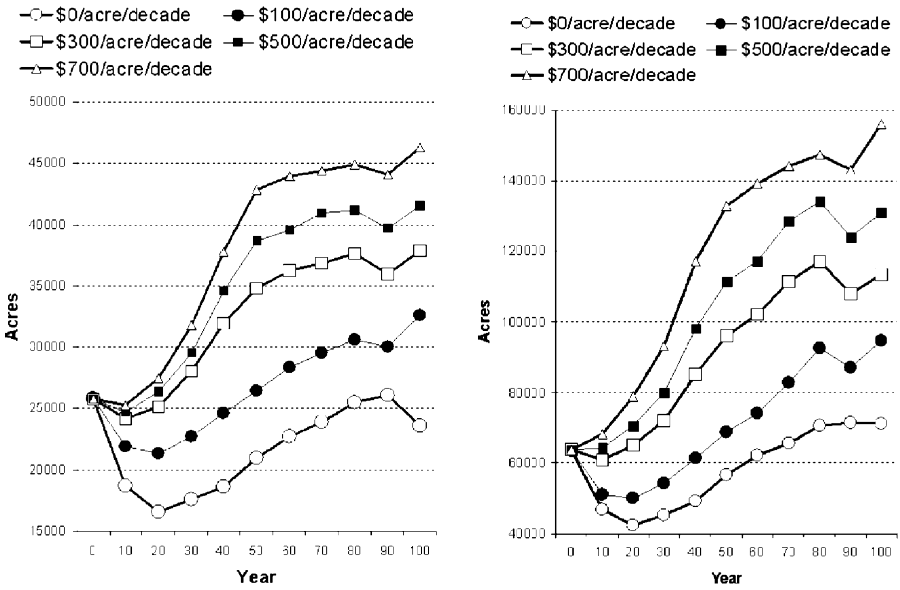


Fig. 8 Impact of core area values on core area production for five model runs for the Chippewa National Forest

It was used extensively to help address spatial detail important for the preferred management alternatives for both the Chippewa and Superior National Forests (USDA Forest Service 2004).

Results did not suggest that harvest blocks should be large in order to later produce more core area. Core area was more difficult to produce in the short- term, with large harvest blocks tending to reduce what core area is already present. Assuming core area production of mature forest is important, results strongly suggested that planning should focus more on what remains on the landscape in the short- term than on the size of the harvest blocks.

Model applications were performed using a Dell workstation with two 1.8 GHz processors and 512 MB of memory. Each iteration of DPspace took approximately one hour for the Chippewa National Forest and two hours for the Superior National Forest. Applications demonstrate the potential to integrate spatial and aspatial models in substantial detail with potential to address much larger than one might consider possible using optimization techniques. Early and intermediate iterations of the process can use smaller subproblems (moving windows) that solve faster. The ability to use current dual price estimates to screen and trim stand level alternatives is also a key component of the linking the modeling systems. Intermediate iterations of the integrated model are done simply to help find better estimates of the dual prices associated with the forest-wide aspatial constraints. Once good estimates are found, the DP model seems quite effective for finding near-optimal spatial solutions for the problem as it is formulated mathematically. The ability to decompose large problems into parts makes the system especially appealing. It suggests opportunities for large-scale collaborative planning involving multiple ownerships and model expansion opportunities to consider additional facets of the problem.

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Designing a Forested Landscape in Finland Under Different Climate Scenarios

Jordi Garcia-Gonzalo, Manfred J. Lexer, Heli Peltola and Seppo Kellomäki

1 Introduction

Forests provide multiple services beyond timber production, such as maintaining biodiversity, watershed and soil protection and recreation. An additional service provided by forests is what Jarvis et al. (2005) called "carbon forestry", which emphasises the direct role of forest management in maintaining forest carbon stocks and enhancing forest sink capacity for carbon. However, the amount and quality of services depends on the condition of forest resources and that is influenced by forest management which modifies the state of the forests. Interest in forest management has increased recently with the acknowledgement of the role of forests in the global carbon cycle through the adoption of the Kyoto Protocol. Moreover, it is known that management activities have several direct and indirect influences on the productivity of forest ecosystems and their carbon (C) sequestration potential (Karjalainen 1996a,b; Nabuurs and Schelhaas 2002). One of the main issues is that the preference of C sequestration in management may induce opportunity costs for timber production. Thus, it is important to investigate how timber production and C sequestration should be combined in order to balance these two management objectives in a sustainable way. Furthermore, to determine how a managed forest landscape should be shaped for simultaneous production of timber and C sequestration and for biodiversity.

Taking into account the long production period, traditional forest planning is based on the principles of constancy and long term stability (Gadow et al. 2007). In reality, periodic reorientation and frequent changes of forest policy are quite common. Furthermore, silviculture is influenced by the changing policies, as well as the economic and environmental conditions of the time. Therefore, silvicultural treatment programmes which are designed for conditions that are assumed to remain constant often turn out to be largely theoretical (Gadow et al. 2007). Surprisingly, the issue of optimising forest management under the conditions of climatic change has

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not attracted much attention. One of the few examples has been presented recently by Nuutinen et al. (2006), who employed linear programming to optimise timber production on a regional scale for a planning period of 30 years under conditions of climatic change.

In order to simulate forest development under the climate change, the selection of a proper model is important. It is necessary to understand the different modelling approaches and their main objectives and characteristics to decide the most convenient tool to be used for further scenario analyses. For example, empirical growth and yield models are widely used to support decision-making in forestry. Usually, these models utilise inventory data representing the past growth and development of a forest under specific growing conditions. The applications of such models in simulating the future growth and development assume that the future growing conditions are similar than in the past. Therefore, any changes in the growing conditions may bias the simulated growth and development. Optionally, one may use Gap or Patch models (Botkin 1993), which explicitly assess the impacts of temperature, water and nutrients on the growth and development of trees. However, the main goal of these models is to simulate vegetation patterns over time based on the regeneration, growth and death of individual trees and on the interaction between different tree species. The Gap models are used, for example, for assessing the potential vegetation patterns and changes in the vegetation distribution under the climate change. Nevertheless, the Gap models normally exclude physiological mechanisms linking the growth and development of trees with the climatic and edaphic factors. This may limit their applicability for impact studies compared to mechanistic models or process-based models, which include physiological response mechanisms to changes in environmental conditions (Waring and Running 1998).

Until now, the use of process-based models in forestry decision-making has been limited. This is because the applications of process-based models may need, for example, data not provided by conventional forest inventories. However, process-based models are able to provide the same prediction capacity under practical management situations as statistical models (Matala et al. 2003; Hyytiäinen et al. 2004). Moreover, process-based models help to understand how forests grow and develop under the current and/or changing climate (Mäkelä 1997; Landsberg and Waring 1997; Lindner 2000; Mäkelä et al. 2000a,b; Sands et al. 2000; Sievänen et al. 2000; Matala et al. 2006) and how management should be developed in order to avoid detrimental impacts and utilise the opportunities that may be provided by the climate change (Lindner 2000). Few studies, at the landscape or forest management unit (FMU) level, have been presented which deal with multi-objective forest management under the conditions of climatic change, and none of these have included optimisation of tactical management plans considering tactical planning as the production of a list of management actions (operations and treatments) that should be performed in the forest.

Although assessing the robustness of predefined management strategies under the conditions of climatic change may yield valuable general insights into the potential of adaptive management, the approach is unsuitable for identifying optimised management plans. In contrast, a standard approach is to apply stand growth models to provide production and state variables for each stand of a unit under a set of feasible stand treatment options, and then to apply optimisation methods to assign a treatment plan to each stand of the FMU in a way that the unit level objectives are met and decision constraints are satisfied. Different approaches have been proposed to accomplish this task (e.g. Hoganson and Rose 1984; Kangas and Hytönen 2001; Bettinger et al. 2002; Falcão and Borges 2002; Díaz-Balteiro and Romero 2003; Kurttila and Pukkala 2003). If multiple objectives have to be considered, the combination of multi-criteria decision making (MCDM) techniques with optimisation heuristics is frequently recommended (e.g., Pukkala 2002).

The objective of this contribution is to present a methodology for studying the effects of climate change on forest ecosystems using a process-based model. In addition the combined used of process-based models with multi-attribute-utility theory and optimisation methods are presented for designing a forested landscape to provide multiple services at the FMU level under climate change conditions.

2 Outlines of the FINNFOR Process-Based Model Used to Study Impacts of Management and Climate Change

2.1 Structure and Dynamics

The process-based model FinnFor was originally developed, by Kellomäki and Väisänen (1997), to simulate the development of Scots pine (*Pinus sylvestris*), Norway spruce (*Picea abies*) and silver birch (*Betula pendula*) stands growing in the boreal conditions. The model provides predictions on the photosynthetic production, growth, timber yield and carbon balance of stands in response to different climate conditions and different management regimes, thus predicting the effects of management under current and changing climate (Kellomäki et al. 1993; Strandman et al. 1993; Kellomäki et al. 1997a; Kellomäki and Väisänen 1997; Kramer et al. 2002; Matala et al. 2003).

The physiological core of the model is the biochemical sub-model of photosynthesis developed by Farquhar et al. (1980) and Von Caemmerer and Farquhar (1981) where the photosynthetic rate per unit area of foliage is calculated on an hourly basis (Table 1). The dynamics of the boreal forest ecosystem are directly linked to climate via photosynthesis, respiration and transpiration. Furthermore, hydrological and nutrient cycles indirectly couple the dynamics of the ecosystem to the climate change through soil processes, which represent the thermal and hydraulic conditions in soil, and the decomposition of litter and humus with the mineralisation of nitrogen.

Stocking controls the dynamics of the ecosystem through mortality and management by modifying the structure of the tree population. This results in changes in canopy processes and availability of resources for physiological processes and growth. The photosynthetic production is used to calculate tree growth and dimensions. Moreover, the gross photosynthesis over a year (as the sum of the hourly values) provides the total amount of photosynthates available for the maintenance

Table 1 Structure and properties of FinnFor model (see more details in Kellomäki and Väisänen 1997)

Main modelling objectives and management options	
Modelling objectives	Long term dynamics of forest ecosystem as controlled by environmental conditions (climate, soil) and management; boreal forests
Management options	Thinning and final cut; regeneration (natural regeneration, planting), nitrogen fertilisation, tree species choice; Scots pine, Norway spruce and birch spp.
Ecosystem structure	
Stand structure	Cohorts of single tree species in terms of number, age, height and diameter.
Tree structure	Foliage, branches, stem, coarse roots and fine roots
Soil structure	Litter on soil, soil organic matter (humus), mineral soil profile down to selected depth and divided up to ten soil layers
Model structure	
Model type	Mechanistic, deterministic
Time step	Hourly for physiological processes, annual for ecological and management processes
Environmental control by atmosphere	Radiation, temperature, precipitation, air humidity, wind speed, CO_2 concentration
Environmental control by soil	Soil moisture, soil temperature, available nitrogen
Functioning of the model processes	
Tree and stand level processes	
Photosynthesis	Biochemical model for photosynthesis driven by atmospheric and soil factors listed above
Autotrophic respiration	Day respiration and maintenance respiration controlled by temperature, growth respiration as a fraction of photosynthesis allocated to growth.
Stomatal conductance	Controlled by radiation, temperature, air humidity, CO_2 concentration, soil temperature and moisture (Jarvis type)
Transpiration	Penmann-Montheith type
Mortality and litter	Probability of an individual tree to die and stand level self-thinning, organ specific turn over rates for foliage, branches, course roots and fine roots
Seasonality	Temperature controlled dynamics in photosynthetic capacity, respiration and phenology, including frost resistance
Soil processes	
Temperature	Soil temperature controlled by radiation balance and physical properties of soil
Water	Soil moisture controlled by precipitation, evapo-transpiration and outflow of water
Nitrogen	Available nitrogen controlled by litter fall, decomposition of litter and humus and uptake of nitrogen by trees
Carbon	Dynamics controlled by heterotrophic losses under the control of soil moisture and temperature and quality of litter

Table 1 (Continued)

Main model outputs	
Water balance	Precipitation, evaporation, transpiration, runoff (surface and/or groundwater), available soil water
Nitrogen cycle	Uptake, deposition, litter fall, decomposition, available nitrogen
Carbon balance	Gross primary production, autotrophic respiration, heterotrophic respiration, carbon in trees and soil
Structure and properties of tree stand and harvested timber	Tree growth with ring width and stem form, including accumulation of biomass, trees and stand structure as described above. Harvested timber (logs, pulp), carbon in harvested timber

and growth of trees and their organs (foliage, branches, stem, coarse and fine roots). The allocation of the available photosynthates among the organs is based on the allometry between the mass of organ and total mass of the object tree (Matala et al. 2003). The mass of the stem and its growth is used to annually calculate tree height, diameter at breast height of the stem (DBH) (1.3 m above ground level) and the height of the object trees.

The rate of tree mortality is updated every five-years by calculating the probability of survival of trees in each cohort with regard to: (i) the stocking in the stand, (ii) the trees' classification (dominant, co-dominant, intermediate and suppressed) and (iii) the lifespan of the trees (Hynynen 1993; Matala et al. 2003). Litter from dead trees (dead organic material from any mass compartment of trees) including residues from cuttings are decomposed in the soil. Decomposition makes the nitrogen in the soil organic matter (litter and humus) available to the trees (Chertov and Komarov 1997).

2.2 Management

Management includes regeneration through planting, thinning and selection of the rotation length. In planting, the user provides the density of the initial stand and the distribution of seedlings into different size cohorts. Thinning can be from above or below. In the former case, mainly dominant and co-dominant trees representing the upper quartile of the diameter distribution are removed, and in the latter case suppressed and intermediate trees representing the lower quartile of the diameter distribution. Thinning disturbances increase litter on the soil in the form of logging residues.

The management in terms of thinning is based on the reduction of the basal area in the tree population. This reduction is converted into the number of trees (PF(i)) to be removed from each diameter class:

$$\mathrm{PF(i)} = \mathrm{VJ} \times \mathrm{RL(i)} \times (\overline{\mathrm{DBH}}/\mathrm{DBH(i)})^{\mathrm{HT}}, \tag{1}$$

where VJ is a factor used to balance the basal area after thinning to that defined by the thinning rate, RL(i) is the number of trees in diameter class i, $\overline{\mathrm{DBH}}$ is the mean diameter for trees in the stand weighted with the basal area before thinning, DBH(i) is the diameter in class i. The factor HT defines the thinning pattern and the relative reduction in the number of trees in diameter classes; i.e. values > 0 for the thinning from below and values < 0 for the thinning from above. Typically, the value of $\mathrm{HT} = -2$ for the thinning from above and $\mathrm{HT} = 2$ for the thinning from below are used. The values of factor VJ are increased iteratively step by step until the basal area of the stand after thinning is equal to the amount defined by the thinning intensity, with removal of trees from different tree cohorts.

Trees removed in thinning and final cut are converted to sawlogs and pulpwood. The concept of sawlog refers to the butt of the stem with a minimum diameter of 15 cm at the top of the log, while pulpwood refers to the other parts of the stem, with a minimum diameter of 6 cm. The remainder of the stem, in addition to branches and needles, represents logging residue. The amount (m^3) of different timber assortments from the stem are calculated based on empirical tables (Laasasenaho and Snellman 1983). These tables provide the amount of sawlogs, pulpwood and logging residue as a function of the breast height diameter and tree height.

Thinning reduces foliage area (LAI) and this reduction is linearly correlated to the foliage mass in the removed trees. The foliage area of trees represents the foliage mass converted to the area by utilising the values of the species-specific specific foliage area (area of foliage per mass unit (SLA), $\mathrm{m}^2\ \mathrm{kg}^{-1}$) presented by Ross et al. (1986), Kull and Niinemets (1993) and Niinemets and Kull (1995) for different species. Recovery of LAI is a function of the thinning regime and the growth of remaining trees. Thinning controls, also through LAI, the transfer of radiation in the canopy and water to the soil. Furthermore, thinning disturbances increase litter on the soil in the form of logging residues, thereby increasing nitrogen availability after litter decomposition.

2.3 Performance

FinnFor has been parameterised from long-term forest ecosystem data and climate change experiments (see Kellomäki et al. 2000), and successfully evaluated in various ways; (i) model validation against growth and yield tables (Kellomäki and Väisänen 1997), (ii) validation against measurements of short-term stand-level fluxes of water and carbon monitored by means of the eddy covariance method, along with (iii) model evaluation against five other process-based models (Kramer et al. 2002) and (iv) model evaluation against measurements of the growth history of trees in thinning experiments (Matala et al. 2003). Furthermore, parallel simulations using FinnFor and the statistical growth and yield model Motti have been carried out and model predictions have been compared (Matala et al. 2003; Briceño-Elizondo et al. 2006). These analyses show that the physiological model used in this study is capable of simulating the growth and development of stands under current climate in a similar way than typical management models (statistical models). Moreover, climate sensitivity analyses have been carried out using FinnFor to evaluate the effect on forest growth (Lindner et al. 2005; Briceño-Elizondo et al. 2006).

2.4 Impact Analyses

As previously explained, management and climate are variables introduced in the model for further simulations. Thus, it is possible to make sensitivity analyses dealing with these two variables both at stand and landscape levels. For example, studies have been presented where the impacts of the forest management regimes and the climate change on C stocks and timber production were assessed at the stand level Briceño-Elizondo et al. (2006) and at the landscape level (or management unit level) (Garcia-Gonzalo et al. 2007a,b). The growth of stem wood and timber yield (saw logs and pulpwood) were analysed as timber productivity. The total, for both growth and yield, were calculated for the 100 year simulation period $(\mathrm{m}^3\ \mathrm{ha}^{-1})$. The variables that characterise C stocks in the forest ecosystem were the C stock in trees (C in the above and below ground biomass) and the C stock in soil and were calculated in terms of the mean C storage over the simulation period $(\mathrm{Mg\ C\ ha}^{-1})$. In addition, the total C stock in wood products can be calculated using a wood products model.

Several other process-based models have been presented to study forest growth and dynamics under the current and/or climate change. Among others, Thornley and Cannell (2000), Mäkelä et al. (2000a) and Gracia et al. (2005) have developed mechanistic growth models to be applied in solving stand-level management problems. On the other hand, some features of process-based models have been integrated into empirical growth and yield models with a success to have a solid ground for stand-level simulation for forestry purposes (e.g. Peng 2000; Porte and Bartelink 2002; Matala et al. 2006; Peng et al. 2002b).

For example, Matala et al. (2005) used FinnFor model to introduce the effects of temperature and CO_2 elevation on tree growth into a statistical growth and yield model (Motti). For this purpose, they developed species-specific transfer functions based on the FinnFor model to describe the increase in stem volume growth as a function of elevating T and CO_2, stand density and the position of trees in stands of Scots pine (*Pinus sylvestris*), silver birth (*Betula pendula*) and Norway spruce (*Picea abies*). Furthermore, Nuutinen et al. (2006) introduced the transfer functions into the large-scale forestry scenario model (MELA) optimised forest management in regard to maximum sustainable timber production under changing climate. To our knowledge, the study by Nuutinen et al. (2006) is a first attempt to optimise forest management at the regional level under varying climate scenarios.

3 Example of Heuristic Optimisation Under Climate Change for a Forest Management Unit

3.1 Outlines

The example presented by Garcia-Gonzalo et al. (2007c) demonstrates the combined use of a process-based growth model, a wood products model, and a multi-objective optimisation heuristic in the analysis of optimised management plans for a forest management unit (FMU) under changing climatic conditions. There is a basic understanding that not only one, but a variety of treatment plans (i.e. stand treatment programmes) may be suitable for each stand. Thus, generating multiple options and evaluating them with regard to the services that they produce is an important task. Moreover, designing a forested landscape involves searching for a combination of management paths which provides a desirable mix of services to the landowner (Gadow et al. 2007).

In multi-objective forest planning the problem is modelled in such a way that the planning model takes into account several simultaneous management objectives. There are different planning model types (Pukkala 2002): (i) one objective is minimised of maximised in the objective function and the other goals are controlled through constraints. This corresponds to the use of linear programming (LP), (ii) the LP is modified so that the objective function measures the deviations of several objective variables from their target levels. These target levels are presented in other equations of the problem formulation and strict constraints may be added to the problem formulation. This corresponds to the use of goal programming (GP), (iii) a single objective variable appears in the objective function, which is augmented with a penalty function. This penalty function has the same unit as the objective variable and measures how much a set of additional objective variables deviate from their target values, (iv) a multi-attribute utility function is developed and used as the objective function. Mathematical programming (LP, GP) are typically used as a solution method for the formulations i and ii whereas heuristics are used for solving formulations (iii) and (iv). However, any problem that can be solved with mathematical programming can also be solved with heuristics. Heuristics, unlike exact methods, do not guarantee optimal solution. But the advantage of the heuristics is that they can solve very complicated problems.

For the multi-objective planning model, Garcia-Gonzalo et al. (2007c) applied multi-attribute utility theory. They developed a multi-attribute utility model which was used as an objective function to be maximised with a heuristic optimisation procedure. The management unit used in the study consisted of 1018 stands; each having six alternative stand treatment programmes (STPs) called "treatment paths" by Gadow et al. (2007). Thus, the total number of possible plans was 6^{1018}.

3.2 Input Data for Simulations

3.2.1 Forest Management Unit

The management unit used in the study was located near Kuopio in eastern Finland (63°01′N 27°48′E, the mean altitude is 94 m above sea level). The area is characterised by small, privately-owned forest properties, with an average area of 30–50 ha. These forests have always been of crucial importance for the local people by providing work and income. During the last 300 years, these forests have been utilised quite intensively, first in shifting cultivation and nowadays in forestry for timber production. This forest landscape is mosaic of stands and numerous small lakes.

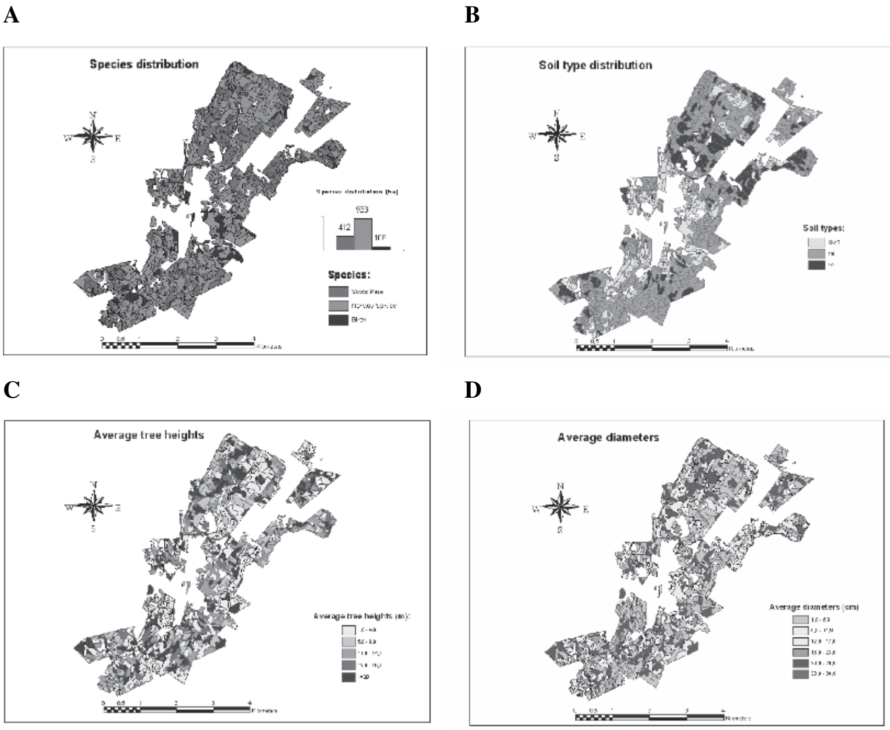

Fig. 1 Thematic maps for (A) Species distribution, (B) Soil type distribution, (C) Average tree heights and (D) Average tree diameters in the forest management unit

The study area (FMU) included 1018 separate tree stands covering 1451 hectares (based on inventory data from 2001) (Fig. 1). The tree stands dominated by Norway spruce accounted for 64% of the total area (933 ha), while Scots pine dominated stands covered 28% (412 ha) and silver birch dominated stands 7% (106 ha). The sites were of *Oxalis Myrtillus* (OMT), *Myrtillus* (MT) and *Vaccinium* (VT) types (Cajander 1949). Most of the stands were located on sites of medium fertility (MT 621 stands, 876 ha). A total of 170 stands were on the poor sites (VT, 275 ha) and 227 stands on the most fertile sites (OMT, 300 ha). The dominant tree species on the fertile sites (OMT, MT) was Norway spruce, whilst on the poor sites (VT) Scots pine was the most abundant species. In the inventory each stand was described in terms of the dominant tree species, average stand age, mean height and mean DBH (both weighted by basal area), stand density and site type.

3.2.2 Climate Scenarios

Three climate scenarios, each lasting 100 years, were used for the stand simulations: the current climate and two transient climate change scenarios. The current climate was represented by detrended weather data from the period 1961–1990 which was repeated consecutively to cover the entire 100-year simulation period. The two future climatic scenarios were generated for each site based on the predictions from global circulation models (GCM), ECHAM4 and HadCM2. The first climate change scenario was based on the output from the HadCM2 GCM (Erhard et al. 2001; Sabaté et al. 2002). The second climate change scenario was based on ECHAM4 scenario data compiled by the Max Plank Institute, Hamburg, Germany. Data for both scenarios were based on the greenhouse emission scenario IS92a (Houghton et al. 1990). Spatial interpolation of the GCM climate parameter anomalies for the study site was done using Delaunay triangulation. The daily weather statistics from the different climate scenarios were scaled down to an hourly basis using the weather simulator (Strandman et al. 1993). The climate data for the study were provided by the Potsdam Institute for Climate Impact Research (see Kellomäki et al. 2005a).

In the scenario representing the current climate, the annual mean temperature and precipitation for the period 2071–2100 were 3.1°C and 478 $\mathrm{mm\ yr^{-1}}$, respectively. Under the HadCM2 scenario, for the same period, these figures increased by 4.2 °C and 85 $\mathrm{mm\ yr^{-1}}$. Under the ECHAM4 scenario, the increase was 5.5 °C and 113 $\mathrm{mm\ yr^{-1}}$ with different seasonal allocation within the year compared to the HadCM2 scenario (Fig. 2). Table 2 shows the changes in temperature and precipitation in the climate change scenarios (ECHAM4 and HadCM2) related to the current climate based on 10-year averages the under current climate and under the climate change scenarios.

Under the current climate, the CO_2 concentration was 350 ppm, whereas in addition to the increase in temperature and rainfall, the HadCM2 and ECHAM4 scenarios presupposed a gradual and nonlinear increase up to 653 ppm over the simulation period (2000–2100). This increment in CO_2 concentration (CO_2) during the early phase of the simulation was smaller than that in the latter phase and followed the Equation (2)

$$\mathrm{CO_2\,(t) = 350^*\,exp\,(0.0063^*\,t)} \qquad (2)$$

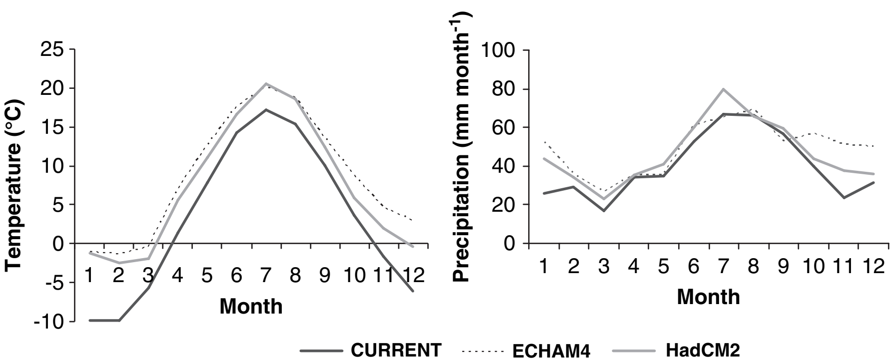

Fig. 2 Mean monthly temperature (°C) and precipitation (mm) in the last 30 years (2071–2100) of the simulation period for the three climate scenarios used in the study

Table 2 Changes in temperature ($°C$) and precipitation (mm) in the climate change scenarios (ECHAM4 and HadCM2) related to current climate scenario for the period (2001–2100). The differences are based on 10-year averages under current climate and under a climate change scenario

Period	Δ Temperature (°C)		Δ Precipitation (mm)	
	ECHAM4	HadCM2	ECHAM4	HadCM2
2001–2010	+1.1	+0.8	−2	−55
2011–2020	+2.0	+1.2	+93	+55
2021–2030	+2.2	+1.7	+46	+81
2031–2040	+2.6	+1.6	−28	−65
2041–2050	+4.1	+3.1	+53	+37
2051–2060	+3.4	+1.9	+35	−15
2061–2070	+4.7	+4.0	+50	+53
2071–2080	+4.2	+2.9	+185	+61
2081–2090	+5.2	+4.1	+205	+109
2091–2100	+6.5	+4.4	+62	+110

where t is year of simulation and 350 ppm is the initial CO_2 concentration in first year of simulation ($t = 0$, year 2000). Relative humidity and radiation were not affected by the scenarios.

3.2.3 Forest Management Alternatives

The species and site specific management rules recommended before 2006 were used to define the business-as-usual stand treatment plans (Basic Thinning BT(0,0)) (Yrjölä 2002). The new rules apply the same principles than the previous ones with small modifications, e.g. in thinning intensity and timing of terminal cut. The recommendations employ the dominant height and basal area for defining the timing and intensity of thinning (Fig. 3). Therefore, the cuts are not done at pre-defined years; whenever a given upper limit for the basal area (thinning threshold) at a given dominant height is encountered, a thinning intervention is triggered leading to a reduction in stocking to a level not lower than a given value (basal area after thinning). This cycle repeats until the trees forming the stand attain maturity for final cutting (i.e. age or DBH limit). Thus, the timing of thinning was adjusted to the growth and development of the tree population to take place before the occurrence of mortality due to crowding. In this work, stands with a dominant height $\geq 12\,\mathrm{m}$ (threshold to allow thinning) were thinned from below and trees removed to achieve the basal area recommended for the respective dominant height. Prior to reaching a dominant height of 12 m, trees are susceptible to natural mortality as a result of crowding. Random mortality occurred before and after the threshold value of the dominant height. In order to simplify the calculations, the thinning rules for the medium and most fertile site types (Yrjölä 2002), comprising 83% of the area, were used for all stands in the simulations.

As stated previously, a variety of treatment programmes may be suitable for each stand. In the study, to achieve various STPs, the basic thinning (BT(0,0)) given in management recommendations (Yrjölä 2002) was varied by combining the changes

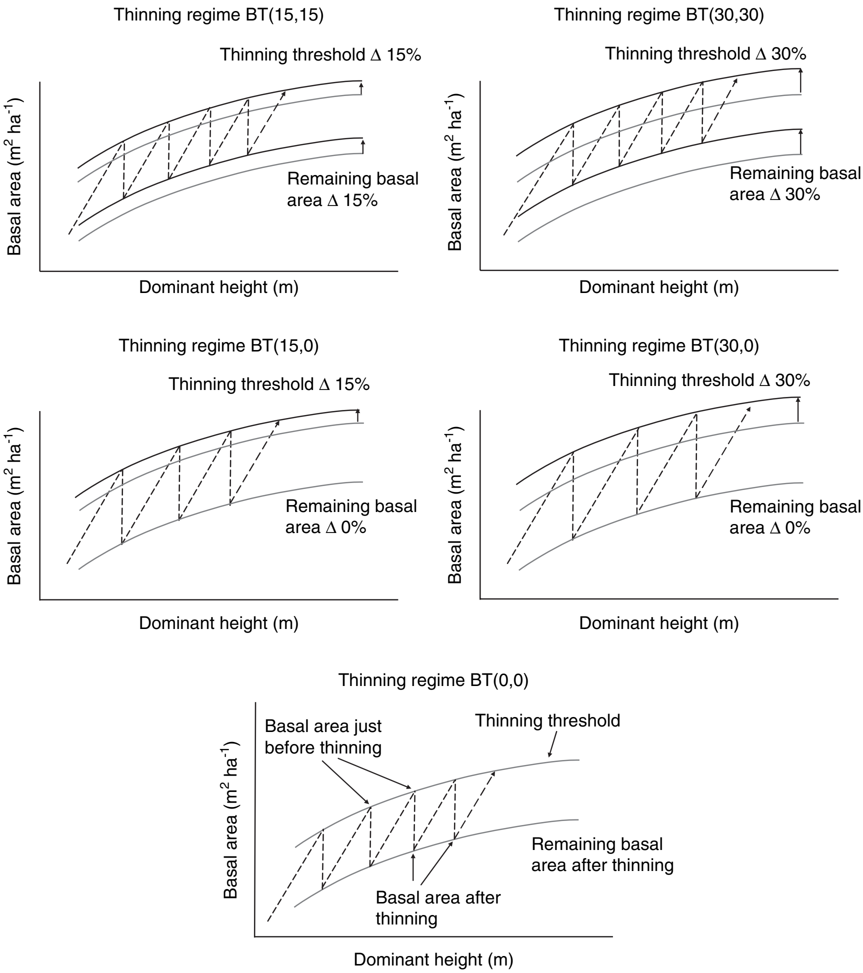

Fig. 3 Principles defining the thinning regime with the help of development of dominant height and basal area. The figure includes all the different thinning regimes used in the analysis. *Grey lines show the limits used for the Basic Thinning (BT(0,0))

in thinning threshold and the remaining basal area after thinning as explained in Garcia-Gonzalo et al. (2007a) and Briceño-Elizondo et al. (2006). The basal area that triggers the thinning and the remaining basal area after thinning can be varied in many ways. Therefore, in order to limit the final number of the thinning regimes to a practical level, a preliminary analysis at the stand level was completed. A matrix of 25 STPs was constructed by combining the changes in the basal area remaining after thinning and the thinning threshold, both with the variables 0%, $\pm 15\%$ and $\pm 30\%$. Then, we simulated the development of Scots pine, Norway spruce and silver birch stands (with 2500 saplings ha^{-1}) growing on MT site type during 100 years with a fixed final clearcut at the end of the simulation period. In addition, each of the species was simulated without thinnings, with only clearcutting at the end of simulation period.

According to the analysis, only a limited number of STPs were practical in terms of total stem wood production; i.e. the total stem wood growth was not less than that obtainable with the current recommendations (business-as-usual). Furthermore, the programmes with an excessive number of thinnings and a small volume of harvested timber were excluded. In such cases, the economic profitability was expected to be very low for any forest owner and/or forest company (based on stumpage prices). The only thinnings that fulfilled these criteria were those where the upper limit that triggered thinning was increased, either alone or concurrently with the remaining basal area (compared to current recommendations). Therefore, a total of six STPs for each species (five thinning regimes and one unthinned) were used for further analyses. These five thinning regimes were (Fig. 3): (i) Basic Thinning BT(0,0), following the management recommendations used until now for Finnish forestry (Yrjölä 2002), (ii) two regimes based on increasing the thinning threshold by either 15% (BT(15,0)) or 30% (BT (30,0)), allowing further growth before thinning and (iii) two other regimes combining both an increase in the thinning threshold and an increase in the remaining basal area in the stand after thinning by 15% (BT(15,15)) or 30% (BT(30,30)), allowing higher stand stocking in the forests over the rotation. Additionally, a regime with no active thinnings over the rotation was simulated for all species, with only a final clear cut being applied (UT(0,0)).

The simulations covered a hundred-year period under the different forest management and climate scenarios. Regardless of tree species, site type, and management regime, the stand was clear-cut when it was 100 years old or earlier if the average DBH of the trees exceeded 30 cm (Yrjölä 2002). If a stand was clear-cut before the end of the simulation period, the site was planted with the same species that occupied the site prior to clear-cut. The initial density of planted stands was 2500 saplings ha^{-1} regardless of the site and tree species. Once the stand was established, the simulation continued until the end of the simulation period.

3.3 Projection Time Windows and Initialisation of the Landscape

As previously described, for defining the timing and intensity of thinning (i.e., basic thinning) the management rules employ the dominant height and basal area. Thus, the timing of thinning is adjusted to the growth and development of the tree population. Therefore, as the STPs are adaptive to the effects of climate change on growth rate, a longer time than the normal planning period is required to get a clear climate change signal on the shares of allocated STPs. This advocates the use of a long planning period (100 years), which in turn calls for robust and reliable models to project the likely consequences of a plan.

The design of a forested landscape is necessarily based on data which describe the current state of the mosaic of stands. To initialise the landscape for simulations it was taken the real state of the management unit at year 2001.

3.4 Predicting Outputs from Models

3.4.1 Simulation of Forest Development by the Process-based-model FinnFor

In order to reduce the number of simulations, representative stands were selected from the forest management unit. First, all 1018 stands were classified into groups with the same dominant tree species (Scots pine, Norway spruce or silver birch), age class (10 year intervals) and soil fertility type (most fertile, medium fertile and poor sites). Second, a typical stand representing the normal growing situation was selected from each group. A total of 42 representative stands were selected for the simulations and their results were then extrapolated to each group.

The number of trees in each representative stand was distributed evenly over three cohorts, assigning to the first cohort the mean height and DBH from the inventory, for the second cohort those values were increased by 15%, while for the third they were decreased by 15%. In each representative stand the initial mass of organic matter in the soil was assumed to be $70\,\mathrm{Mg}\,\mathrm{ha}^{-1}$ which is representative for MT soils according to (Kellomäki et al. 2005b), these soils cover the main part of the study area. The value was based on calculations on permanent sampling plots all over Finland. The stands development was simulated over a 100 years using various management and climate scenarios. The data obtained from simulations for the 42 representative stands were then applied to the represented stands.

In this context, the net present value (NPV) of timber yield over the simulation period was calculated based on the average stumpage prices for different timber assortments for the decade 1990–1999 (Finnish Statistical Yearbook of Forestry 2001) and costs (e.g. planting and other regeneration costs) using the discount rate $\mathrm{p} = 0.02$. The discounted stumpage value of the standing stock at the end of the planning period was included in the NPV. This information is not directly available as an output for FinnFor.

3.4.2 Calculation of Carbon Storage in Wood Products by Wood Products Model (WPM)

Based on the FinnFor simulations, harvested timber was used as the input into the wood products model (WPM) (Briceño-Elizondo and Lexer 2004) in order to calculate C resilience times within wood product categories. The wood products model tracks the flow of C in harvested timber through production processes and its subsequent storage in wood-based products until it is released into the atmosphere. The model operates on a yearly time step and requires input files containing Mg of C in timber yield, separated into different assortments. The C contained in the assortments is directed to several production lines (e.g. sawmill, plywood and pulp and paper industries) or used as fuelwood. The products are assigned to different lifespan categories and after the end of the product lifecycle are either recycled, deposited in a landfill or burnt for energy production. The structure of the WPM as applied in this study closely follows the conception and parameterisation from

Karjalainen et al. (1994) and Eggers (2002). The parameters for those studies were estimated based on data from the Finnish yearbooks of forest statistics and on an extensive parameterisation scheme for Europe based on the FAOSTAT databases (FAO 2000; Eggers 2002). Further details of the model are found in Briceño-Elizondo and Lexer (2004).

3.5 Optimisation of an Additive Multi-criteria Utility Model

3.5.1 Additive Utility Model

The simulations from the models (FinnFor and WPM) provided the input data for the development of multi-objective management plans. A multi-attribute utility model was developed to calculate a utility index for optional management strategies with regard to a set of objective variables. This approach requires the transfer of the original values of the objective variables onto a common scale of preferability. It is this common scale that eventually allows to aggregate utility values over several objective variables (i.e., the comparison of "apples and pears"). In general form the utility model can be described as follows (Eq. 3),

$$U = \sum_{j=1}^{n} w_j u_j(q_j) \tag{3}$$

where U is the total utility of a management plan, n is the number of management objectives, w_j is the relative weight (i.e., importance) of the partial objective (j) $(j = 1, \ldots n)$, u_j is a sub-utility function for objective j, and q_j is the value of objective variable j.

In this study the total utility of a forest management plan is composed of two components, the aggregated utility component at the stand level ($U(sl)$, Eq. 4) and the utility component at the unit level (U(ul)). Equation (4) calculates the utility $\mathrm{U(sl)_{io}}$ for each stand (o) under a specific treatment option (i) with regard to a set of management objectives (j),

$$U(sl)_{io} = \sum_{j=1}^{n} w_j U_{ioj} \tag{4}$$

where U_{ioj} are partial utilities, w_j is the relative weight (i.e., importance) of the partial objective (j) $(j = 1, \ldots n)$, respectively. The weights have to be non-negative and add up to 1. The utility of a stand treatment option is calculated from preference functions which map the preferability of an alternative with regard to a set of decision criteria characterising the management objectives on the dimensionless scale [0–1].

Partial management objectives (j) in the case study application were: timber production (TP), C sequestration (CS) and biodiversity (BD). The net present value of costs (silviculture, harvesting) and revenues from timber production and the mean annual timber increment over the simulation period (MAI) were used to characterise the timber production objective. The C sequestration criteria, C stock in the forest ecosystem (CS-F) and in wood products (CS-WP), were calculated as an average stock over the 100-year planning period $(\mathrm{Mg\ C^{-1}\ ha^{-1}})$. Biodiversity was represented by the amount of average annual fresh deadwood.

In calculating the total utility of a management plan, the constraints and objectives at the unit level have to be considered. As an example, in Garcia-Gonzalo et al. (2007c) the utility component at the unit level for a given management plan (l) $(U(ul)_l)$ was calculated with regard to the minimum required decadal timber harvest constraint $(\mathrm{TH_{min}})$ and the corresponding achievement index for the requirement of an even harvest flow over the planning period $(\mathrm{TH_{flow}})$.

Equation 5 combines the utility components at stand and unit level. The coefficients w_r represent the relative importance of each component. The stand level utilities are aggregated by an area weighted (a_{rel}) average over all stands of the FMU.

$$U_l = w_1 \cdot \sum_{o=1}^{1018} a_{rel,o} \cdot U(sl)_{io} + w_2 \cdot U(ul)_l \tag{5}$$

$$\sum_{r=1}^{2} w_r = 1 \tag{6}$$

Figure 4 shows examples for preference as well as achievement functions. When the number of candidate plans to be evaluated is high, as in the case of numerical optimisation, the use of preference and achievement functions allows the automated calculation of utility values as opposed to standard approaches in multi-criteria decision support such as pairwise comparisons or direct rating methods (Pukkala 2002). Estimation can be based equally well on expertise or subjective value information, or on objective measurements or information produced by empirical research (Kangas et al. 2001).

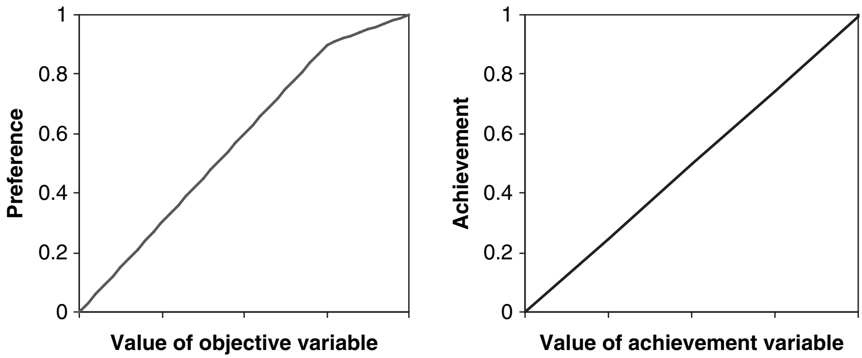

Fig. 4 An example of preference function at the stand level (left) and achievement functions at unit level (right)

The utility model can then be used as an objective function in numerical optimisation. In fact, the combination of numerical optimisation and utility theory is a much-used approach to multi-objective forest planning (Pukkala 2002). Thus, the utility model presented here was combined with a heuristic optimisation method in order to find optimal forest plans for the selected management unit. Even though heuristics, unlike exact methods, do not guarantee optimal solutions, in the following sections we refer to heuristically maximised management plans as optimised plans.

3.5.2 Optimisation

Numerous heuristics have been proposed for the optimisation of multi-objective forest management plans, the most commonly used in forest planning problems are: simulated annealing, tabu search, random ascent and genetic algorithms (Reeves 1993; Borges et al. 2002). In the example presented here, the utility function (Eq. 6) is maximised by random and direct search components. To start the heuristic optimisation process, one stand treatment programme (STP) is randomly selected for each stand to obtain an initial management plan. In the current study, this is repeated 500 times. The five random management plans with the highest overall utility are used as a starting point to continue with a direct search procedure. One stand at a time is examined to see whether another STP increases total utility. As an additional constraint at the stand level, the utility of a treatment had to be at least as good as the business-as-usual stand management to replace another treatment in the optimised plan. The rationale for this constraint is that the trade-off of utility at the stand level, where the value added of forest management is generated, for improved achievement values with regard to unit level constraints has to be limited.

Once all the STPs of each stand have been revised in this way, the process is repeated several times (cycles). In the study, 15 cycles were executed for each of the five initial random plans. The optimisation stops either after the last specified cycle or when no improvement in utility can be achieved over two consecutive cycles. To see whether the optimisation has been effective, a user-specified proportion of the STPs are replaced randomly after termination to check if the utility of the plan is increased. In its core features, this procedure is similar to the HERO approach as originally presented by Pukkala and Kangas (1993).

3.5.3 Scenario Settings and Analyses

In the presented case study application, three objective scenarios were analysed to represent contrasting views on forest management objectives. Two scenarios had a clear focus on a single objective; timber production (MaxTP) or C sequestration (MaxCS). The third scenario (multi-objective; MO) assumed equal importance for all management objectives. Tables 3 and 4 present all weight parameters of the utility model (Eqs. 3–6) as used in the analysis. In the example, the weight parameters were set based on expert consultations. The aggregated utility from the

Table 3 Weight parameters of the stand level component (Eqs. 3–4). NPV = net present value including discounted stumpage value of the standing stock in year 100 (p = 0.02), MAI = mean annual timber increment, CS-F = mean carbon storage in the forest ecosystem (above- and below-ground biomass of trees and carbon in soil), CS-WP = mean carbon storage in wood products, fDW = average fresh deadwood. W_1 = weight of stand level component in (Eq. 6). The objective scenarios were: MaxTP = timber production objective, MaxCS = carbon sequestration objective, MO = multi-objective scenario (timber production, carbon sequestration, biodiversity)

Objective scenario	W_1	TP	Criteria for TP		CS	Criteria for CS		BD	Criteria for BD
			NPV	MAI		CS-F	CS-WP		fDW
MaxTP	0.7	1.0	0.5	0.5	0.0	0.7	0.3	0.0	1.0
MaxCS	0.7	0.0	0.5	0.5	1.0	0.7	0.3	0.0	1.0
MO	0.7	0.34	0.5	0.5	0.33	0.7	0.3	0.33	1.0

stand level component was considered more important than the unit level component. In the objective scenarios MaxTP and MO, the unit level criteria were assigned equal importance; in MaxCS the even flow of timber harvests was not considered.

Management plans were generated for each of the three objective scenarios (MaxTP, MaxCS and MO) under current climate (current) and the two climate change scenarios (ECHAM4, HadCM2); in total nine objective/climate combinations. To indicate the within scenario variability regarding the share of selected STPs, five optimised plans were produced for each of the nine combinations of objective and climate scenarios. For each objective-climate combination, the solution with the highest overall utility was chosen as the best management plan. In contrast to the optimised plans, the objective functions for MaxCS, MaxTP and MO were also calculated for plans consisting of one specific STP exclusively. These STP-specific plans were compared with the optimised plans to identify the potential of mixing STPs over the unit. Moreover, the assignment of STPs in the optimised plans was compared among the different scenarios. Finally, to evaluate the potential for considering climate change in the optimisation, each management objective scenario was analysed by applying the management plan optimised for the current climate under the climate change scenarios.

Table 4 Weight parameters of the unit level components, THmin = minimum harvested timber per decade, THflow = even timber harvest flows. W_2 = weight of unit level component (Eq. 6). The objective scenarios were: MaxTP = timber production objective, MaxCS = carbon sequestration objective, MO = multi-objective scenario (timber production, carbon sequestration, biodiversity)

Objective scenario	W_2	Priority for achievement functions	
		TH_{min}	TH_{flow}
MaxTP	0.3	0.5	0.5
MaxCS	0.3	1.0	0.0
MO	0.3	0.5	0.5

3.5.4 Main Findings

Application of Same Stand Treatment Programme (STP) for all Stands

Regardless of the climate, the largest amount of timber harvested and also the highest NPV were found when BT(30,30) was used over the entire FMU. This STP allowed a higher timber stocking and later thinnings than BT(0,0) resulting in a higher proportion of logs. As a consequence, BT(30,30) produced the highest amount of C in wood products due to the long-lived nature of products obtained from sawlogs. On the contrary, the maximum amount of C in the forest was found when no thinnings were applied (UT(0,0)). Moreover, business-as-usual (BT(0,0)) always gave less NPV than all the other actively thinned STPs (only being superior to UT(0,0)).

Optimised Management Plans: Effect of Climate

The aim of the optimisation was to maximise aggregated preferences regarding various criteria. Five optimised management plans (5 repetitions) were generated for each of the nine objective/climate combinations (3 objective $\times 3$ climate scenarios). Chi^2 tests on the shares of STP by area yielded no significant differences within the 5 repetitions for each of the objective/climate combinations ($\alpha = 0.05$), indicating that despite the random initial conditions, the optimisation procedure was clearly converging towards scenario-specific optima. In Table 5 the plans with the highest total utility are shown. For a given climate scenario the optimised solutions (shares of STP by area) differed substantially between the management objective scenarios (chi^2 test significant at $\alpha = 0.05$). The results under the climate change scenarios contrasted somewhat to the results under the current climate. A generally

Table 5 Distribution of stand treatment programmes (STP) over stands (ha per STP) in optimised management plans for all objective/climate scenario combinations. MaxTP = timber production objective, MaxCS = carbon sequestration objective, MO = multi-objective scenario (timber production, carbon sequestration, biodiversity). Shown are the plans with the highest total utility. Chi^2 test was used to indicate if the differences in areas per STP were statistically significant ($\alpha = 0.05$). For further explanation, see the text

Objective scenario	Climate scenario	Hectares per treatment programme					
		BT(0,0)	UT(0,0)	BT(15,0)	BT(15,15)	BT(30,0)	BT(30,30)
MaxTP	Current	167.4	72.1	454.1	171.3	289.9	295.9
	ECHAM4	68.8	220.0	406.0	37.4	397.7	320.8
	HadCM2	124.0	189.5	214.4	30.7	506.8	385.3
MaxCS	Current	66.2	649.6	117.4	82.2	149.7	385.6
	ECHAM4	8.6	670.9	106.6	93.0	214.9	356.7
	HadCM2	11.0	668.6	118.2	30.7	236.4	385.8
MO	Current	64.4	875.5	268.1	15.8	108.3	118.6
	ECHAM4	1.9	835.1	186.3	29.5	299.3	98.6
	HadCM2	0.0	885.0	145.4	0.0	356.5	63.8

observed pattern was that under the climate change, in all objective scenarios, the share of some STPs were increased compared to the results under current climate. These STPs allow a higher stand stocking over the rotation and later thinnings and/or final cutting than BT(0,0). Chi^2 tests on differences in the share of STPs between climate scenarios within objective scenarios yielded significant differences ($\alpha = 0.05$).

The relative increase in the total utility of optimised plans due to the climate change differed somewhat between the objective scenarios. For MaxTP scenario, the maximum increase was 16.8% (ECHAM4), for MaxCS it was 9.9% (HadCM2), and for MO 11.3% (ECHAM4). This pattern was consistent with the results observed with the management plans relying on only one specific STP.

Optimised Management Plans: Potential Benefits of Adaptive Management

The optimised management plans for current climate were also used under the two climate change scenarios (ECHAM4 and HadCM2) and the results compared with the findings of plans specifically optimised for these two climate change scenarios. This was done in order to analyse how a management plan optimised for current climate performed under conditions of climate change. Using the plan for the current climate under the climate change scenarios decreased the utility at both the stand and management unit level when compared to the plan optimised for the climate change (Table 6). However, for some of the individual criteria the solution for the current climate gave even higher values than the specific optimal solution for the climate change conditions. Overall, due to the assumed trade-off relationship between stand level and unit level utility components, the use of an optimised management plan for a specific climate increased the total utility between 3.4% and 9.2%.

4 Discussion

Forests provide different services beyond timber production, such as C sequestration, maintaining biodiversity, watershed and soil protection as well as recreation. It is known that management activities have several direct and indirect influences on the timber production and C stocks (Karjalainen 1996a,b; Nabuurs and Schelhaas 2002; Garcia-Gonzalo et al. 2007a,b). Because the climatic conditions influence the growth and development of forest stands, it could be expected that the climate change will affect the dynamics of forest landscapes.

In this context, process-based models are valuable tools for studying forest growth and dynamics under the climate change. In recent years, several process-based models have been created, which have been applied successfully to study forest growth and dynamics under the climate change (e.g. Kellomäki et al. 1997a,b; Thornley and Cannell 2000; Mäkelä et al. 2000a; Sabaté et al. 2002). Most of these studies have focused on the assessment of how forests grow under the climate change by applying the current management practices, and mainly at the stand level. In the example presented here, a process-based model was used to study how

Table 6 Opportunity cost of not adapting management plans to climate change. The optimised plan under current climate is applied to climate change scenarios (ECHAM4, HadCM2) and compared with plans specifically optimised for the respective climate scenario (optEcham, optHad). NPV = Net Present Value including the discounted stumpage value in year 100, p = 0.02 [€ha^{-1}], MAI = mean annual timber increment [m^3ha^{-1} yr^{-1}], CS-F = mean carbon storage in the forest (above- and below-ground biomass of trees and carbon in the soil) [Mg ha^{-1}], C-WP = mean Carbon storage in wood products [Mg ha^{-1}], fDW = average annual fresh deadwood [m^3ha^{-1} yr^{-1}], THflow = coefficient of variation of decadal timber harvests [%], THmin = minimum harvested timber per decade [m^3ha^{-1}], U(sl) = aggregated stand level utility, U(ul) = aggregated unit level utility, MaxTP = timber production scenario, MaxCS = carbon sequestration scenario, MO = multi-objective scenario (including timber production, carbon sequestration and biodiversity). The plans with the highest total utility for each of the objective/climate scenario combinations are presented

Objective scenario	Climate scenario	NPV € ha^{-1}	MAI m^3ha^{-1} yr^{-1}	CS-F Mg ha^{-1}	CS-WP Mg ha^{-1}	fDW m^3ha^{-1} yr^{-1}	TH_{flow} (%)	TH_{min} m^3 ha^{-1}	U(sl)	U(ul)	Utotal
MaxTP	**Current**	**7850.7**	**6.3**	**113.4**	**10.3**	**0.4**	**29.7**	**46.6**	**0.4107**	**0.6930**	**0.4954**
	OptEcham	**8982.7**	**7.9**	**122.4**	**11.2**	**1.0**	**24.4**	**47.4**	**0.5139**	**0.7292**	**0.5785**
	ECHAM4	9120.5	7.8	114.1	12.0	0.6	36.2	46.7	0.5119	0.6551	0.5549
	OptHad	**8895.5**	**7.6**	**121.2**	**11.3**	**0.8**	**24.1**	**47.1**	**0.4993**	**0.7288**	**0.5682**
	HadCM2	8947.9	7.5	114.5	11.7	0.5	38.2	45.8	0.4955	0.6381	0.5383
MaxCS	Current	7274.5	6.1	133.7	9.2	1.6	57.2	37.7	0.4939	0.452	0.4814
	OptEcham	**8417.8**	**7.6**	**139.0**	**10.4**	**2.2**	**58.6**	**43.5**	**0.5290**	**0.5217**	**0.5268**
	ECHAM4	8421.5	7.6	138.1	10.6	2.1	59.0	38.8	0.5283	0.4653	0.5094
	OptHad	**8296.9**	**7.4**	**140.6**	**10.2**	**2.1**	**58.5**	**43.2**	**0.5336**	**0.5189**	**0.5292**
	HadCM2	8279.1	7.4	138.8	10.4	2.0	57.3	36.5	0.5291	0.4381	0.5018
MO	Current	7047.3	5.9	133.5	8.7	1.8	33.1	33.6	0.5497	0.5949	0.5633
	OptEcham	**8292.1**	**7.5**	**139.4**	**9.9**	**2.3**	**28.2**	**38.3**	**0.6165**	**0.6521**	**0.6272**
	ECHAM4	8115.9	7.4	139.1	9.7	2.5	39.9	32.1	0.6119	0.5459	0.5921
	OptHad	**8106.4**	**7.2**	**141.0**	**9.8**	**2.3**	**28.9**	**35.9**	**0.6175**	**0.6332**	**0.6222**
	HadCM2	7986.9	7.1	139.6	9.5	2.4	43.8	26.3	0.6040	0.4883	0.5693

Note: Bold figures mean that the numbers correspond to the plan specifically optimised for the respective climate.

management and changing climatic conditions affect the growth, timber yield and C in a forest management unit (FMU) located in central Finland (boreal forest). In addition, in order to identify optimised management plans under multiple objectives, the combined use of a process based growth model, a wood products model and an optimisation heuristic have been presented. The example presented here further develops previously presented approaches to multi-purpose management planning (e.g., Pukkala and Kangas 1993; Kangas et al. 2001; Kangas and Hytönen 2001; Pukkala and Kurttila 2005) by additional features. The novelty of the example shown in this paper is the use of a process-based model to make a multi-objective optimisation of forest management under the changing climatic conditions and the study of the importance of adapting the forest planning to climate change.

The impacts of transient climate change scenarios on currently existing forests have been addressed to a very limited extent and are usually based on business-as-usual management (e.g. Lindner 2000; Lexer et al. 2002). On the contrary, for the work presented here, six different species-specific stand treatment programmes were used. These differed from each other in the sense that mean stocking in the tree populations over the rotation was increased or decreased compared to business-as-usual management. This allowed the identification of how sensitive growth, timber yield and C stocks are to the management. In addition, three different climate scenarios (the current climate and two climate change scenarios) were used in order to study the sensitivity to the climate change. As previously described, these management recommendations are species-specific and for defining the timing and intensity of thinning (i.e., basic thinning) they employ the dominant height and basal area. Thus, the timing of thinning is adjusted to the growth and development of the tree population. As the STPs are adaptive, per se, to the effects of climate change on growth rate, a longer time than the normal planning period is required to get a clear climate change signal on the shares of allocated STPs. This justified the use of the long planning period used in the study (100 years).

In the example simulation, timber production, C sequestration and biodiversity were considered as management objectives. A utility model has been employed to combine all objective variables and constraints in an overall utility index, which was maximised by a heuristic optimisation method. In our approach, the management objectives are specified at the stand level and all stand treatment options are evaluated by means of criterion-specific preference functions. This approach requires additional criteria at the unit level to satisfy constraints such as liquidity demand or spatial considerations (i.e., habitat requirements). These unit level constraints usually make the objective function non-additive which in turn favours heuristics instead of mathematical programming techniques. One advantage of the employed approach is that interpretation of the model coefficients as relative weights of objectives and criteria is intuitively possible. This also makes the approach potentially suitable for multi-stakeholder planning situations in public participation (e.g., Kangas and Hytönen 2001; Munda 2004).

Heuristics commonly used in forest planning problems are: simulated annealing, tabu search, random ascent and genetic algorithms (Reeves 1993; Borges et al. 2002). In the study presented here, the optimisation heuristic used was similar to the HERO method as presented by Pukkala and Kangas (1993). This approach potentially carries the risk of getting trapped in a local optimum. However, Pukkala and Kurttila (2005) found that simple techniques such as HERO and random ascent are suitable approaches especially when spatial objectives are not included in the problem. The management unit used in this study consisted of 1018 stands, each having six alternative treatment programmes. Thus, the total number of possible different plans was 6^{1018}. This clearly shows that it is not practical to compare and evaluate all available alternatives. Instead, one relies on efficient numerical tools to search the decision space for feasible solutions.

A question, which one may ask, is why there is a need for an optimisation at the FMU level; why not use an optimal STP at the stand level for the entire FMU? As shown in the results, no solution generated by applying one treatment programme for the entire FMU seems acceptable in practice, even if that STP maximised one of the criteria. For instance, choosing one STP for the entire unit just because it maximised the NPV implied very uneven harvest schedules and low carbon sequestration. On the contrary, the STP that maximised carbon sequestration yielded a very low net present value, a very uneven flow of timber harvests and an extremely low minimum harvested timber volume per decade.

As expected, significant differences between the optimised management plans for the different objective scenarios were found. In addition, significant differences were also found between optimised plans for the different climate scenarios, indicating that climate does affect optimal planning solutions. Moreover, the optimal plan for current climate was applied under the two different climate change scenarios (ECHAM4 and HadCM2) and results compared with optimal plans for ECHAM4 and HadCM2 climate scenarios. This was done in order to analyse how a management plan specifically optimised for current climate performed under conditions of climate change. In our example, when the optimal plan for current climate was applied under the two different climate change scenarios (i.e climate change effect), the gain in total utility was between $+1\%$ and $+12\%$ depending on the objective and climate scenario. These gains are produced by the increased production due to climate change. In addition, the gain in total utility through optimising a management plan specifically to changing climatic conditions was between $+3.4\%$ and $+9.2\%$, depending on the objective and the climate scenario. This corresponds to the potential benefits of optimising management plans to climate change.

To summarise, in the light of these findings, it seems recommendable to include possible future climatic changes in forest planning. In addition, the study demonstrates that the combined use of process-based growth modelling and multi-objective optimisation heuristic is an efficient tool to support forest planning and decision making in identification of optimised management plans under multiple objectives and the climate change conditions.

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Part IV
Computer Graphics and Visualization

Visualization in the Design of Forested Landscapes

André O. Falcão

1 Introduction

Human pressure on the environment has led to successive and often dramatic landscape changes in the industrialized regions of the world. Large portions of the land surface have been modified and "humanized"; and original environments have been altered, sometimes beyond recovery. The "Natural Landscape" is, in the modern day urban society, more and more considered as a very valuable resource which is not commonly available. As people are looking towards their roots, searching for their identity, an appreciation of the rural space increases, as well as the desire for its unadulterated conservation, and closer to what is generally perceived as a pristine state. However, the presence of man in the environment causes changes, and it is often not possible to undo those changes. Careful planning must therefore ensure that ruinous environmental or landscape effects can be adequately anticipated and discussed before the envisaged activities take place. If, in a general situation, participation in the real world is irreplaceable by a computer simulation, mechanisms should exist that are able to provide to stakeholders the perception of a landscape in the future, derived from changing management visions, or simply to perceive how the landscape will evolve in accordance with known ecological dynamics. In forest management, pressure against change, sometimes catalyzed by intensive practices like clearcuts or the introduction of new species leads frequently to public revolt and protest. Participatory planning is a key to involve the public in landscape design; yet, it is generally difficult to convey the landscape impact of future silvicultural operations without adequate tools.

The perception of a global landscape is sometimes difficult to achieve at once, even when one is physically there. The landscape is not a painting and complete appreciation of all its characteristics requires direct involvement. Natural scenes offer many different views, and each scene may present an unknown yet aesthetically pleasing sight. What may appear to some as a dull and an uninteresting locale, may,

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if observed from a different point of view, present itself as an inspiring and beautiful environment. Specific computational tools may thus facilitate the comprehension of a landscape for management. During the last decades, public opinion has been shown progressively preoccupied with the visible effects of landscape and environmental changes (Luymes 2001; Sheppard 1989). There is a growing sensitivity of the general public for environmental and landscape issues, which the increased demand of tourism in rural areas tends to inflate.

Recent advances in computing resources and graphics hardware provide the functionality to integrate landscape visual elements within the forest management planning process (e.g. Wherret 2001; Pukkala 1998). Enhanced graphical data and visual information processing capabilities available in mass market PCs make it possible to anticipate landscape-level visual impacts of management plans with small hardware budgets. Progress in information processing during the last two decades further allowed the inclusion of landscape aesthetic factors as ways to impact traditional forest management models. Thus, it is currently possible to use common household computers to visualize actual landscapes and even future landscapes, years before their visual impacts become apparent. The works of McGaughey (1997), Orland and Uusitalo (2001) and Danahy (2001) show that it is possible to present the effects of different management strategies in images thus that the effects are promptly identifiable by end users (Luymes 2001). It is also currently feasible to compare the landscape effects of different management scenarios for different observation points, as well as execute virtual flights over prospective landscapes (e.g. Falcão et al. 2006).

Visualization of forested landscapes is a complex subject that can be approached from many different directions. For this chapter, it was decided to provide a broad non-technical view of the problem so that the reader may appreciate the major aspects involved in the subject. As such, initially the cognitive aesthetic basis for visualizing natural landscapes, with special emphasis on the forest, is discussed. Secondly, several systems that use forest visualization techniques are presented, and their respective capabilities and limitations are highlighted. Problems related to data requirements for a visualization system are also addressed. Finally, some specific comments are provided which relate to the integration of visualization tools within a decision support framework.

2 Landscape Perception and Aesthetics

People increasingly use images as a means of communication. Effective participatory planning therefore requires visual tools for communicating management plans to the general public, so that alternative strategies may be adequately perceived and aesthetically evaluated. Luymes (2001) argues that visual simulations are rhetorical products intended to persuade an audience, albeit not with the purpose of deceiving, but conscious of the fact that any product is conceived with a specific audience in mind. The author further maintains that such an attitude may be proclaimed openly, so that a more sustainable forest management plan might be better understood and perceived.

The environment is where we live and the majority of people have concrete and definite points of view abut it, as changes affecting it directly influence their lives. Further, when evaluating a landscape aesthetically, a stakeholder might not be formally objective as the sentiments related to the functional utility possibly gathered from a new situation may prevail over an impartial analysis. Daniel and Boster (1976) developed a scenic beauty estimation method that aims at harmonizing preferences between users and produces a ranking of preferred landscapes. As shown by Bruce et al. (1994) different people at different states of mind, depending on factors as knowledge, experience or culture, perceive certain objects and patterns differently. This not only makes it difficult to compare preferences among different end users, but also complicates an assessment of how preferences change within a single user.

The inclusion of user landscape preferences in forest planning has been addressed by Wherret (2001). As existing state-of-the-art prototypes show, it is already possible to communicate the general characteristics of a landscape to people from diverse backgrounds. Notwithstanding, Wilson and McGaughey (2000) showed that the credibility of the viewers may be lost if some details they are familiar with are lacking. These authors as well as Sheppard et al. (2001) noted some of the future challenges when using visualization tools for the integration of public participation in the decision making process. Sheppard and Meitner (2005) tested some static visualization procedures with stakeholders using multi-criteria analysis, in order to get scenario preferences in a public participation study. These authors showed that such techniques may have a high impact on how the general public perceives landscape changes and allowing for effective public participation in the decision making process. Interesting insight related to what the public evaluates positively in landscapes has been addressed by Kaplan (1988) and summarized by Bell (2001). Accordingly, aesthetically pleasant landscapes generally contain (a) coherence, that is, landscapes should not be chaotic and present patterns easily discernible; (b) complexity, meaning that monotonous environments are not sensory stimulating; and finally (c) mystery, referring to the fact that a view should not be comprehended all at once, preserving elements able to be perceived only after some time.

The question of whether aesthetically pleasing scenes are sustainable is a current research topic (Sheppard et al. 2001). Several authors (e.g., Gobster 1995) defend that there might exist a discrepancy between sustainable forestry practices and the public perception of the sustainability of a given forested landscape. In fact, as Kohm and Franklin (1997) have noted, many times the reverse is true. The public tends to actually devaluate management practices conducted to increased sustainability or biodiversity. According to Sheppard (2001), visual quality objectives, which establish levels of allowable landscape change, were responsible for major reductions in available timber supply. As a result, these types of objectives have become secondary. The same author shows that the traditional conflict between timber production and landscape aesthetics is being replaced by the resource conflicts of timber production versus ecology and landscape aesthetics, and between ecological and aesthetic objectives. The latter type of conflict is particularly important when the search for aesthetically pleasing landscapes is part of a planning objective.

3 Landscape Visualization Systems

The basic information for any kind of landscape visualization process is geographically based. Geographical information systems (GIS) have therefore become some of the most common initial platforms, capable of producing images with three-dimensional (3D) content. GIS allows the representation in map format of several important components of the wild, all at once. This 3D–2D transformation largely increases our capability to perceive an infinite amount of points of view all at once and within a single abstract framework. Yet it is this same 3D–2D transformation that allows for such powerful analysis, which is one of the largest obstacles for perceiving the landscape realistically. GIS vendors have for a long time provided support for including 3D visualization by using raster based Digital Elevation Maps (DEM), upon which any geographical layer could be draped, allowing for the perception of land use across large areas. Several vendors (e.g., ESRI's ArcGIS) even provide tools for dynamic visualization, which present interesting results when the impact of the vegetation is not important.

Draping over the DEM a remotely sensed image can sometimes enhance visualization of land cover through a GIS. This can offer to the manager some initial perception on how different forest cover types may be present in specific locations on the map. Yet the usage of remotely sensed images is not without drawbacks. In the first place, to be immediately perceived at medium range, the spatial resolution must be accurate enough so that landscape features may be immediately perceived (typically resolutions of 1 pixel per m^2, common in promptly available satellite imagery). Also, despite the fact that the above resolution is usually sufficient to provide users a sense of some landscape characteristics, for very large areas or when allowing high depth vision fields, images become so large as to be unmanageable in most platforms, without compression. Secondly, at close range, these seemingly detailed images will fail to convey much useful information. Finally, accurate and recent remotely sensed imagery may not be immediately available, due to the high cost of obtaining and processing it. A popular and readily identifiable landscape visualization tool available is Google Earth (http://earth.google.com) which uses these principles to convey a sense of the landscape for the whole world, matching GIS data, DEM as well as remotely sensed data (Fig. 1.)

The visual impact of actually representing the vegetation over the terrain has been acknowledged in most studies and has been pursued independently by various authors. The natural elements of the landscape make the problem overtly complex and, so far, no unified strategy has been able to solve all the technical issues faced by such systems. Therefore, various research teams have independently developed a variety of products, each able to use the available technology to produce increasingly better images. Real-time navigation requires dynamic rather than static simulations (Zube et al. 1987). Dynamic visual simulations present the landscape as seen by a moving observer, while static simulations depict the landscape as viewed by a stationary observer sometimes with multiple snapshots. The former does not constrain the observer to predetermined viewpoints, yet requires more advanced computing resources. Initial approaches for virtual landscape visualization have focused on

Fig. 1 A 3D view of the Serra de Monchique (Portugal) captured from Google Earth representing a landscape as observed at 1.12 km above see level. The effects of forest vegetation over the landscape can be perceived, although the deficiencies of the scale are apparent for the nearby regions (Vertical scale is 2.0 $\times$ the original value)

static imaging as computational resources were not sufficient for dynamic views. However, this is being progressively substituted by dynamic simulations, as most present day research efforts show.

The image quality of static visualization systems has steadily increased as well as the necessary computational requirements. SmartForest II (e.g. Orland 1997; Uusitallo and Orland 1997) is an interactive forest visualization tool that allows the user to simulate stand management options and to assess visual impacts of insect plagues. A more recent approach, Soler et al. (2003) used radiosity and illumination models to produce realistic images reported that the rendering of a scene representing a plantation of 233 trees required 4 hours to process a single image. For static visualization, there are currently very sophisticated products capable of depicting almost real world images. General purpose modelers and renderers like Maya (http://www.autodesk.com/maya), 3D Studio MAX (http://www.autodesk.com/3dsmax), or LightWave (http://www.newtek.com/ lightwave) are known to produce almost real landscape images although they are also only adequate for producing static views, or fixed animations. These packages nonetheless have huge requirements in data, man power and computational resources that generally are difficult to get for the majority of studies.

Not fully dynamic, yet allowing for a high degree of interactivity is the approach by McGaughey (1997). This author developed a working prototype, the *Stand Visualization System* (SVS) which is anchored on a forest inventory system and on geographical data. This tool was reported to produce visually clear results and is being used mainly to plan silvicultural operations and to assess the visual quality of small terrain plots. *EnVision* (Wilson and McGaughey 2000) is a visualization system based on SVS that can link to a forest management information system and, provided the data is available, is capable of providing static depictions of full forest stands or of producing multiframe animation sequences.

The dissemination of accelerated graphics cards provided developers a medium for exploring dynamic simulations with reduced hardware costs. Weber and Penn (1995) developed a modeling system able to adequately represent trees and were able to produce life-like landscape representations, even though the number of trees in the landscape is not very large. Seifert (1998) developed two advanced software packages: TreeView and L-Vis. TreeView allows three-dimensional interactive visualization of forest stands. L-Vis enables prospective forest representations by integrating individual tree growth models. These systems have been tested with relatively small geographical datasets. Meyer (2001) developed a visualization prototype that uses volumetric texture techniques for regular forested areas. This framework includes different levels of detail and shadow information, allowing dynamic lightning conditions. This prototype was also tested with a relatively small geographical dataset composed of about 1000 trees and was reported to produce a high quality output using state-of-the-art hardware available at the time. Zach et al. (2002) presented a combination of continuous and discreet level-of-detail algorithms for vegetation rendering over irregular terrain, being able to execute real-time flights over a very large area $(120 \times 120\,\mathrm{km})$, albeit with somewhat small forest density. Deussen et al. (2002) have developed a visualization module able to depict in real time small areas, yet with very high geometrical complexities, including landscapes with flowers and grassland. Falcão (2004) developed a prototype with linkages to other modules of a decision support system (DSS) and reported visualization and real-time navigation capabilities using a 500 hectares test forest area. Prototype interfaces enabled users to experience virtually the impacts of different management scenarios. Despite the connection to real world data, the visual results were not appealing and a view frustum of only 500 meters was used. Emphasis on the visual quality of the produced images was pursued by Decaudin and Neyret (2004). These authors used volumetric rendered trees combined with level-of-detail algorithms to develop an interactive visualization tool producing pleasant results, albeit for a limited size landscape, composed of about 2300 trees. Dietrich et al. (2005) have produced a prototype allowing for real time exploration of a small area, of about 9 ha, with very detailed vegetation representation, comprising shrubs, trees and grass, producing convincing outputs. *Lenné3D* (Werner et al. 2005) is a powerful system that allows both static and dynamic visual simulations. This system is able to represent medium sized landscapes (up to 100,000 trees) with high quality, and very detailed vegetation data. Falcão et al. (2006) developed a dynamic real-time landscape three-dimensional visualization tool for large areas which is capable of producing dynamic simulations of prospective forest management scenarios (Fig. 2). This system is linked to a forest resources decision support system encompassing a GIS, a forest stand simulator and decision models. In this study, the authors have demonstrated the prototype applied to a large dense forested area

$(32 \times 32\,\mathrm{km})$, with a viewing distance of up to 2.0 km. Pykäläinen et al. (2006) use the *Virtual Reality Modeling Language* (VRML) allowing users to visualize and navigate through virtual landscapes over the internet. This system is also anchored within a decision support system. One of the most impressive efforts in landscape visualization hardware is the *Virtual Landscape Theater* (MLURI 2007). A special room with dedicated hardware enables full landscape perception simultaneously for a number of users. This system allows the appreciation of current, past and prospective landscapes.

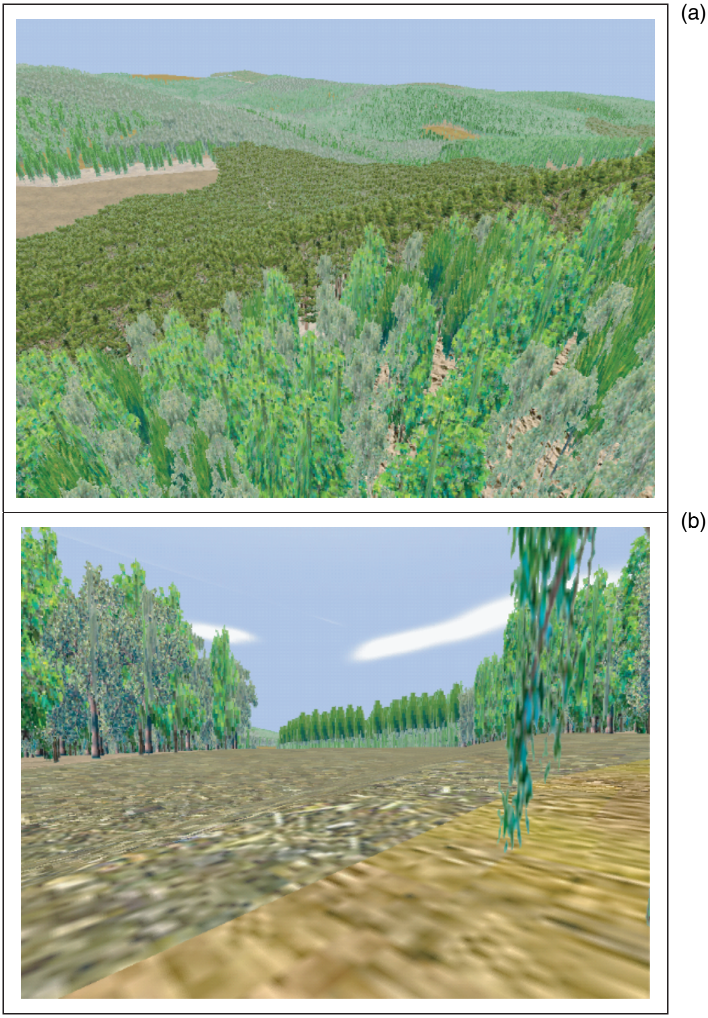

Fig. 2 3D depictions of prospective landscapes of the area depicted on figure 1, produced with the system of Falcão et al. (2006). (a) above coppice view,showing the land cover mosaic of the surrounding landscape with a 2.0 km view distance; (b) near ground view of a clearcut with the edging stands, as well as the characteristic soil texture

4 Factors Influencing Design

Computational 3D representation has a number of important aspects that need to be confronted before system implementation. Computers are exceptional at processing large quantities of information, yet, rendering outdoor scenes is still a major challenge even with state-of-the-art hardware. Three-dimensional objects are represented inside computer memory as a set of polygons, and adequate textures that cover the generated surfaces, are producing life like images. Polygons in 3D space can be efficiently displayed, as each polygon can be decomposed in triangles. These are the easiest shape types able to be represented with graphics hardware. As such, when measuring the geometric complexity of a scene, the metric used is often the number of triangles or, equivalently, the number of vertices.

The natural landscape is particularly difficult to depict accurately due to two important factors: (a) The geometric complexity of the vegetation, namely trees, is generally quite large and (b) The observation of the landscape horizon forces the representation of very large areas to accurately perceive the environment. As an example, humans can recognize and appreciate landscape features at least 5 km away, depending on the viewing conditions. Assuming a typical viewing angle of 90°, this amounts to a simultaneous view, in optimal conditions, of up to 2500 ha, an area with a dimension that may be unmanageable even with present day computing resources, especially when considering vegetation cover. The problem in representing landscapes, differently from what happens in other visualization types, is that potentially every feature is visible from everywhere. Systems that depict detailed urban or indoor scenes benefit from the fact that many objects occlude others, and it is thus possible to determine at run-time which ones are visible at every single instance, which drastically reduces the depicted geometric complexity of the observed scene. On the other hand, when visualizing natural landscapes, the eye should perceive the richness of wide horizons and get a holistic appreciation for the whole.

Another critical aspect in the use of visualization for forest management purposes has to do with the fact that each object represented in a computer must exist as a virtual entity modeled a priori. Therefore, each represented tree must exist in a modeled form beforehand. Trees, in particular, are complex objects, whose size, if modeled with a large amount of detail, require hundreds of thousand of vertices and appropriate textures.

It is important to recognize how the problem escalates computationally. Considering a moderately detailed tree model with 50,000 vertices, each polygon would requires 3D spatial coordinates (x, y, z), 4 color components (red, green, blue, transparency factor) and 2D texture coordinates. These account for 24 bytes per vertex, which accounts in total for 1.14 Mbytes, just for one tree, excluding texture information. A typical high end machine with 4GB of RAM, with the above requirements, thus is able to store in volatile memory about 3600 trees. Considering a moderately dense forest with 500 trees per hectare, the above number corresponds to 7.2 hectares, which is equivalent of a square of just about 270 meters wide. Such an area is clearly insufficient for any landscape perception. It can be argued that farther away trees could be represented in smaller detail when rendered, while the closest ones could be represented in full. Yet, if a dynamic visualization is required, a far away tree may, in a fraction of a second, become a closer one, and therefore the information must be either available or produced in real time. This latter method is possible, and some studies have been able to produce good results with level-of-detail algorithms (e.g. Zach et al. 2002; Werner et al. 2005).

According to these issues, three basic factors are necessary to consider when envisaging the development or selection of a forested landscape visualization tool, and each entails the answer to a specific question:

(A) Landscape size – Is the tool able to confront large landscapes with large viewing frustums or will it instead focus on small scale visualization problems to confront typical forestry or micromanagement objectives?
(B) Visualization type – Does the visualization tool allow real time navigation (dynamic) or is it only able to show a limited number of fixed snapshots (static)?
(C) Time frame – Is the area to be visualized in its present state or does the user need to look into the future?

The following paragraphs will discuss each of these topics in detail.

4.1 Landscape Size

The visualization of very large forest areas is still a challenge even for the most advanced current dynamic simulation tools. The amount of geometric information necessary to represent terrain and vegetation is usually too large for tools that are mostly suitable for small to medium size areas (below 2000 ha, or about 100,000 trees). The improved capability of producing near-photorealistic results has not been matched by enhanced user interactivity. Selecting the total area to visualize will impact on a number of decisions that have to be taken beforehand. A small area will usually not call for full GIS support, and only in some cases information about altimetry will be required. Visualizing small areas is adequate when the most rigorous information about trees is present and data exists regarding each tree position and biometric characteristics. As the total depicted information is usually small (depending naturally on the amount of detail present for each tree), it is usually possible to produce very realistic images and assure dynamic visualization aspects.

On the other hand, confronting large forested landscapes entails a set of problems that may be difficult to solve appropriately. The first one is data completeness and accuracy. For areas larger than 50 hectares it is, for most cases, difficult if not impossible, to get accurate data for all the information required. Fairly accurate land cover data and digital elevation models which are fundamental for realistic depiction of the terrain are generally available. However, similar information is often absent for the vegetation. As noticed before, for each tree present it is necessary to know at least its position and the visual and biometric characteristics. Assuming an average stand density of 500 trees per hectare, this would require inventorying and positioning about 25,000 trees just for 50 hectares. For a real perception of a typical landscape, usually at least 50,000 hectares are required, which allows for a viewing distance of several kilometers (usually from 2–5 km). At this scale it is not feasible to obtain real-world individual coordinates for trees, and simplifications have to be assumed. The procedure may sometimes involve the automatic generation of trees according to stand level biometric variables (e.g. Falcão 2004).

4.2 Visualization Type

Real-time visualization of large outdoor scenarios has been a computational challenge since the inceptions of Computer Graphics. The heterogeneity of landscape elements and the difficulty of finding efficient algorithms for the reduction of the displayed geometrical complexity were paramount in explaining the large research investment in this area.

As referred, a 3D landscape visualization system can be either static or dynamic. Static visualization aims at producing still images of the landscape at usually predefined points of view. Dynamic visualization allow for users to virtually navigate over the landscape. A dynamic scene is generally demanding for the computational hardware, as each image has to be generated in fractions of a second in order to ensure the illusion of movement. The limiting factor in such systems is typically the geometric complexity of each scene, which is generally accounted for with the total number of vertices present. A higher number of vertices is usually present in large landscapes due to the inherent increase in the number of existing objects, or when individual tree detail is high. For such cases, as the number of far trees is generally much larger than the number of closer ones. Thus, the use of vegetation-based level-of-detail algorithms may be a workable option (e.g. Zach et al. 2002). These methods are able to render far trees with much less detail than the close ones, generally decreasing the effective geometric complexity of the scene. Notwithstanding this approach may sometimes not be the best option, as there is a significant increase in the complexity of the rendering application, and when the landscape and the viewing distance is so large, that it is not even feasible to increase the geometric detail of nearby trees.

4.3 Depiction Time Frame

A landscape can be shown as it is in the current state or how it is going to look like in the future according to a provided management scenario. Current state visualization tools allow for a landscape manager to perceive all the major aspects of the current space without physically being in the field. It is a convenient means to virtually evaluate landscape aesthetics from points of difficult access, even allowing aerial views. Visualizing current state forests has the added benefit of using real world vegetation data and land cover information, therefore escaping one important degree of uncertainty (Fig. 3).

Visualization of future landscapes is a generally more complex subject as it involves the simulation of several aspects: vegetation growth, ecological succession,

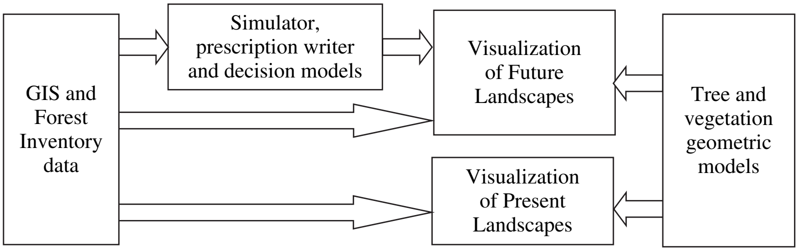

Fig. 3 Data flow for present and prospective landscapes visualization systems

human intervention and management practices. These can be integrated in the framework of a decision support system, as discussed later in the chapter.

When addressing vegetation growth and changes in the landscape, it is important to know how species present in a given landscape will grow over time. This is a complex subject, as for extensive areas, basic information at tree level is often not available and most simulation models do not include individual tree growth. A further important aspect is related to vegetation changes after silvicultural practices like clearcuts or reforestations with different species. Models should therefore exist that may be able to encompass all possible variations. For simple forest management planning, such models may, without significant inconvenience, be used, as only stand level data is relevant for production problems. However, it can be a simulation challenge to extrapolate such data to the individual tree level, required for visualization purposes.

5 Data Requirements for Landscape Visualization

A well-designed real time 3D visualization tool should be based on the construction of 3D scenes with a hierarchical structure containing the representation of arbitrary modeled objects. Those objects are then rendered according to its characteristics and spatial position relative to the viewpoint. It is expected that the main components of the rendered scenes will include:

(a) Geographical information data including land cover and the digital elevation map of the landscape study area;
(b) Vegetation data, comprising trees, bush compounds, under-story and grassland.

5.1 Geographic Information System Data

GIS data is fundamental for landscape visualization as to provide location specificity to every feature present. Both raster and vector data is necessary for adequate modeling. Raster information may include elevation and image based data. On the other hand, information related to spatial division of homogeneous areas (like forest stands), management as well as administrative and land division marks are generally stored in vector format. All this information is seamlessly integrated in a GIS, which can later be exported for a visualization package.

5.1.1 Soil Representation

For accurate experience of a virtual landscape, soil representation is an important factor to increase the illusion to the user. Similar to tree texturing, soil texturing is an important component in a landscape visualization system. It allows the visual simulation of typical soil texture that generally changes according to land cover characteristics, provided by the GIS. Alternate approaches mapping ortho-rectified satellite or aerial photo imagery over the study area DEM may be adequate if available. The advantage of such an approach is that it may depict immediately identifiable landscape features like roads, tracks, massive rock formations and other physiographic characteristics. The presence of such location points is often of paramount importance for users to situate themselves in the virtual landscape, and might be difficult to display on the scene otherwise. On the other hand, this approach portrays the soil visible from above, and stand coppices will appear projected on the forest soil. This can be confusing for the user, especially if visualizing prospective landscapes, where a former densely forested area might appear bare after a clearcut. Further, even with small scale remotely sensed data, using a pixel size equivalent to $1\,\mathrm{m}^2$, may destroy the visualization illusion, as the soil will often appear blocky or blurred when observing forest characteristics at close range.

5.2 Vegetation Information

The landscape is composed of its physical characteristics and man made constructions, yet its aesthetic characteristics are often directly influenced by the existing vegetation. For large landscapes, retrieving and storing vegetation data is a difficult problem for which no simple answer exists. Assuming an average occupation of 300 trees per hectare for a medium sized area of 50,000 hectares, it may be necessary to know the information for 1.5×10^6 trees. Just storing the tree coordinates requires about 170 Mbytes of data. The problem is obviously more severe when the full vegetation geometry is actually represented in the scene. A single tree can reach geometric complexities of over 200,000 vertices (Fig. 4a), when all branches and leaves are individually shown.

To address the potentially large amount of vegetation geometrical data present in the landscape, and when there is a necessity for fast rendering as in dynamic visualization systems, a typical approach is the use of a simplified representation. Two crossed rectangles are often used per tree, to which an image of a tree is applied (Fig. 4b). To produce transparency, around the tree, a fourth channel of color is present that graduates the opacity of the different parts of the image. Results are in general satisfactory, yet this procedure is only adequate when the object to display is radially symmetric, as is usually the case with trees. A major drawback of this

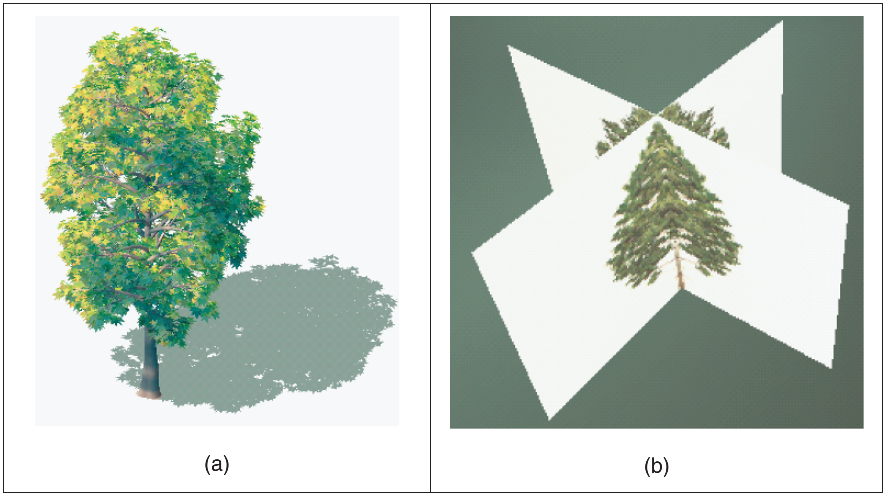

Fig. 4 Two tree representations: (a) a fully lit and shadowed 3D computer generated tree with 267,000 triangles (Generated by OnyxTree Broadleaf 6.0); (b) a simplified model of a tree composed of two crossed rectangles (4 triangles)

approach is when looking to the ground from above, the simplification becomes apparent, and it is possible to identify the original geometric components and ruin the illusion. The most basic attributes are required to depict such trees. In the first place, it is required to know the image corresponding to its species and age class. Then a form factor can be applied to the tree according to its diameter at breast height, or crown diameter, and appropriate tree height. All these variables can be extrapolated through simple statistical interpolations of stand level variables, creating realistically looking scenes (Falcão 2004).

Dynamic visualization, requiring accurate outdoor modeling, which includes not only trees, but also grass and shrubs, is outside the capabilities of present day computing when the vegetation density is large and the viewing distance extends for several kilometers. Several computational techniques exist to improve the display of extensive geographic cover, as the intrinsic nature of topographic geometry allows the use of level-of-detail algorithms, in which the represented geometrical complexity of a model depends on the proximity to the observation point.

5.2.1 Geometry Texturing

Using real photographic images for displaying trees is at the present moment not a feasible option, for several important reasons: In the first place it is quite difficult to get accurate individual images due to the following factors: (a) It is difficult to find isolated good samples. When isolated trees are found they often do not show the typical shapes they display within a forest stand; (b) with common photographic equipment parallax errors distort the image of the tree. For smaller focal distances, trunks will generally appear thick and the coppice thin. Longer focal distances, on the other hand, may complicate the acquisition process as they require the photographer to be farther from the tree, thus reducing the possibilities of getting a clear line of sight and adequate lighting conditions. Falcão (2004) has verified that artificial images may provide adequate results, as generally the user does not focus on a single tree in particular, but the general aspect of the forested landscape. Further, artificially crafted images are sufficiently simple and familiar so as not to distract the user with any systematic flaw, yet at the same time able to communicate the most important characteristic of the species.

As mentioned before, ideally a forest landscape visualizer should have all the detailed information related to all terrain and tree characteristics. It should further have the computational power to display all that information rapidly, allowing for full interactivity. Yet, the nature of forest information, due to its larger quantity and diversity, makes that objective an elusive goal. For instance, it would be necessary to register tree characteristics and positions for all individual trees in the intended landscape, which is never a viable option. In the majority of situations, just the breast height diameters for all the trees within a measurement plot and occasionally some tree heights are available.

5.3 Image Enhancing Techniques

The essential characteristics for landscape representation have been described. Yet current technologies allow the enhancements of the produced images with additional features that facilitate a further degree of immersion to the potential user of such systems.

(a) Vegetation movement and dynamics. The use of vertex shaders present in most graphics hardware allows the representation of vegetation movement that simulates the presence of wind.
(b) Grassland. Grass effects may be displayed as particles over the soil, without largely increasing the scene's geometric complexity.
(c) Presence of different elements with accurate and dynamic renditions. For example, water in ponds, swamps, lakes or rivers, can convey a sense of freshness and livelihood to the environment, and it is possible to accurately represent it, again taking advantage of programmable graphics hardware.
(d) Human constructions and other landscape features. The presence of landscape features different from soil cover data or vegetation models are very important for the end users and are fundamental for providing reference points when situating a given image of a virtual landscape. Yet, these are seldom available, usually require separate modeling efforts and apply solely to specific conditions.
(e) Lightning and atmospheric effects. Current hardware allows the application of complex lightning models that allow a more vivid representation not only of the vegetation but of the whole scene. Different materials with different optical properties can be assigned to parts of the scene, producing convincing illusions.

All these aspects increase the sense of reality for the user when perceiving an artificial landscape. However, each incurs an extra cost resulting from data production and modeling, and the development efforts for implementing such features. These will also generally increase the geometric complexity of each rendered scene that will leverage the hardware, in which the system can run.

6 Decision Support Integration

The integration of temporal and spatial interactions of management options at the landscape level is a fundamental aspect to address sustainable forest management planning concerns. However the impacts of a management plan on the visual characteristics of a landscape are often hard to anticipate. Combining decision support tools with interactive visualization tools may be a workable option, allowing stakeholders such as land owners, local citizens, and visitors to evaluate prospective management strategies, therefore effectively participating in the decision making process. Brabyn (1996) and Wilson and McGaughey (2000) discussed advantages and disadvantages of using visualization aids in forest management planning and underlined the need to avoid misinterpretation of photorealistic depictions. Nevertheless, it is possible to present the visual impacts of management strategies in a format that may be readily identified by forest managers and the public (e.g. Wilson and McGaughey 2000; Luymes 2001). The visualization of prospective forested landscapes may provide valuable information to assess impacts of forest management planning. Visualization can also contribute to addressing increasing concerns with the visible effects of environmental changes.

Among some of the working prototypes that allow direct connection to decision support tools, it is worth citing the following projects. The Stand Visualization System (McGaughey 1997) was one of the first of its kind to develop a working prototype, solidly anchored on a forest inventory system and geographical data, and focusing mainly on management of silvicultural operations for inventory plots. Uusitalo and Orland (1997) developed *SmartForest*, a tool that allows interactive selection of trees, albeit also in a limited area. Pykäläinen et al. (2006) demonstrated the integration of a web-based visualization system within a larger scale decision support system. Falcão (2004) and Falcão et al. (2006) defined the steps for the integration of a visualizer within a forest management decision support framework designed for Mediterranean ecosystems (Falcão and Borges 2005).

Basically, the requirements for the integration of a landscape visualizer within a decision support system are similar to the ones defined previously for the depiction of future landscapes. Nevertheless, information management within the system requires some careful planning and clear definition of the assumptions followed, as data is also used for different purposes within the system. Central to the process is the existence of accurate base information, supported by a GIS. Data must be simulated and several management alternatives considered (Fig. 5).

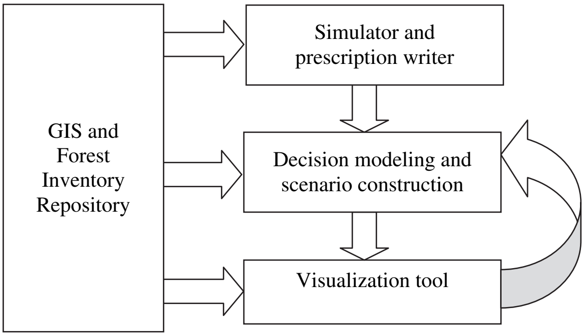

Fig. 5 A simplified view of a generic decision support system incorporating a visualization module

The simulation may involve different policies, or different land use strategies, which may be different for each homogeneous land unit (HLU). The second phase involves the selection of the most appropriate alternative for every HLU according to a set of constraints. These may involve production and forest wide objectives, as well as localized targets with spatial or ecological targets. Scenarios are then constructed and finally observed in a virtual environment, to fully comprehend the aesthetic impact of the proposed management plan. It is then generally necessary to observe each generated virtual landscape for every period within the simulation time-frame. Finding aesthetical problems within a prospective scenario may involve a re-construction of the proposed model and reiterating the process until a compromise is found between the primary management objectives and aesthetical effects.

7 Conclusions

Effective use of forest resources should take into account not only the value of the private benefits to the owners but also all the ecological, environmental and economic benefits to the society in general. Forest management practices are usually based primarily on technical criteria. Non-market values, landscape aesthetics, and externalities are barely considered. As a consequence, the linkage between management spatial scales (e.g., the stand vs. the landscape) scarcely exists and information available to define multifunctional forest ecosystem plans is lacking. Recent research has focused on the linkage between stand and landscape management scales and on the integration of aspects pertinent to participatory planning and further aiming at narrowing this gap by providing integrated functional tools that may allow users to visit and interact with several prospective forested landscapes. Graphic environments are then produced and offered to end users to better support decision making through the evaluation of landscape scenic impacts associated with different management plans.

Nonetheless, the visualization of large forested areas is still a challenge due to the intrinsic geometric complexity of the natural landscape, for which there are still few adequate alternatives, even for some of the most advanced current dynamic visualization tools. The amount of geometric information necessary to represent terrain and vegetation is usually too large for tools that are mostly suitable for small to medium size areas. The improved capability of producing near-photorealistic results, as is the case for high-end rendering systems, has not been matched by enhanced user interactivity when large areas are addressed. However, with current graphical hardware capabilities almost doubling each year, it is expected that hardware produced in the near future will be able to tackle areas with extremely high geometric complexity and render such scenes in real time. By then, the real bottleneck present in forest visualization systems will become even more apparent, which is data availability. Most studies in this field have addressed the computational and graphical aspect of the problem, assuming that adequate data would exist. However, this is not so, especially for the visualization of simulated scenarios when so many assumptions have to be made that any produced image should be checked for its realistic content, no matter how perfectly rendered it is.

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Visualization in Support of Landscape Design

Rüdiger Mach

1 Introduction

Visualization involves the rendering of realistic or abstract images that represent an underlying data set. The data set may be the product of physical measurements of the real world, or it can be the product of a computational model. Visual imagery is relatively easily interpreted by humans and helps people to understand complex data and processes. We may distinguish between data visualization which presents large amounts of data, such as in meteorology, in a comprehensible way, or performance visualization which shows the results of a spatio-temporal analysis so that decision makers can see the effects of human activities in the landscape.[1]

Visualization technology includes methods for creating images, diagrams, or animations to communicate a specific message (Shneidermann et al., 1999). Forest managers, landscape designers and engineers are not only responsible for the technical realization of projects. They may also have to take on the role of an author, presenting a story, which often includes pictures as well. Any in-depth analysis and consultation should go hand in hand with the professional realization of a visualization project. Thus, a well-executed visualization project will fascinate both the visualizer and the viewer and can often be regarded as a work of art in its own right.

1.1 Target Groups

Before starting work on the realization of a project, it is essential to reflect on the needs of different target groups and to decide how to present the key aspects of a project in the most practical and comprehensive manner, using currently available technology. The Stand Visualization System (SVS), for example, generates forest images for forestry specialists. It depicts particular stand conditions which are represented by a list of individual stand components.

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[1] (http://www.alphaworks.ibm.com/visualization/newto?open).

K. von Gadow, T. Pukkala (eds.), *Designing Green Landscapes*,

Fig. 1 Tools like 3ds max enable the user to create highly sophisticated visualizations

More sophisticated tools like *3D Studio Max (3ds max)*[2] enable the user to create rather realistic images (Ervin and Hasbrouck, 2001). *3ds max* is a 3-dimensional graphics- and animation software, which is being used in the areas of computer games, animation, motion pictures and design. Figure 1 is an example of a landscape created by *3ds max*. Such images are better suited for non-specialists.

The two examples demonstrate that one can distinguish between two kinds of 3-dimensional visualization. The first kind is known as a "technical visualization" and is used as a means of communication in planning and designs among professionals, e.g. foresters who are able to distinguish different tree species by their crown shape. The second type is a "realistic" visualization. These are representations used as a tool by the media for presenting a concept to the general public.

1.2 Some Basic Concepts

It is useful to have a look at some basic concepts before going into the finer details of visualization techniques. Sheppard (1989), for example, defines five principles for landscape visualization:

1. *Representative character*: visualizations should represent typical or important aspects or conditions of the landscape.
2. *Exactness*: visualizations should simulate the factual or expected appearance of the landscape (at least involving the aspects of interest).
3. *Optical clarity*: details, components and content of the visualization should be clearly identifiable.
4. *Interest*: the visualization should arouse the interest of the viewers and hold their attention for as long as possible.

[2] http://www.autodesk.com/3dsmax/

5. *Legitimacy*: the visualization should be justifiable and its level of exactness should be verifiable.

For decision makers, some of the advantages of using 3D visualization are (Mach, 2004):

- The model can be viewed from different angles.
- Being able to look at an object 3 dimensionally corresponds to our natural way of looking at things resulting in a much faster comprehension of complex surfaces and geometries, especially in the field of terrain data.
- There is the possibility of interaction with existing 3 dimensional data.
- It is always easier to represent problem areas (weak points) of complex constructions in a 3 dimensional way.
- Most landscape and vegetation data already exist 3 dimensionally anyway, and are edited 3 dimensionally in (pure) planning.
- Avoiding misunderstandings saves a lot of time.
- A 3D visualization enables an "engineer-like" analysis of data.
- A 3D visualization facilitates comprehension for non-experts.
- Complex concepts and ideas are easier to explain using 3D visualizations.

Visualization is a special means of communication (Preece, 2002). The creation of every still, movie or interactive composition used for communication purposes is subject to the basic principles of an accomplished composition. The more specific these are, the easier it is to convey the subject matter to be translated into a visualization.

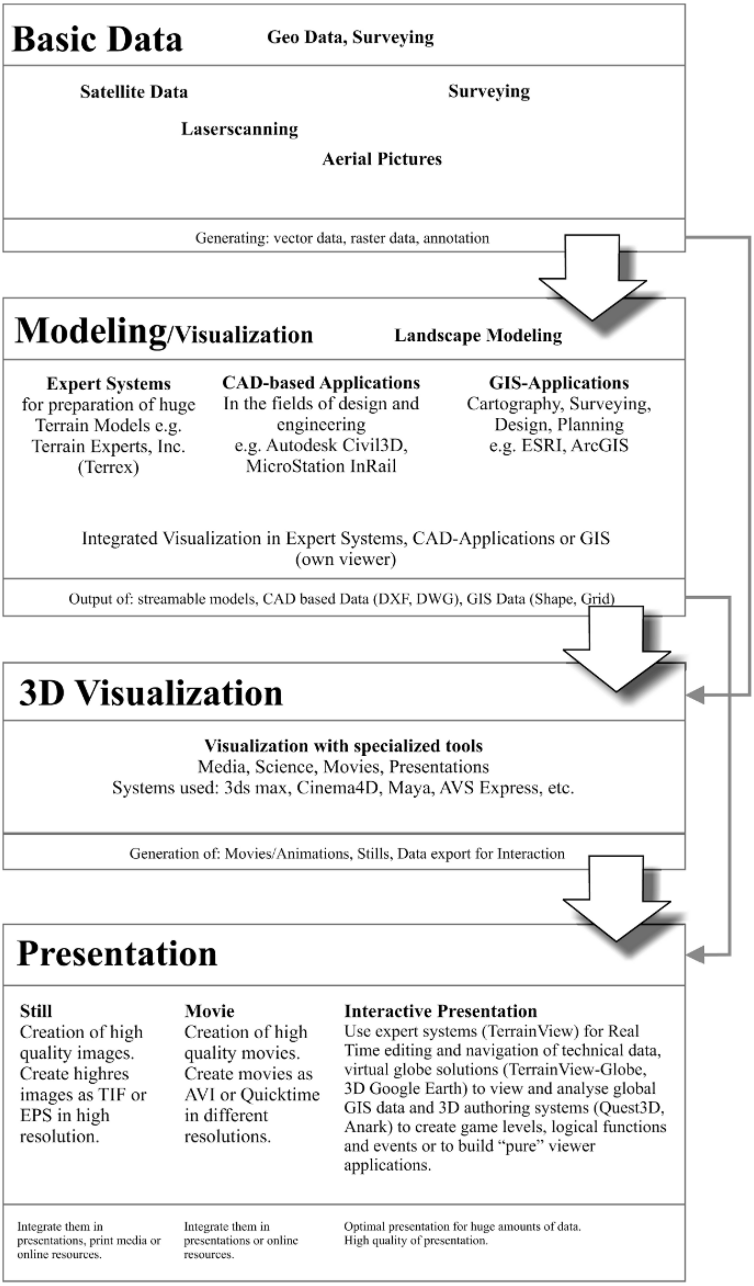
2 The workflow of Visualization

Mach and Petschek (2007) describe a typical workflow of a visualization project (Table 1). The process begins with the "raw" data which may be obtained from different sources. Especially important in landscape design are data obtained from remote sensing. The next step involves selection of an appropriate modeling approach. Today, it is possible to choose among a variety of modeling methods. Next, the visualization needs to be pre-processed and different tools are available for this task. Finally, several possible options for presenting a visualization are available. These steps are described in the sequel.

2.1 Basic Data

Visualizing a forested landscape is a particularly challenging task. The most laborious and demanding part of this task involves the acquisition of the basic data. "Basic data" in this context refers to measured data, and not to data connected to planning.

Table 1 Typical workflow of a visualization project, after Mach and Petschek (2007)



As exciting as it may be to create an arbitrary landscape according to one's own imagination – in whichever form this may be done – planners usually have to edit, convert and process reality-based data first.

In the majority of cases, reality-based data represent a great challenge. Very often, the data are not clearly defined, and they usually come in large packages – large packages meaning in this context ASCII files exceeding 100 Mbytes. Consequently, the necessary tools to view and process such large datasets are not always available. There are four different types of basic data, which can be used for creating a visualization (Coors and Zipf, 2005):

- Imagery data
- Elevation data
- Vector data
- 3D object data

The methods of collecting basic data are manifold. It is possible to make use of satellite or aerial photo data, or to use laser scanning methods to create a digital terrain model. Alternatively, data from field surveys may be used. More important, however, is the processing of these data.

2.1.1 Imagery Data

On the one hand aerial images and satellite photos constitute the basis for the creation of digital terrain models. On the other hand, such data are an important part of 3D visualizations, because they can be used as a basic material for establishment the relation to reality without sophisticated modeling.

2.1.2 Satellite Data

A satellite image is a representation of the surface of the earth which can be obtained either by passive systems (photographic cameras, digital or CCD cameras, infrared and multi-spectral sensors), which register sunlight or other forms of radiation reflected from the earth in the form of geometrical images; or active systems which radiate self-generated electromagnetic waves (radar and other radio waves, laser) and scan the resulting diffuse reflection.

The production and handling of satellite images involves standard methods of remote sensing and geo sciences (refer to other relevant chapters in this book). It also helps in the cost-effective production of maps and in the analysis of large areas in the field of environment protection. The classification and interpretation of satellite image information provide(s) important databases for geo information systems (GIS).

2.1.3 Aerial Photos

Images of existing landscapes may be taken during flights over a specific terrain. The purpose may be to selectively assess a single object, or to systematically cover a specific area. In the latter case, the area is covered in parallel strips, with an overlap of 30%–50%. These overlapping strips are then assembled and analyzed. When Orthophotos are not available, "normal" photos have to be synchronized, which means that the distortion of a photo, resulting from the curvature of the lens, is corrected by using appropriate software. In Germany, the "Bundesamt für Eich- und Vermessungswesen" creates digital terrain models using aerial photo information. When using stereo photographs, specific points in a defined 3D coordinate system are calculated and a grid DTM is interpolated from these.

2.1.4 Elevation Data

There are various types of data, which describe elevation (Westort, 2001). The most common kinds are:

- **Grid data** (or raster data) involving a regular grid with a specified "width" and an elevation allocated to each grid. The advantage of using grid data is that 3D representations can be generated rather easily, and that the representation does not require excessive amounts of computer memory.
- **TIN** (Triangulated Irregular Network). – A TIN surface is always made up of connected triangular surfaces. The elevation of every point on the surface can be interpolated from the x, y and z values of the surrounding triangle which makes it possible to determine the surface features.

Although the above-mentioned types are both Digital Elevation Models (DEM), the term DEM usually refers to grid data.

2.1.5 Vector Data

Vector data are geometrical objects such as points, lines (series of point coordinates), or polygons that are also called areas (i.e. shapes specified by outlines). Examples include boundaries of forest compartments or vegetation types, which are represented as polygons. A well-known vector format is ESRI's shape file.

2.1.6 3D Object Data

The inclusion of 3D objects will make any scene look more realistic. Therefore, the use of 3D models of trees or buildings can greatly enhance a visual presentation. Some well-known 3D formats are: Multigen OpenFlight (*.FLT), Autodesk 3DStudio (*.3DS), VRML 2 (*.WRL), Alias/Wavefront (*.OBJ) or NewTek LightWave 3D (*.LWO).

2.2 Modeling and Visualization Methods

The visualization designer has many methods at his disposal for building 3D models and visualizations. The most commonly used applications are

- **GIS** – (Geographic Information System). A GIS is a system for capturing, storing, analyzing and managing data (Maguire et al., 1991 – for more details, refer to relevant websites for descriptions of *GRASS*[3], *ESRI*[4], *ERDAS IMAGINE*[5] or just *GIS*).
- **CAD** – (Computer Aided Design). This is mainly used for detailed engineering of 3D models and/or 2D drawings of physical components in all technical fields (well-known CAD tools are *AutoCad*,[6] *VectorWorks*,[7] *Microstation*[8]).
- **3D Visualization Tools** – These are specialized tools for high quality visualizations (examples are *3ds max, Maya, Softimage, Cinema 4D*).
- **Landscape, Plant and Vegetation Tools** – These tools specialize in the visualization of landscapes, plants and vegetation (examples are *Terragen, Bryce, Vue d'esprit, Verdant, SpeedTree, XFrog*).

2.2.1 GIS

Having collected all necessary data and being able to read these does not mean that all obstacles in creating a landscape or terrain model have been overcome. Often, additional measures have to be taken on this data before work on the final visualization can begin.

An excellent way of gaining a quick overview of the data at hand is the use of a Geographic Information System (GIS). Products such as *ESRI's ArcGIS, Manifold* or *Civil3D* (*Map3D*) by AutoDesk are probably known to most readers. However, one has to keep in mind that visualizations based on GIS are usually intended to facilitate exchanges among experts (Buhmann et al., 2002). These programs were not developed, nor are they especially suitable, for high quality visualizations.

GIS tools are expert tools for communication among professionals. In the context of landscape visualization, they are mainly used as preprocessors for a presentation which is intended for the general public. To extract 3D data from a GIS application for visualization purposes, the Virtual Reality Modeling Language (known as the *VRML* format) is a good choice.

2.2.2 CAD

CAD programs like *AutoCAD* or *Microstation* are used for planning and design and of course, for preparing presentations. If additional plug-ins like *AutoCAD Map 3D, Civil 3D* or *Bentley PowerMap Field* are used, the borderline between *CAD* and *GIS* will become fuzzy. These programs also use a high quality 3D rendering option which enables the user to present his design data in a convincing way. Rendering

[3] http://grass.itc.it/

[4] http://www.esri.com/

[5] http://www.leica-geosystems.com/

[6] http://www.autodesk.com

[7] http://www.nemetschek.net/

[8] http://www.bentley.com/

still images or a simple camera flythrough is standard. But if one wants to prepare data for a high-quality presentation, a CAD program is certainly not the right tool to use.

2.2.3 3D Visualization Tools

A wide range of visualization tools can be used for high-end presentations. Some of the most well known programs are:

- A*utodesk 3ds max*[9] – (*3ds max*) is one of the most widely used off-the-shelf 3D animation programs.
- *Autodesk Maya*[10] – (*Maya*) is a high-end 3D computer graphics and 3D modeling software package, originally from/by Alias Systems Corporation but now distributed/marketed/owned by Autodesk under its Media and Entertainment division.
- *Softimage XSI*[11] – (*XSI*) by Avid Technology, Inc. is the successor of Softimage | 3D which was a fast, simple-to-use animation package, the first commercial package to feature Inverse kinematics for character animation.
- *Cinema 4D*[12] – (*Cinema 4D*) is a visualization application produced by MAXON Computer GmbH.
- *AVS Express*[13] – (*AVS*) is an application specialized in the field of scientific visualization.

These tools are suitable for visualization of complex 3D models; they are similar in many ways. For importing data generated with a GIS or CAD, one can use various data formats for the exchange of information in 2D or 3D. Most 3D visualization tools are capable of importing the following data directly:

- *CAD* interface – mostly *DWG* and *DXF*.
- Points – mostly *ENZ*[14] or raster data[15] as *ASCII* file.
- *VRML* interface – data format for the description of interactive 3D worlds.
- Various 3D formats – interfaces between 3D tools, the most common being *OBJ*[16], 3DS[17], and *LWO*[18] (see example in Fig. 2).

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[9] http://www.autodesk.com/3dsmax

[10] http://www.autodesk.com/maya

[11] http://www.softimage.com/

[12] http://www.maxon.net/

[13] http://www.avs.com

[14] Easting Northing Z (Elevation) is a column ASCII format.

[15] DEM as header defined ASCII file.

[16] OBJ – Alias Wavefront data format, 3DS – 3D studio DOS format – however, due to its constraints/limitations, the 3ds format is only suitable for small terrain models. LWO – Lightwave Object Format.

[17] 3DS – 3D Studio DOS format.

[18] LWO – Lightwave Object file format.

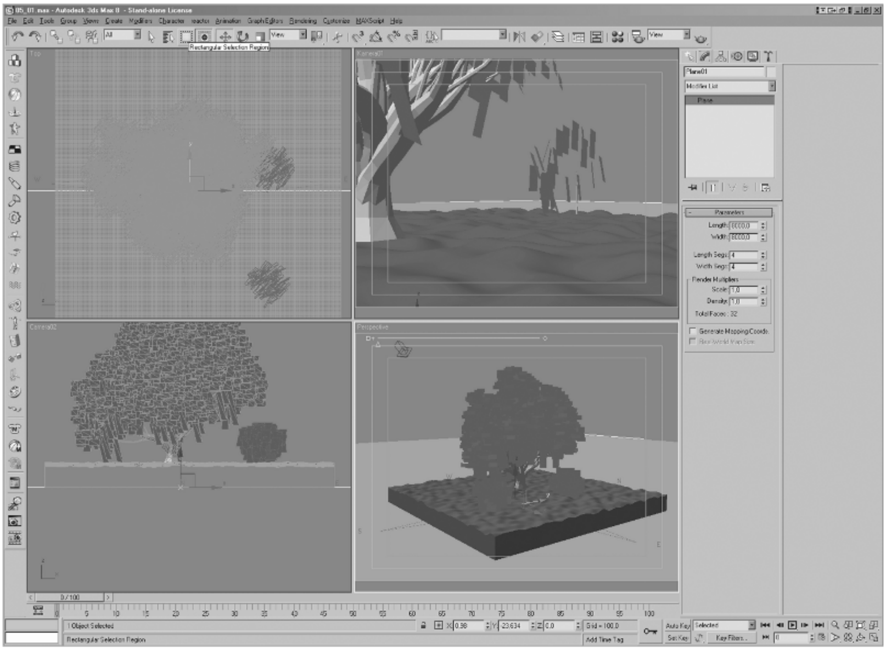

Fig. 2 Scene setup in 3ds max 8

- Import as a *RAW*[19]. file or as a grayscale picture.

But if the data is intended for use in a realistic visualization of natural phenomena or vegetation, one either has to spend some extra time learning additional plug-ins or rather rely on programs which are specialized in plant visualization.

2.2.4 Special Tools for Visualizing Landscapes and Vegetation

When one has to handle large vegetation data sets, it is worthwhile evaluate some specialized tools. The following are some examples:

- *Verdant*[20] by Digital Elements – Verdant is a stand-alone *python*-based plant generator, which allows building procedural 3D plants.
- *SpeedTree* by Interactive Data Visualization, Inc (*IDV*)[21] – *SpeedTree* is a set of tools which are often used in Real Time environments. It runs as a stand-alone

[19] *RAW* – is a raw data format which, depending on the make/manufacturer, compresses without loss and is used in digital cameras. It supports *CMYK*-, *RGB*- and grayscale pictures with alpha channels as well as multi-channel- and lab-pictures without alpha-channels.

[20] http://www.digi-element.com/verdant/

[21] http://www.idvinc.com/

application and can be also used as a plug-in for programs like *3ds max, Cinema 4D* and others.

- *Xfrog* by *Greenworks*[22] – *Xfrog* is an organic modeling and animation tool. It runs as a stand-alone application and can also be used as a plug-in for 3ds max, Cinema4D and others.
- *Bionatics*[23] – *Bionatics* has a mix of modeling tools for the needs of high quality 3D visualization, as well as for architects and the Real Time world.
- *Forest Pack* by *Itoo Software*[24] – These are vegetation tools which have been specialized for the integration into *3ds max*.
- *World Construction Set* (*WCS*) or *Visual Nature Studio* by *3D Nature*[25] – The WCS is a stand-alone CAD-like application. It is the only visualization tool which is able to import GIS data directly.
- *Planta3D* by *Lennee 3D*[26] – enables 3D plant modeling, optimized for a Real Time approach.
- *Bryce*[27] – An easy to use and fascinating toolkit to create landscapes and a lot of vegetation.
- *Vue d' Esprit*[28] – Who doesn't know the movie Pirates of the Caribbean? A lot of matte paintings were done with this tool.

Figure 3 presents an example of a scene setup in *Vue 6 Infinite* and the rendered result.

The scene setup in Fig. 3 shows a sample scene of Vue 6 Infinite. The picture in the background shows a screen shot of the program, the image in front the rendered result.

3 Types of 3D Representation of Vegetation

There are different approaches used in visualizing plants and vegetation. These can be classified according to the different design methods. The most popular types of plant representations can be described as follows (Bishop and Lange, 2005):

- **Symbols** – Instead of trying to visualize real plants, simplified symbols are used. The quickest and easiest way of generating a symbol is by choosing basic, primitive forms, like for instance a cone. The advantage is obvious: this saves memory

[22] http://www.xfrog.com/
[23] http://www.bionatics.com
[24] http://www.itoosoft.com/
[25] http://www.3dnature.com/
[26] http://www.lenne3d.de/
[27] http://www.daz3d.com/i.x/software/bryce/-/?
[28] http://www.e-onsoftware.com/

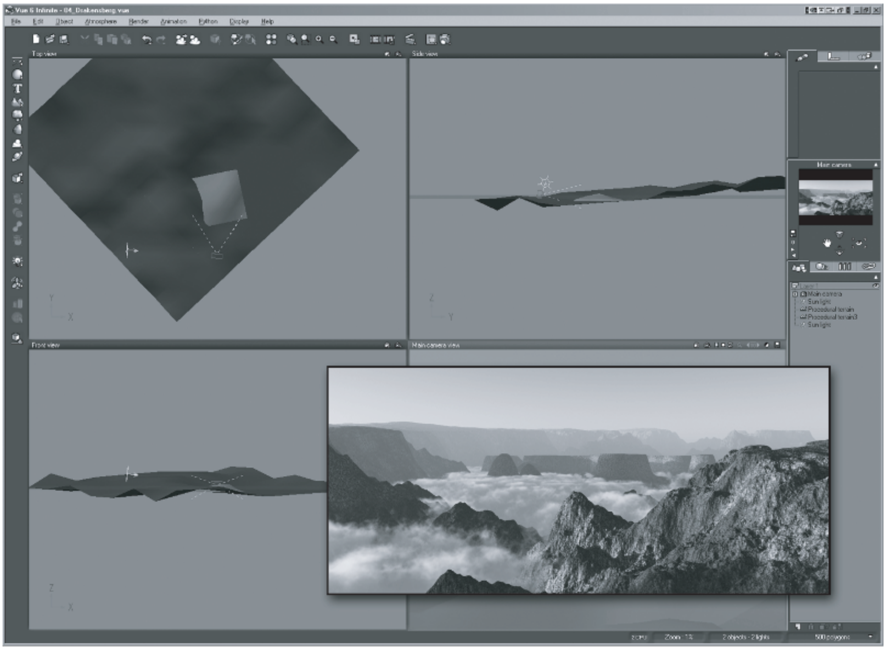

Fig. 3 Scene setup in Vue 6 Infinite and the rendered result

and resources. The abstraction down to basic symbols facilitates the quick generation of a 3D scene of good quality.

- **Plane representation** – The first method of plane representation is suitable only for the background, adding plants or zones covered by vegetation as a picture into the background of a 3D scene. The second method of plane representation, also called “billboard”, enables a first step towards representing single plants as near natural. Here, a surface is generated and equipped with the picture of a tree, a shrub, or any other plant. The empty space of the picture will appear transparent after rendering.
- **Volume representation** – The entire plant to be represented is modeled – even down to every single leaf if required. Depending on the degree of precision, photorealistic plants can be generated in this way.

3.1 Symbols

Using symbols for the representation of plants may not suit everyone, but it certainly makes sense when having to represent the vegetation of a landscape quickly and effectively. In the representation of models, realism is not always needed. A simplified method of representation, as can be done with symbols, in many cases enables the modeler to convey the important contents quickly without the unnecessary “burden” of a costly visualization. In this case, it is best to follow the principle of reduction, i.e. "less is more".

3.2 Plane Representation

The first method of plane representation is exclusively aimed at generating backgrounds. In most cases a photograph or a painted picture (Matte Painting) is used to compose a suitable background for the scene. This method is excellent for large forest scenarios in the background. The example in Fig. 4 shows the silhouette remaining as an area in front of the background. The area to which the forest was assigned reacts to the existing light sources like all other objects in the scene.

The billboard method is another simple plane or surface-based method for generating realistic-looking plants. It should be noted that the greater the distance to the represented plants, the more realistic these will appear. The plant that is to

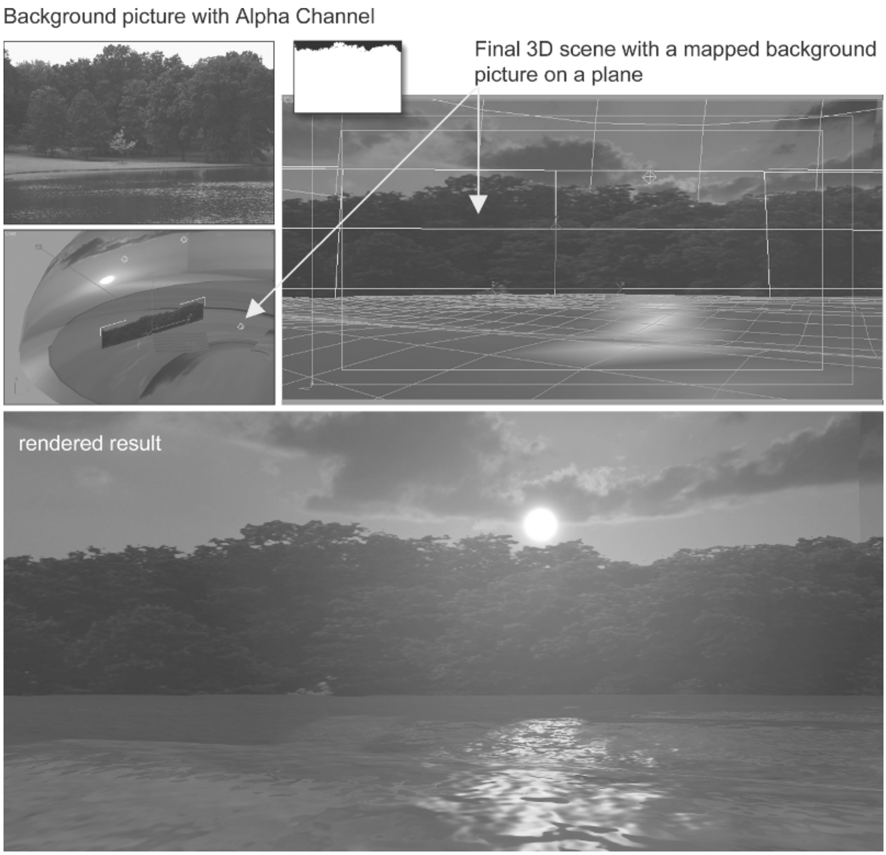

Fig. 4 Background picture with Alpha Channel as texture on one level for a simplified representation of a forest background

be represented is available in a regular picture file and is provided with an Alpha Channel. The Alpha Channel adds transparency information to the picture.

By using an Alpha Channel, certain areas of the picture, depending on the grey scale values, will appear completely transparent or only slightly transparent. The image is mapped as a texture on a simple surface in the 3D program. During rendering, the area equipped with transparency information by the Alpha Channel is calculated as transparent. In other words, it cannot be seen. If a picture format like for instance JPEG cannot support its own Alpha Channel, then a method for supporting their own transparency or opacity channels can be found in most 3D programs (Fig. 5).

For the texture to be used, a grey scale picture is created which takes over the function of the Alpha Channel. The advantage of these opacity channels lies in the fact that not only can the information of the grey scale picture be used for visibility, but within the 3D environment the intensity of the opacity channel can be modified separately.

Not every picture format is suitable for use as a texture with transparency information. The formats supporting an Alpha Channel are, among others:

- TGA – Targa – Truevision Fomat
- BMP – Windows Bitmap Format
- TI(F)F – Tagged Image (File) Format

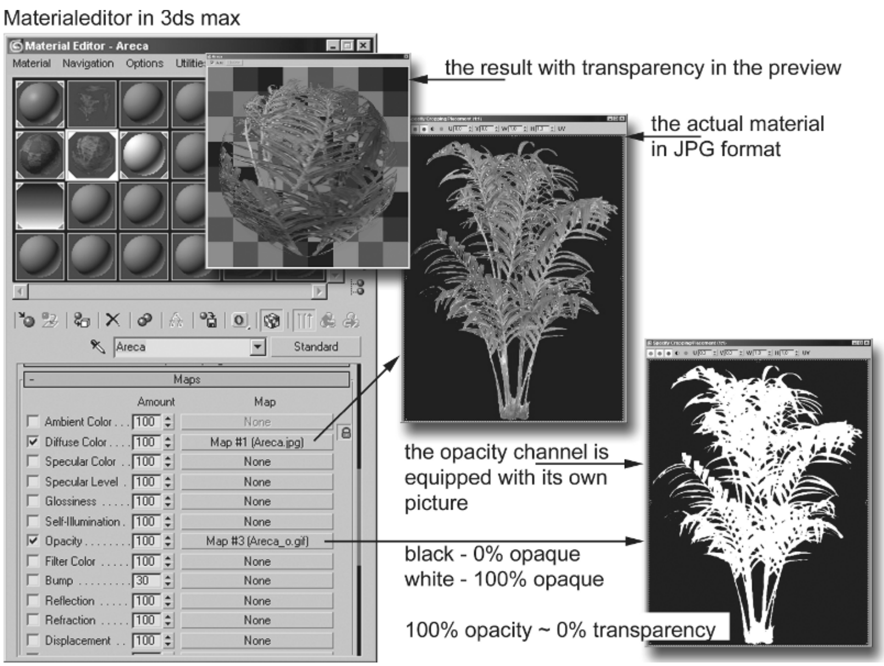

Fig. 5 Creating transparency by a so-called opacity or transparency map

- PNG – Portable Network Graphics
- PSD – Photoshop – original format

An important criterium for a convincing presentation of the pictures used is the quality of the mask. Masks are usually generated in picture editing programs like Photoshop. Depending on the level of experience and the quality of the picture material used, the procedure will be more or less straightforward. The aim is to avoid the conservation of areas, which are supposed to become transparent so that irritating "flashes" do not spoil the image.

Even though billboards are a quick method for creating the impression of near-realistic plants, there are a few important points to consider. In most 3D programs, the different shadow types for "soft shadows" are not able to represent transparency. In this case, the shadow type *Raytrace* must be used, which may on occasion increase calculation time considerably.

A rather simple and quick method to increase the quality of representation in billboard representations is the use of another plane. In this case the existing surface is copied and turned by 90°. One could be tempted to use even more surfaces. However, this would not result in a substantial improvement; therefore the number of surfaces used when working with Billboard types should not exceed two. Figure 6 shows a billboard with a second plane for increased plasticity.

Often the option "look-at" camera is added to the created billboards. This means, that the billboard is always orientated at the camera. So it is guaranteed, that the rendered result shows the full plant or vegetation and not only a small line standing somewhere in the scene.

3.3 Representing Vegetation Volume

In this context we are talking about creating 3D models in a 3D program as realistically as possible. The models can either be designed "manually" by employing

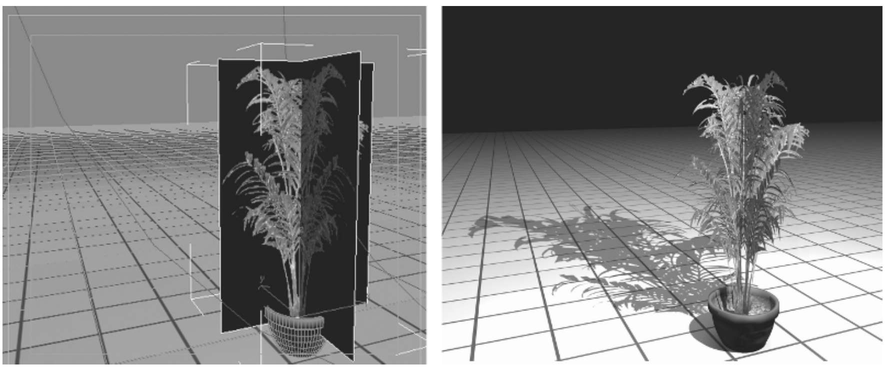

Fig. 6 Billboard with a second plane for increased plasticity

polygons, free form surfaces, or patches, or, alternatively, they can be generated procedurally via the respective algorithms. In the procedural generation, appropriate calculations are used for simulating plant growth. These procedural methods usually generate polygon models.

Deussen (2003) describes different ways of generating "volume" in 3D programs. These are procedural models, regulation of the development of branching, representation via particle systems, and fractal tree models. It would exceed the scope of this contribution to describe the abovementioned methods in more detail, but readers who wish to familiarize themselves with the details of plant growth and the various ways of modeling them, are is recommended to consult the book by Oliver Deussen (2003), which provides an in-depth overview of the complexities of plant simulation.

The advantage of volume modeling is without doubt the high quality reproduction of plant details. The problem is, however, that quality has its price. The sheer number of polygons of a scene, or even of a single plant, can quickly reach the limits of memory capacity.

4 Atmospheric Effects – Important "Add-ons"

With respect to the representation of landscapes and vegetation, atmosphere is synonymous with general mood, frame of mind, or simply the emotional impression which a landscape can evoke in the viewer. The impact of the message of a visualization, indeed of any picture, depends to a large degree upon the "correct" rendition of the atmospheric effects. Fog and mist in a nature scene, haziness in the distance, cloud formations, smoke – all these are important elements for generating a convincing product.

Important details to deal with are color perspective, and the loss of color towards the horizon; mist and fog; the sky depending on the time of day and humidity; the type of cloud cover; rain and snow.

4.1 Color Perspective, Mist and Fog

Especially in landscape representations, the deliberate use of color perspective is not only a means for representing nature convincingly, but also a suitable tool for adding depth to a scene. This effect can be achieved relatively easily, because most 3D programs offer their own fog effects. Some programs specialized in exterior representations go as far as relieving the user of this task by incorporating an automatic color perspective.

Normally the loss of color is achieved by the deliberate addition of white "noise" to the background of a scene. The further the object is away from the camera, the more dense is the noise filter covering the scene, and the weaker and lighter the colors appear. In this respect it has to be kept in mind that the atmospherically loss of color will vary depending on the time of day, the temperature, and the humidity.

On a sunny day in summer, color perspective will appear with a bluish hue, and at sunset on an autumn evening, it will appear more reddish.

4.2 Sky and Clouds

The sky is not always blue. Its color varies according to the time of the day, the humidity and the season of the year. The fact that the sky appears blue (and not black like the sky the astronauts see from the moon), as well as the slow dimming of light after sunset (as opposed to abrupt) can be explained by the diffusion of the light caused by the air molecules and by particles of aerosol (Götz, 1999).

The pure diffusion by molecules is dependent on the wavelengths, increasing to the power of four of the wavelength; in other words, violet light has the highest diffusion factor (it is about 16 times higher than in red light, which has double the wavelength of blue light). The fact that the sky does not appear violet but blue is due to the intensity distribution of sunlight.

Furthermore the sensitivity of the human eye has its maximum in the green range. This overlapping of intensity and sensitivity results in the blue color of the sky.

The red color of the sun at sunset can be explained in the same way as the blue color of the sky. The light has to cover a long distance penetrating the atmosphere of the earth. The blue parts of the light are diffused, which lets the sun appear red. There are especially beautiful sunsets with deep colors when there are many dust or aerosol particles in the atmosphere. When generating a "sky" as a picture background, in most cases a map is used. This can be a normal photograph, any digital picture, or a procedural map which is based on special algorithms.

When trying to create convincing images and representations of nature, it is very important to consider the correct use of sky color (Fig. 7).

4.3 Rain and Snow

Rain and snow are important "small" details to generate the "right" mood of a scene (Buhmann, 2003). Most of the currently available 3D visualization programs are able to create rain and snow via so-called particle systems. Another method is to do some post processing in an application like Photoshop. Some noise and a little soft focus in Photoshop will result in providing rain for a still frame. To go into more detail only makes sense if rain has to be used in an animation. The basic requirements for rain in an animation can be reduced to the following points:

1. A simple particle system for the 3D scene
2. Sufficient particles and a suitable texture
3. A little Motion Blur

Figure 7 shows some atmospheric effects generated with the software *TerrainView-Globe*. Especially in landscape representations adding atmospheric effects like

Fig. 7 Atmospheric effects with TerrainView-Globe.[29] The upper image shows a scene in Switzerland without any atmospheric effects. Distance fog is added to the image in the middle and the image at the bottom has got some additional volumetric cloud layers. All images are screenshots out of a virtual reality environment

distance fog and color perspective is not only a means for representing nature convincingly, but also a suitable tool for adding depth to a scene. Atmospherical effects like the decrease in color richness are necessary, however, to make the 3D representation realistic and convincing (Mach, 2000).

[29] TerrainView-Globe is a virtual globe solution and has been distributed until the end of 2007.

5 Presentation Methods

Finally, the results of a visualization may be presented. This may be done in different ways. Three methods are in common use when considering to prepare a presentation. They are a still picture, also known as a "STILL"; a motion picture, also known as a "MOVIE" and an interactive presentation.

5.1 Stills

The easiest way of creating a still is to create a screenshot from an application using the "Print screen[30]" button on the keyboard. Then, returning to *PowerPoint, Photoshop* or a word processor, the image may be inserted into the application using the "CTRL + V" command. This method may be used in a GIS, a CAD or any other application. If a visualization tool like *3ds max, Cinema 4D* or a similar tool is used, the final picture will have to be rendered.

Rendering is a word used in various differing contexts. It implies picture calculation or generation, and is used as a term in all those programs which generate, by way of complex calculations, new pictures, movies or sounds from digital "raw material" of any kind. The term is used in computer graphics as well as in classic layout. Here, however, it is used for coloring of sketches or scribbles.

Visualization professionals talk of rendering in the field of composing a picture for a website via a browser as well as for calculating a sound sequence for an audio program. In short, the term rendering is used very often and it may have different meanings. With respect to 3D visualizations, rendering is the special procedure which generates a finished picture, a sequence of pictures or a movie, out of the designed 3D scene, the illumination and all the materials. Some basic issues, when dealing with stills are:

- **Image resolution**: How many pixels are allowed for the picture, or, in other words: What are the constraints for the general setup of the 3D animation or the respective still frame?
- **Type of data file**: Which type of format is to be used? Should it be a compressed picture format entailing a certain loss, or would it be better to use a loss-free format?
- **Type of presentation**: Will the finished image be used in a printed book will it be for home use, for a multi-media CD or are the pictures produced for an edition in PowerPoint, for the web or for a poster?

Single images for the web or other screen based applications like for example PowerPoint should be done with at least 96 dots per inch (dpi). Images for the print

[30] These descriptions for shortcuts are only applicable in Microsoft's operating systems.

media should be done with at least 300 dpi for color pictures, and at least 600 dpi for gray scale pictures.

Since the question of the correct resolution comes up again and again, the above formula will be very helpful. It is always advisable to generate a high quality starting size, and to take this as a basis for further use.

- Good formats to begin with are EPS or TIF. Both can administer Alpha Channels, and can be compressed without loss/lossless. These starting formats are also suitable for the generation of still frames. The choice between EPS and TIF is a matter of personal preference. The possibility of administering CMYK certainly indicates a close connection of both formats to the print media. If necessary, it is better to do additional processing with respect to color adaptation achieved via an image editing software such as Photoshop.
- Additional formats: Depending on the field of use, the generation of JPEG or PNG is recommended for multimedia applications. Both can be converted from the basic format.

5.2 Movie

For the edition in movie formats it is best to follow the standards which are by now included as standard features in most 3D programs. Most current movie and video formats will already turn up in lists of the rendering dialog, e.g. PAL (Video) with a resolution of 768×576 pixels (or DV-PAL 720×576).

5.3 Interaction

There is a trend towards using special interactive 3 dimensional applications (Buhmann et al., 2005). It makes sense for a user to be able to navigate on his own, be it when using a communication platform for specialists or simply a medium for presentation. Interaction encourages learning, as the user is able to experience an environment at a speed and in a way that suits himself; it reduces the user's inhibitions to use certain software, and enables further discussions under colleagues. There are diverse ways of visualizing 3D data in Real Time (Buhmann et al., 2006). The quality of visualization can vary considerably and it is hard to find the tool which meets one's needs precisely. It depends on the objectives of the project. Some general demands on Real Time visualization might be described as follows:

1. **Representation of unchanged geometry** – There must be no polygon reduction; if this is unavoidable, it has to remain controllable.
2. **Integration of "large" textures** – These are aerial pictures or other textures compressed without loss.

3. **LOD – Level of Detail** – Elements beyond a defined distance from the camera are simplified.
4. **Velocity** – Fast and easy navigation within the VE (Virtual Environment).
5. **Actions/Behavior** – The option to include links and special behavior like collision or intersection detection.
6. **Operation/Navigation** – Easy use of navigation elements
7. **Platform/Presentation** – Platform independence
8. **Data transfer** – Data interface for the most commonly used programs

These points refer to the viewing of or navigation in Digital Terrain Models (DTM), and to the interaction with this data in an Interactive Environment (IE). The use of extended skills, as in game development environments for instance, has not been considered in this list of demands.

There are dozens of programs which can be used to create interactive 3D scenes, ranging from simple viewers on the one end to fully developed game engines on the other end. They all have one thing in common: the program one uses has to be able to export special data formats; otherwise expensive converter tools have to be obtained. Furthermore, it can be very difficult to master the proper use of many interactive specialized tools, since these are very complex.

When looking for a simple and user-friendly method to present 3D content and designs for a quick result, one will most often end up choosing formats like *VRML and Quicktime VR*, programs like *TerrainView* or authoring systems like *Quest3D* or *Anark Studio*. Each of these formats and applications has advantages and disadvantages. Some of the formats have existed for a long time already. Thus *VRML* and its successor *X3D* are by now well established in the scene of 3D interaction. The good thing is that they are truly platform independent, the bad thing is that the results look different on every single machine, depending on the operating system the viewer is using, like *Blaxxun Interactive* or *Cortona Viewer*, and the graphic card.

Quicktime VR is not really 3D. It is a panorama viewer which pretends to have 3D interactive behavior. All that is needed is the *Quicktime* Player, which many users already have. Very good results can be attained using virtual reality editors like *TerrainView* or 3D authoring applications like *Quest3D* or *Anark Studio*. Virtual globe solutions are comfortable viewer and presentation platforms for digital elevation models and landscapes. A virtual globe solution offers a new way of combining GIS based data (imagery, elevation, vector and annotation) with any kind of 3D content and presenting the result in a spherical earth-like rendering environment.

The "GEO" software industry has realized the importance of *Virtual Globes. ESRI* launched *ArcGlobe* in mid-2003, *Google Earth* was introduced in 2005. *Leica Geosystems* from *Hexagon* distributes *ERDAS Imagine* and *Virtual Explorer* software. *Intergraph* offers a product called *Image Viewer* for *MicroStation. Autodesk* acquired software for sophisticated 3D modeling, visualization and animation (*Civil3D, 3ds Max, Maya*), but, as yet, has no own *Globe* product. *ViewTec* has been developing software products since the late 90's, which are known under the brand name *TerrainView*™. Upstream core technologies required for the realization of applications with *Virtual Globes* are: Remote sensing; Image processing; Geographical and other spatial information processing; Global and local positioning; 3D modeling and texturing. Impressive results may be achieved with such applications.

6 Conclusions

There is a growing need to visualize technical data, especially for decision makers involved in the design of forested landscapes. A great number of tools for planning and design are available and it is usually not feasible if foresters develop such tools. More and more additional features for visualization are being added constantly. Although the software is impressive, it still takes time and effort to develop a convincing visualization presentation, using these available tools.

A great variety of visualization methods are currently available. With specialized programs the user is able to create good stills and realistic animations with many atmospheric effects. The results are sometimes so good that it is difficult to distinguish a rendered image from a good photograph.

But it is not always easy to develop such a high quality visualization. Furthermore, after a still has been rendered or an animation has been composed, it is not possible to quickly change a viewpoint. If one decides to use a different camera position, the whole machinery of 3D high-end rendering has to be started all over again. Thus, a pleasing and comprehensible visualization is in the majority of cases very expensive.

It is possible to make use of interactive technologies to develop visualizations. There are many programs available, and they all have their advantages and disadvantages. But most of them are still rather complicated for use in a quick and easy visualization. Eventually, applications which enable a designer to create an interactive game-like visualization with technical data would be desirable.

Possibly the most important current requirement would be an improvement of the interfaces between technical applications for design and planning, like GIS or CAD and visualization applications like 3D authoring tools or viewer applications. If this can be achieved, then it will be only be a question of time before high quality interactive visualization becomes a standard in presentation, – hopefully not only for decision makers.

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