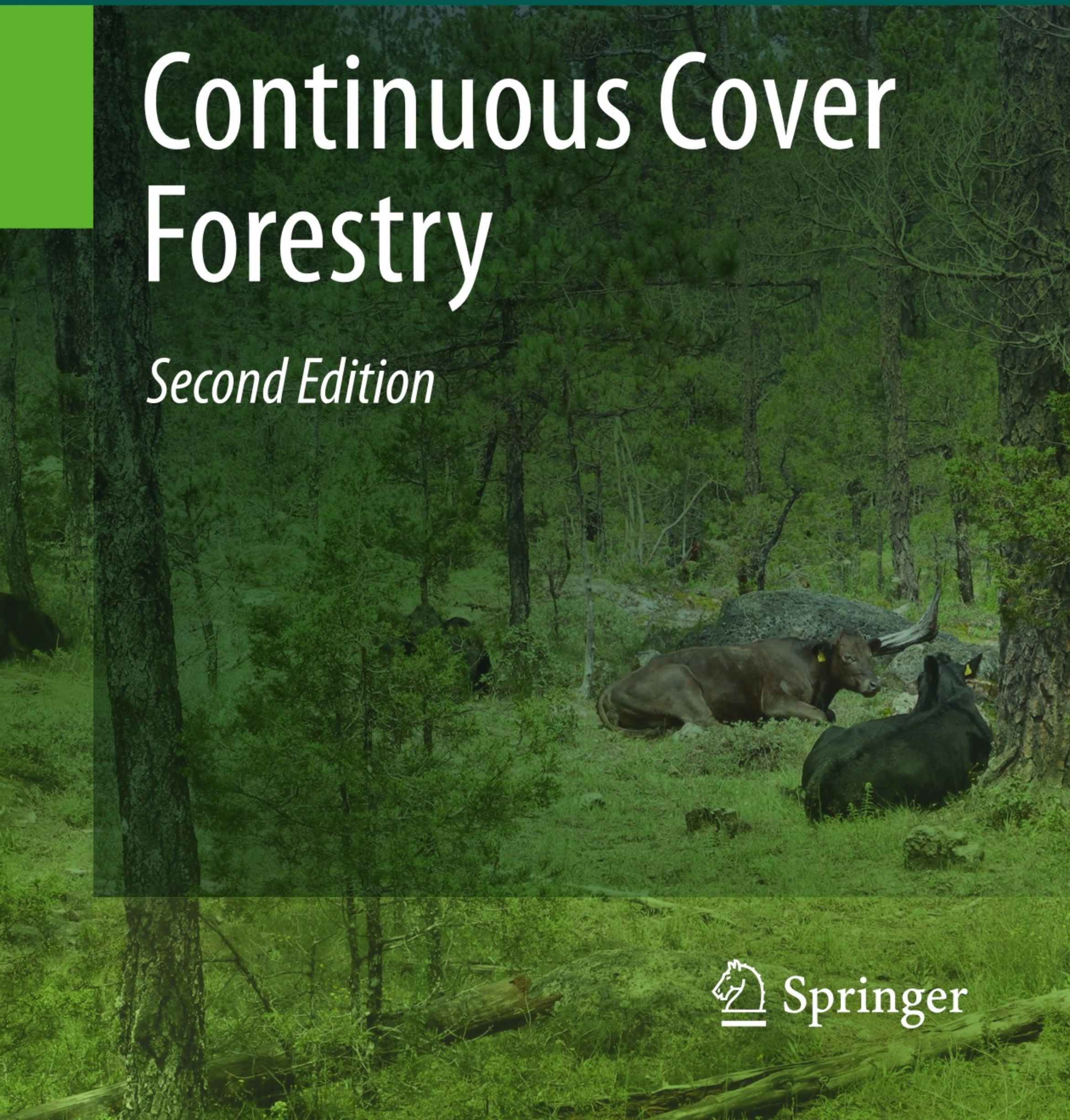
Managing Forest Ecosystems

Timo Pukkala
Klaus von Gadow *Editors*

Continuous Cover Forestry

Second Edition



Springer

Continuous Cover Forestry

Managing Forest Ecosystems

Volume 23

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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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Editors

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ISSN 1568-1319 e-ISSN 1568-1319
ISBN 978-94-007-2201-9 e-ISBN 978-94-007-2202-6
DOI 10.1007/978-94-007-2202-6
Springer Dordrecht Heidelberg London New York

Library of Congress Control Number: 2011940776

Cover illustration: Typical scene in the State of Durango where forests are managed by communities known as Ejidos: management is by selective tree removal, clear-felling is not allowed. Animals (ganado) are part of the multiple use system practiced there. (Photo by K. v. Gadow, autumn 2009)

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Foreword

Although the majority of the world's forest ecosystems are dominated by uneven-sized mixed species stands, forest management practice and theory have focused on the development of plantation monocultures to maximize the supply of timber at low cost. Societal expectations are changing, however, and uneven-aged multi-species ecosystems are often believed to be superior to monocultures in addressing a wide range of expectations.

Rotation forest management (RFM) systems are characterized by standard silvicultural treatments and repetitive cycles of clearfelling followed by planting. Continuous cover forestry (CCF) is characterized by selective harvesting and natural regeneration, resulting in uneven-aged structures and frequently also in multi-species forests. The distinction is usually the result of decisions related to the stand establishment costs, simplicity of management, and various intangible benefits. The oldest and most refined examples of CCF systems are the so-called *Plenter* selection forests found in France, Switzerland, Slovenia and Germany. Today, CCF systems are encountered in various regions of Europe, North America and in some tropical and sub-tropical forests of South Africa, Asia and South America. In a forest managed under the selection system, the stand age is undefined. Forest development does not follow a cyclic harvest-and-regeneration pattern. Instead, it oscillates around some ideal level of residual growing stock, which is assumed to be favorable for natural regeneration and tree growth.

The application of new silvicultural systems – which sometimes is equivalent to a return to traditional methods of tree harvesting – has become a political reality in many parts of the world. This involves a gradual transformation of the current even-aged silvicultural practices towards Continuous Cover Forestry, which is also referred to as "uneven-aged" (North America) or "near-natural" (Europe) forest management. Applications of CCF are usually characterized by uneven-aged multi-species stands, site-adapted tree species mixtures, and selective harvesting. Selective harvesting systems have a long tradition. Specific CCF-related analysis, modeling and economic evaluation tools have been developed, but details about their use are not widely known. Thus, the objective of this volume is to present state-of-the-art research results and methods relating to CCF management with an emphasis on modeling, economics and application examples.

Following the success of the first edition of *Continuous Cover Forestry – Assessment, Analysis, Scenarios* which was published in 2002, Springer approached the current editors with a request to develop an updated edition using a similar title. We have replaced some of the old chapters with new ones and asked the authors who contributed to the first edition, to update their contributions. The nine chapters of this book deal with the structure, silviculture, modeling, economy and optimization of continuous cover forests. The chapters are independent contributions and they can be read in any order. We wish to thank all the authors of the new edition of *Continuous Cover Forestry* for their excellent cooperation and timely submission.

Timo Pukkala
Klaus v. Gadow

Contents

Chapter 1
Historical Emergence and Current Application of CCF*

Jean-Philippe Schütz, Timo Pukkala, Pablo J. Donoso, and Klaus von Gadow

1 Introduction

Almost all of the world's ecosystems are utilized by humans and virtually all primary forests have been domesticated (Kareiva et al. 2007). People manipulate and modify the environment and their activities determine the mix of ecosystem services and the benefits to society. The availability of particular forest services is therefore not so much an ecological, but predominantly a cultural problem. Stream water runoff is not so much affected by "nature", but by foresters' decisions regarding tree cover and age. Tree harvesting activities determine the cash flow, the quality of habitats, nitrogen processes and vistas. A specific strategy, implemented by forest management, generates a particular mix of forest services.

Although the majority of the world's forests are dominated by mixed species stands, forest management practice and theory has focused on the development of monocultures (Gerlach et al. 2002). Industrial timber plantations are often preferred

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[*] **Author contributions** Schütz, Section 2; Donoso, CCF in Chile; Pukkala and Gadow, remaining text.

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_1,

Virgin forest Skole, western Ukraine (Photo: Klaus von Gadow)

because their management is simple and fiber production can be maximized. Plantation forests are established to ensure the supply of a high quantity of wood. Societal expectations are changing, however, and uneven-aged multi-species ecosystems are often believed to be superior to monocultures in addressing a wide range of expectations.

Foresters usually distinguish, albeit somewhat simplistically, two types of silviculture. Rotation forest management (RFM) is characterized by three distinct development phases: establishment – thinning – clearfelling. At least two of these phases, and sometimes all three, occur simultaneously in continuous cover forestry (CCF). According to (Mason et al. 1999), continuous cover forestry is characterized by 'the avoidance of clearfelling of areas much more than two tree heights wide without the retention of some mature trees'. According to Davies et al. (2008), CCF encompasses a range of silvicultural methods, which largely fall into two groups: selection systems and shelterwood systems (we recommend to use the term "irregular shelterwood", which is less restrictive than the classical type). In selection systems, harvesting and regeneration take place simultaneously and continuously, and tree sizes are intimately mixed. The selection systems may further be subdivided into single-tree selection and group selection. Variable-density thinning may also be used in uneven-aged forests to create and promote spatial heterogeneity, which is typical to natural forests. In irregular shelterwood systems, stands are regenerated under the shelter of an existing overstorey, which is gradually removed; the resulting regeneration is more or less even-aged. Davies et al. (2008) recommend that CCF silviculture should be based on three principles:

(a) **Continuous cover**: avoid large clearfellings. According to Mason et al. (1999), the tree cover on areas larger than 0.25 ha should not be entirely removed.
(b) **Stability**: maintain stable forest structures to minimize biotic and abiotic disturbances.
(c) **Naturalness**: use native or site-adapted tree species to support desired levels of biodiversity and stability.

There are other aspects that are more or less closely linked to continuous cover forestry. Through skilful management, continuous cover forestry can achieve substantial benefits in terms of cash flow, biological diversity, ecological resilience and other services (refer to various chapters in this book; see also Pommerening and Murphy 2004).

2 Plenter Forest – Archetype of CCF

Nowadays the plenter forest system (Fig. 1.1) is well known as a comprehensive, original and perfectly sustainable silvicultural system. It is representative paradigmatically for all single tree oriented concepts without demographic interruption, and thus fundamentally different from all silvicultural systems oriented towards collective development of specific tree cohorts. The Plenter forest has no single tree age. All generations of trees are represented. A description of the Plenter forest is presented in virtually all silvicultural text books.

The emergence of the Plenter forest silviculture as a corner stone of conventional silvicultural wisdom went historically through painful debates. In fact, at a time

Fig. 1.1 Cross section through a classic plenter forest, the famous communal forest Couvet, in the Swiss Jura, compartment I/9

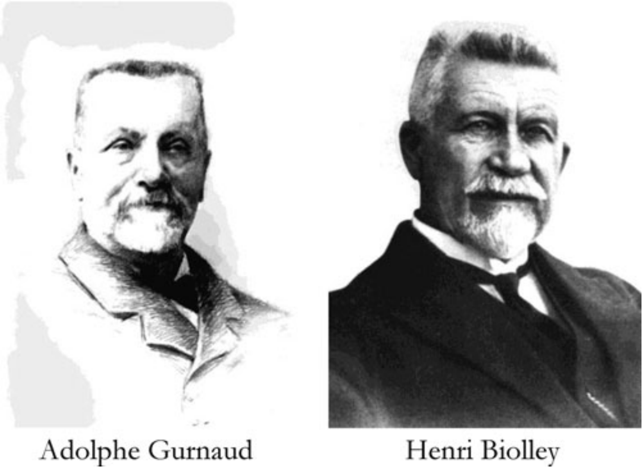


Fig. 1.2 Adolphe Gurnaud (1825–1898) developed the check method for forest management and Henri Biolley (1858–1939), Swiss district forester in Couvet, upgraded the plenter system to a comprehensive and very modern silvicultural system

before the mid-eighteen century, when forests had been overused by uncontrolled timber utilization, the French term *jardinage* or German *Femelung* and *Plenterung* were synonyms for picking trees anywhere without proper planning or control. These methods therefore were associated with the negative connotation of disorder. One of the main tasks of the newly created forest administrations in Western Europe was to rehabilitate extremely devastated forests by strict harvest control and temporary interruptions of harvesting to allow the forest to recuperate without disturbances, particularly after using the forest as pasture. Area-based methods of yield regulation were strictly enforced, which naturally involved the use of clearfelling and organizing the forest in age classes. The first forest laws (France Forest law 1827 and Great Duchy Bade 1833) prohibited explicitly the use of single stem harvesting (*Plenterung*).

At that time only isolated forest professionals, particularly from mountainous regions, like Julius Ernst von Schütz in the Ore Mountains (1757) or Dralet (1820) in the French Pyrenean mountains emphasized that clearfelling is much more detrimental to soil conservation and forest regeneration than application of strict management rules, and that therefore a regeneration without continuous canopy cover should not be undertaken. Dralet uses explicitly the term *jardinage* taking inspiration from the good examples in Switzerland, presumably from the Emmental region, reported by Baron von Tschudi, a pundit in land use. Both these silvicultural pioneers were fiercely criticized by the ruling class.

A trend reversal was initiated by the French Forest Officer Adolphe Gurnaud (1825–1898) who rebutted the ruling forest management system based on area-based regulation (Fig. 1.2). Gurnaud stated in 1878 that the real observed timber increment should be the key (the control) for determining the prescribed cutting and that not rotation age but continuous renewal should be considered leading to multi-age recruitment. His Control Method (or check method) is not merely an inventory method, but includes the vision that silvicultural treatment should be driven by observing the proper development of the stands realizing simultaneously regenerative, nurturing and selective objectives and using the different growth potentials of single trees (Gurnaud 1882, Fig. 1.2). This represented a paradigmatic change in forest use, liberating it from predetermined renewal based on rotation age.

Gurnaud's ideas were met with heavy resistance, particularly by the French forest administration. However, they found enthusiastic support in the neighboring Swiss Jura region, where Henri Biolley had established the check method in 1890 in the communal forest Couvet and successively in the Canton Neuchâtel. His approach has since been the principal management method practiced in those regions. Biolley enhanced the single stem treatment (the plenter system) and formalized it as a comprehensive silviculture system (Biolley 1901) as it is acknowledged today. Full periodic inventory follows the same goals as presented by Gurnaud. Moreover, they allow approaching the ideal standing volume, which ensures the system sustainability. The continuous search for an ideal structure was designated by Biolley (1897) as "experimental treatment".

Since then the plenter system implies a continuous and sustainable tree-by-tree treatment (maintaining the demographic equilibrium) for uneven aged and uneven sized forest, mainly applied in mixed fir-beech spruce mountain forests. Such a treatment had been used empirically and transmitted in a father-to-son tradition, long before its formal definition, mostly in peasant forest regions in the Swiss Emmental, the Black Forest of Baden or the Allgäu in Germany, in the Bregenz district of Austria, in Northern Slovenia and in other regions. A major incentive was the opportunity to sell large fir trees for naval construction to the Netherlands at a good price (the so-called *Dutch cuttings*), which were floated down along the river Rhine (Fig. 1.3). The forest was considered like a savings bank ensuring self-reliance and autarchy of farms. These ideas were more decisive than any theoretical consideration. Thus, farmers continued to treat their forest in that way, albeit against the law.

Gurnaud's original vision had a tremendous impact on foresters' opinions. Thanks to this, the plenter question has been intensively and controversially debated in corporate forestry at the end of the nineteenth century, revealing clear adepts and opponents. The check method found application in different regions in the perialpine belt (Mlinsek 1972) and the plenter system lingered until now, but practically only in the belt of mixed fir dominated forest at montane elevations. The reason is that the perfect irregular system works efficiently with shade tolerant conifers, which are able to maintain spindle-shaped narrow crowns when individualized, in contrast to broad-leaved species whose crowns enlarge rapidly when having more space. This feature gives conifers an advantage in space utilization and allows sustainable production in irregular uneven-sized stands with higher growing stock volume (350–450 m^3/ha) than with a broad-leaved species like beech (250 m^3/ha; Schütz 2006). In fact the particular advantage of the plenter system lies in the production of large

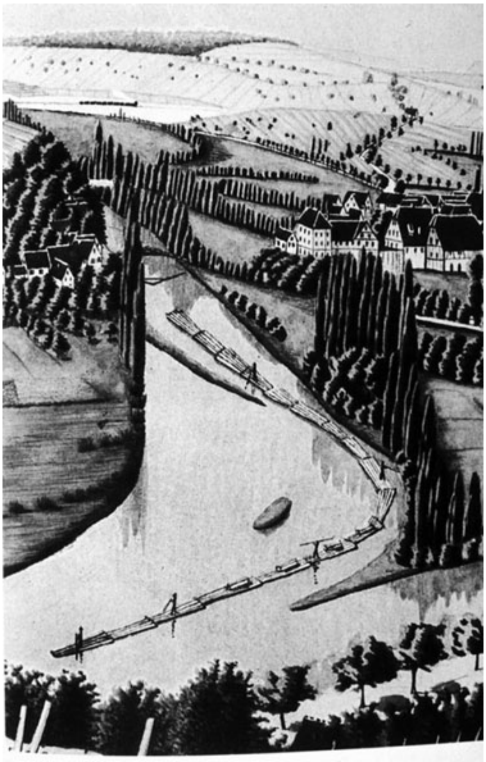


Fig. 1.3 Floating of large fir timber of high quality to a Dutch shipyard. Water color, about 1896

timber sizes of good quality thanks to the positive effect of shade on maintaining thin branches (Fig. 1.4, left) and to minimize the proportion of small tree dimensions.

A comparison of the typical diameter distributions of the age class system and the plenter system shows that the plenter system needs one third less stems in small dimensions and produces 50% more in large dimensions (Fig. 1.4, right). Jointly with utilizing the cost-free natural regeneration and diminution of the cost for tending operations, all this gives the plenter yield system high economic advantages (Mohr and Schori 1999, Knoke 1998).

Despite these advantages the total area managed in the plenter system is relatively modest. A survey by Schütz (2001a) acknowledges something like 370,000 ha of classic fir-beech-spruce forests in temperate Western Europe. These forests occur practically only in montane regions with a natural presence of fir. In the best cases (Slovenia and Switzerland) the plenter forests amount to less than 10%. However, its extent could be much larger. Considering the fact that the plenter forest fulfills almost perfectly practically all forest uses, including their resistance against wind storms (Dvořak et al. 2001) as well as undeniably better resilience, it is astonishing that such a system has not had more success in application. It is not a matter

Pure beech plenter forest from Langula, Thuringia (Photo: Jean-Philippe Schütz)

of specific site conditions. The plenter forest has been practiced successfully in different site conditions. Nor is it a matter of tree species, as illustrated by the pure beech plenter forest in Thuringia.

For Switzerland Schütz (1999) estimates the potential for applying the plenter system in about 56% of the total forested area. Therefore, the main limiting factors are willingness, silvicultural skill and continuity rather than biological constraints. Transforming regular forests into uneven sized ones needs particular skills, enough time (more than 50 years) and some courage, particularly in the starting phase of transformation. Decision support techniques help to evaluate the relevant risks (Schütz 2001b).

3 CCF in Other Regions

Selective harvesting systems have been practiced for long periods of time, but the first scientific methods of calculating sustainable harvest levels were developed not much more than a century ago. Prompted by a rising interest in continuous cover forestry, specific methods have been developed in many regions. This section briefly introduces some approaches.

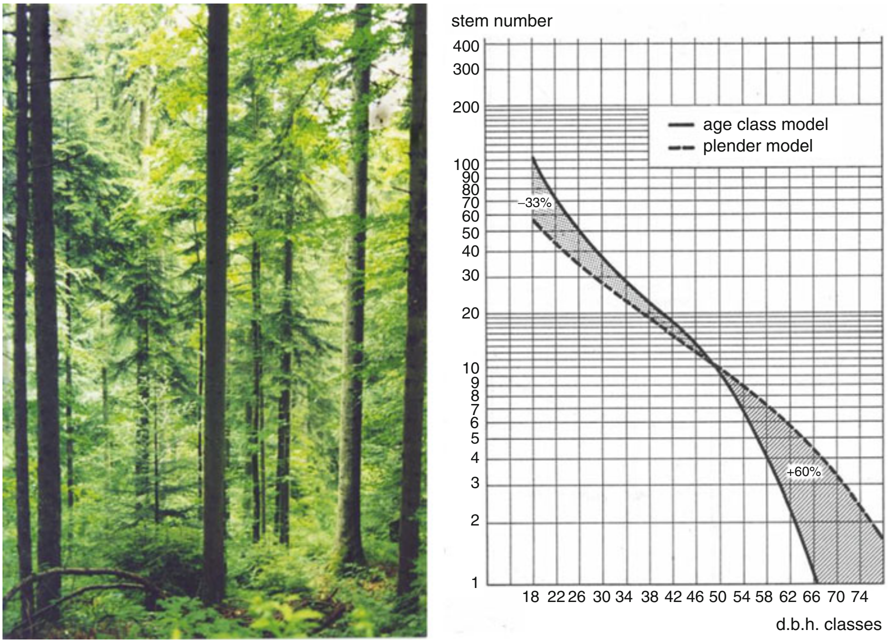


Fig. 1.4 *Left*: The plenter forest allows the landowner to produce large-sized timber of high quality thanks to the beneficial nurturing effect of shade on constraining broad branchiness. *Right*: Comparison of diameter distributions between the age class system and the plenter system for spruce, after Schütz (1997). The age-class system corresponds to a whole chronosequence of a yield table

3.1 China

A frequent objective of CCF management is to mimic natural disturbance by tree harvesting. Hui et al. (2007), for example, proposed that selective harvesting should provide specific forest structural adjustment in mixed uneven-aged hardwood forests in North-Eastern China. Their approach, called “Structure-based Forest Management” (SBFM), uses spatially explicit descriptors of forest structure to adjust forest communities so that they resemble some assumed “natural” state. These variables, which can be assessed during routine forest inventories, are described in more detail in Chap. 2 of this book. In the sequel, we present a brief description of the SBFM approach applied in China.

The assumption is often made that a primary forest, which is not disturbed by human intervention, represents a stable community. This is not entirely true of course. Forest ecosystems are not stationary, but subject to continuous change, due to tree mortality, regeneration and natural disturbances. In the SBFM approach, two situations may occur, a primary forest may be available to serve as reference or not be available.

When a primary forest is available to serve as a reference for management, some of its characteristics may be used to guide harvest decisions. Each ecosystem is unique, and to be able to mimic natural disturbances requires careful observation and continuous learning. The current state of the primary forest will change through time, there is usually not one unique primary state, but a set of forest structures that may be classified as “natural”. Although a definite target state is difficult to define, it is nevertheless useful to establish and monitor reference ecosystems that may guide management. An example is the vision followed by the city of Lübeck in Germany (Fähser 1997).

The SBMF approach to CCF management is guided by a defined degree of naturalness. Beginning with a very limited knowledge, we have to make assumptions about certain structural attributes that may be characteristic and that are likely to be desirable. In the SBFM approach, these attributes include *minimum tree cover*, irregular *spatial distribution* patterns, some degree of spatial *species mingling* and some *diversity of tree sizes* at close range.

Regular assessment of field data provides essential information for evaluating the potential of selective harvest events. There are numerous field sampling methods and each has specific merits. The choice depends on the required detail, available manpower and other resources.

It is not possible to release all trees from competition in a forest with a complex structure and high species diversity. In the SBFM approach, structural adjustment is mainly focused on releasing the larger trees of climax species from competition pressures, and to a lesser degree on creating favorable conditions for natural regeneration. The SBFM approach assumes that selective harvesting is possible when the crown cover is at least 70% of full canopy closure.

The assumed degree of naturalness is evaluated based on several quantitative characteristics, including species relative abundance and spatial mingling, stand basal area, diameter distribution and spatial mingling of tree diameters, and regeneration. Specific structural variables, such as the uniform angle index (W), species dominance (U) and species spatial diversity (TSS) are used to this purpose. These variables are described in Chap. 2 of this book.

The SBFM approach is not limited to natural ecosystems, but may also be applied to man-made or heavily disturbed forests. Seven forest types have been identified to this purpose: (1) Scrub forests, (2) Plantation monocultures with exotic species, (3) Plantations with native or a mix of native and alien species, (4) Mixed plantation forests with only native species, (5) Degraded secondary forests, (6) Productive secondary forests, (7) Primary high forests. Potentially targeted for removal are trees (see Chap. 2 for explanation of variables):

(a) with a *uniform angle index* value equal to 0.25 or less;
(b) with a *species mingling* value equal to 0.25 or less;
(c) with a *dominance* value equal to 0.75 or greater.

Selective harvesting in the Jiaohe Forest, Jilin Province, China (Photo: XiuHai Zhao)

These values can be easily identified without any measurement effort in the field for all dominant trees and implementation is rather uncomplicated. Practical applications of structure-based forest management were developed in three forest regions in China:

(a) in the mixed forests of broad-leaved species and *Pinus koraiensis* of Northeast China,
(b) in the Xiaolongshan forests of Northwest China and
(c) in the evergreen and broad-leaved mixed forests of Guizhou.

Large demonstration areas have been established and a group of forestry technicians has been trained to carry out field operations.

3.2 Chile – Tipo Forestal Siempreverde

Among the 12 forest types which have been identified in Chile, the evergreen forest type (known in Chile as *Tipo Forestal Siempreverde* (TFS) covers 4.15 of the 13.4 million hectares of Chilean native forests, the largest area of all types. 1.4 million hectares of the evergreen forests are located in areas protected by the State (CONAF-CONAMA 1999). Although silvicultural options for the sustainable use of these forests do exist, the majority of the multi-aged native forests are subject to mismanagent through both illegal and legal selective cuttings (Lara et al. 2003). Selective cuttings are authorized to remove up to 35% of the basal area per ha in 5-year cutting cycles. If this schedule is followed, after three re-visits (at year 0, 5 and 10) the original multi-age forest structure may have been completely modified, its natural structure destroyed and its productivity lost. Such legal removals as well as the illegal cuttings have produced thousands of hectares of degraded forests.

It is essential to revert this situation for the old-growth native forests in Chile. There is a need to develop sustainable management approaches for such forests, specifically to provide valuable wood from species tolerant and mid-tolerant to shade, such as *Podocarpus nubigena*, *Laureliopsis philippiana*, *Aextoxicon punctatum, Saxegothaea conspicua*, *Persea lingue*, *Laurelia sempervirens*, *Drimys winteri* and *Eucryphia cordifolia*. Accordingly, several years ago, studies were initiated to assess the potential of selection silviculture in these forests.

Growth and regeneration studies have shown that balanced structures with similar crown covers for small, medium-sized and tall trees, produce more abundant regeneration and better tree growth (Donoso 2002, 2005; Donoso and Nyland 2005; Donoso et al. 2009). In those studies, the diameter distributions showed constant ratios (q) between the number of trees in successive dbh-classes ranging between values of 1.26 and 1.35. These q-factors could serve as an initial reference for regulating the structure by selection silviculture. Nevertheless, it appears necessary to evaluate various alternative structures for achieving sustained forest growth through applications of CCF.

In a representative forest stand on a good site in the low lands of the province of Valdivia, Donoso (2002) proposed to maintain residual basal areas of 33 m^2/ha and maximum basal areas of 47 m^2/ha. These levels are based on observations of the original primary forest which had basal areas close to 100 m^2/ha. The proposed residual basal areas would provide annual basal area growth rates of 1.4 m^2/ha assuming a 10-year cutting cycle. It has been suggested to increase the relative basal area proportion of immature trees and to reduce the proportion of trees of very large dimensions found in the primary forest, and to set the maturity threshold to 90 cm dbh.

In a balanced reverse J-shaped distribution the ratio between successive *DBH* classes is defined by a constant, sometimes referred to as the *k ratio* or *q ratio* (Cancino and Gadow 2001). The *k ratio* defines the curve's shape (its "steepness" or "flatness") and permits the calculation of an ideal number of trees for each *DBH*-class in a stand with a given basal area, resulting in a balanced uneven-aged diameter structure. In such a forest, the assumption is made that the current growth can be removed periodically while maintaining the ideal *DBH* distribution, basal area and stand volume. Donoso (2005) proposed a balanced structure with a q ratio between 1.2 and 1.4 and a balanced crown cover among different crown layers.

Selection harvesting in a Chilean evergreen forest (Photo: Pablo Donoso)

3.3 Mexico – Sierra Madre Occidental

Mexico has a great variety of unique forest ecosystems which are home to about 50% of all the known species of the genus *Pinus* and *Quercus*. The pine-oak forests cover 16% of the national territory and occur throughout the major mountain ranges of the *Sierra Madre Occidental*, the *Sierra Madre Oriental*, the *Sierra Madre del Sur* and the *Transvolcanic Belt* (Rzedowski 1978). Eighty percent of the forest area is owned and managed by about 8,000 rural communities known as *Ejidos* and *Comuninades* who manage their land with some level of governmental control (Thoms and Betters 1998). The predominant forest types on the Mexican *Sierra Madre Occidental* are pine-oak stands, often mixed with *Pseudotsuga*, *Arbutus*, and *Juniperus*. These forests, classified as “Nearctic subtropical coniferous forests”, represent a unique ecosystem with one of the highest species diversities in the world. The altitudes above sea level fluctuate between 1400 and 3000 meters (García 1989, SMN 2006).

The CCF system used in Durango is locally known as the *Método Mexicano de Ordenación de Bosques Irregulares* (MMOBI). It was formally developed in 1944 and first applied in the management plan of the Atenquique forest in Jalisco (Rodriguez 1958). The theoretical basis of the MMOBI system has been described

CCF practiced in El Salto, in the State of Durango, Mexico (Photo: Klaus von Gadow)

by Mendoza and Rodriguez (1959) and Torres (2000). Clear fellings are almost never practiced in the region. Specific applications depend on local site conditions and further details are presented by Hernández-Díaz et al. (2008).

3.4 Knysna Forests South Africa

Various techniques have been devised for ensuring sustainable harvests in CCF systems. The most common type involves recurring visits to a given location and to simply skim off the accumulated increment at each visit. To ensure sustainable yields, the forest is subdivided into *n* blocks each covering one *n*th of the total area. Each block is defined by a specific structure involving the distribution of species and tree sizes. The urgency and priority of harvesting depends on the number of years that have passed since the last harvest event.

The indigenous evergreen forests which occur in the Southern Cape coastal belt of South Africa between Mosselbay and Humansdorp, known as the Knysna forests, represent a natural resource of great cultural, scientific and economic importance (Laughton 1937). Altogether 98 tree species occur in these forests. In multi-species forests, it is obviously useful if tree species and sizes are considered simultaneously.

Grootrivier forest near Knysna, South Africa (Photo: Klaus von Gadow)

An example of such an approach is the *Knysna harvest control method* developed during the 1960’s. In the Knysna CCF approach, forests are classified according to the moisture regime (Breitenbach 1974). The second step involves a classification of different tree species and size classes on the basis of their current and future potential as follows: I = mature trees of commercially valuable species that have reached a minimum threshold diameter; II = immature trees of commercially valuable species which will reach class I when mature; III = other non-commercial tree species; IV = weed trees and undesirable exotic intruders (Fig. 1.5).

Finally, a normal residual basal area distribution is defined for each forest type and tree class. The Knysna approach is easy to implement and has the advantage that not only tree size but also species is considered. However, because of its simplicity, the definition of residual basal areas for particular species tree and size classes is not always straightforward.

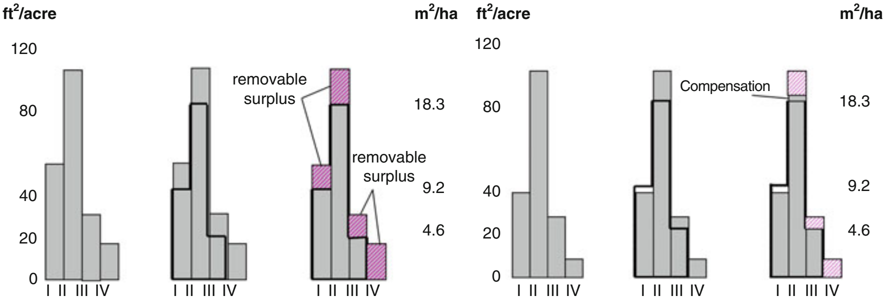


Fig. 1.5 Basal area normalities for a medium-moist forest. *Left*: basal area distribution over tree classes before the harvest, with normal basal area distribution superimposed (*centre*) and removable surplus (*right*). *Right*: The basal area of tree class I is less than the norm. This is compensated by a temporary increase of the norm in class II. The ingrowth into class I will be greater than normal so that the class I basal area will reach normality sooner. The surplus in tree classes III and IV will be harvested

Forest ecosystems may be highly heterogeneous and the structure may change at close range, depending on the terrain and management history. It is thus not always possible to characterize a desirable residual state as easily as in the Knysna forest example. More detailed analyses are often required to adapt rigid prescriptions to specific local circumstances. This requires more specific and spatially explicit descriptions of forest structure.

4 Adaptive CCF

The classical Plenter forest is based on a long-term vision of an ideal forest structure, implemented by the inverse J-shaped diameter distribution. Once attained, that structure is to be maintained in perpetuity by selective harvesting. This approach has been very successfully practiced in Switzerland and a few other regions in Europe. O’Hara and Gersonde (2004) present descriptions of alternative stocking control approaches that have been developed for uneven-aged stands. They discuss the North American *BDq* approach and propose that stand density index can be allocated among diameter classes to form a variety of structures. Similarly, leaf area index can be allocated among age classes or canopy strata without the constraints of a *reverse-J* diameter distribution. In view of fluctuating timber markets and anticipated changes in environmental conditions, a firm commitment to ideal target structures may be inconvenient by limiting the potential for adaptation.

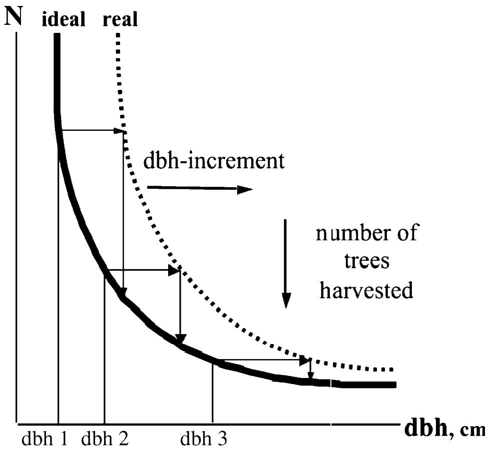


Fig. 1.6 *Left*: Schematic representation of an ideal balanced target structure (*thick line*) and a "real" structure (*dotted line*)

4.1 Ideal Balanced CCF Structures, – Are They Really Desirable?

Uneven-aged forests sometimes exhibit a negative exponential *DBH*-class distribution with a constant *q*-ratio, which is also known in the literature as a *reverse J-shaped* distribution. Figure 1.6 shows schematically an ideal balanced target structure and a real structure.

The "real" curve represents an observed relation between the number of trees per unit area and the breast height diameter. This structure is usually assessed after a period of undisturbed growth. The periodic harvest in a particular diameter class is equal to the difference between the number of trees on the *real* line and the prescribed *ideal* number of trees in that class. An example is the model defined by Susmel (1980) and Virgilietti and Buongiorno (1997) for mixed fir, spruce and beech forests of the Trentino region of the Italian Alps. Trees are harvested with reference to their model, which is defined by the following parameters:

$$N = 300; \quad G = 0.97 \cdot H; \quad V = 0.33 \cdot H^2; \quad D_{\max} = 2.64 \cdot H; \quad k = \frac{4.3}{H^{1/3}}$$

where N is the number of stems per ha with a breast height diameter greater than or equal to 17.5 cm; H is the potential tree height, i.e. the maximum dominant height compatible with the ecological characteristics of the site (m); G is basal area (m^2/ha); V is volume (m^3/ha); D_{max} is maximum diameter (cm) and k is a constant ratio between the number of trees in a given diameter class i and the next class $i+1$. A similar example of such a "guide curve" approach, known in North America as the *BDq* method, that serves as a target stand structure, is presented by Guldin (1991) for a pine/oak forest in the Southern USA. He defines a target basal area (*B*), a maximum residual squared DBH (*Dq*) and the shape of the guide curve. Tabaku (1999) analysed the diameter distributions of uneven-aged plenter forests in Albania

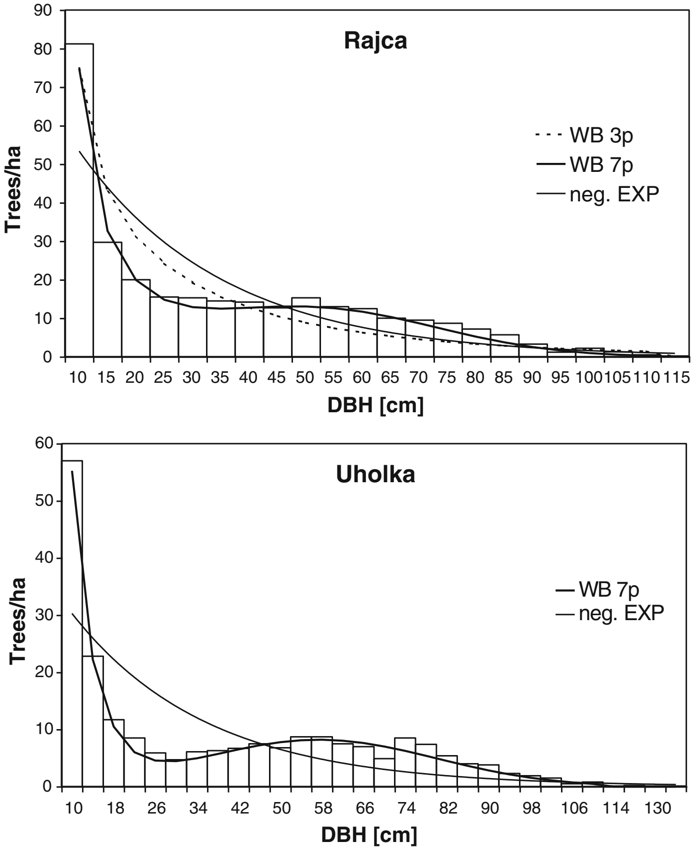


Fig. 1.7 Observed diameter frequencies and diameter distribution models of two virgin forests of European Beech. The *thick solid line* represents the fitted bimodal Weibull-function (Westphal et al. 2006)

and in the district of Bleicherode in Thüringen, Germany. The main species was *Fagus sylvatica*. Tabaku's examples also represent negative exponential dbh-class distributions. Wehenkel et al. (2011) also found similar relationships between the tree diameters and the number of trees in pine-dominated Mexican forests which have been subject to CCF management.

4.2 Natural Beech Forests

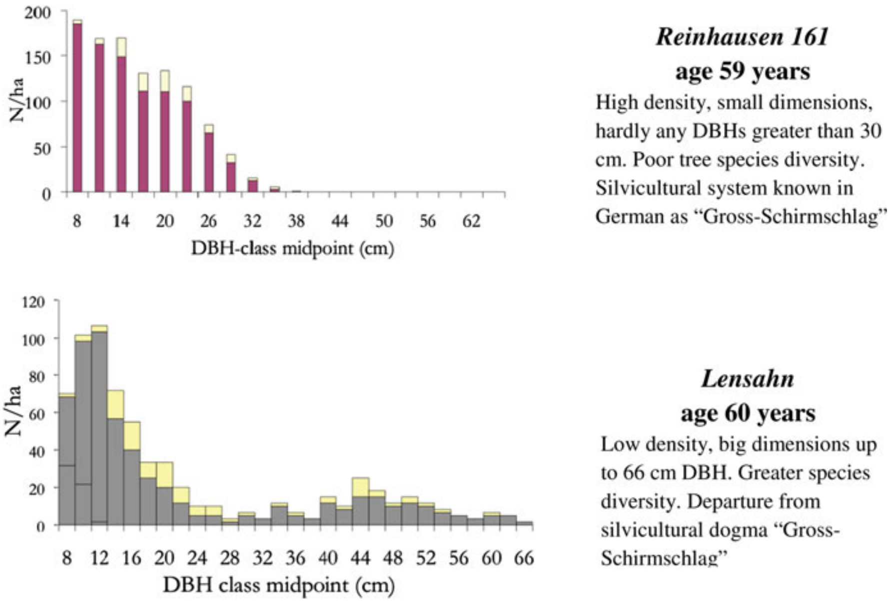
The ideal target structure is an artificial construct which is easy to implement and thus useful for management. The problem is to define it. This problem is aggravated when foresters wish to mimic natural processes by harvesting timber. Westphal et al. (2006) studied the diameter frequency distributions of several virgin Forests of European Beech. Two of their distributions are presented in Fig. 1.7.

Figure 1.7 shows the observed diameter frequencies, and the fitted negative exponential distribution (*neg. EXP*) as well as the unimodal (*WB 3p*) and bimodal (*WB 7p*) Weibull distributions. The virgin structures are clearly bimodal and the modification by management, to fit the negative exponential distribution, would create an artificial structure which is not compatible with the natural one. The assumption that there is a unique and ideal diameter distribution cannot always be substantiated by empirical evidence. The concept of an ideal diameter distribution does not always have a biological foundation and the economic benefits of maintaining a forest in some equilibrium state, which requires constant periodic harvests, are doubtful. The application of the guide curve method for managing forest ecosystems can therefore be recommended only for situations where there is a lack of better analytical tools required for generating and evaluating alternative treatment options. Most important among such tools are growth models which enable foresters to predict forest development in response to a given series of harvest operations. Such models were presented, for example, by Schütz (2006) to predict the "demographic dynamics" of a beech plenter forest in eastern Germany.

4.3 Lensahn – An Example of Adaptive CCF

In a great CCF silviculture has a long tradition in central Europe. It has been practiced in great variety of forests with different structures, silvicultural preferences and forms of ownership. As a consequence, there are no fixed rules that are universally applicable to all CCF forests. Each locality has very particular objectives and constraints and the multitude of specific cases eludes meaningful generalization and scientific abstraction. Mitscherlich (1952), for example, could show that there is no one unique composition, but a wide variety of Plenter forest structures in the Black Forest of Southern Germany that could be designated as *balanced*. Forest development is greatly influenced by harvest events, specifically by the type and weight of thinnings in the early phase after regeneration (Fig. 1.8).

A heavy thinning at a young age soon produces large tree dimensions. Weak thinning and high densities are used to obtain high volume increment per unit area. This effect is well-known and the diameter distribution reveals much about the past silvicultural treatment. Figure 1.8 shows the diameter distributions of two German beech forests which had been naturally regenerated at the same time under a CCF type of silviculture known as *uniform shelterwood system*. Both forests enjoy excellent site conditions. Sixty years after regeneration, the difference between the two structures is striking. At the age of 59 years, Reinhausen is characterised by high stem numbers and small dimensions. Except for a few Ash (*Fraxinus excelsior*) specimens, almost all the individuals are beech (*Fagus sylvatica*) trees.



Reinhausen 161
age 59 years
High density, small dimensions, hardly any DBHs greater than 30 cm. Poor tree species diversity. Silvicultural system known in German as “Gross-Schirmschlag”

Lensahn
age 60 years
Low density, big dimensions up to 66 cm DBH. Greater species diversity. Departure from silvicultural dogma “Gross-Schirmschlag”

Fig. 1.8 Diameter distributions reflecting past silviculture in two beech forests growing in optimum site conditions. The *light shading on top* of the columns indicates the removals during the last harvest event

The Reinhausen structure at this stage of development is an example of inflexible CCF: long periods during which trees cannot be harvested (except for firewood) are followed by periods during which trees must be harvested (to pre-empt red heart formation), for example. The Lensahn structure exhibits a significant departure from the shelterwood silvicultural dogma (known in Germany as *Gross-Schirmschlag*).

The Lensahn forest, at virtually the same age, is characterized by considerably fewer trees per ha, a high species diversity and a bimodal distribution of diameters. A considerable part of the population has DBHs exceeding 30 cm, and individual beech trees have reached well above 50 cm DBH. Trees of varying dimensions and different species may be harvested at any time, depending on the market. But there is no urgency and no need to salvage anything because even the large size individuals are still very young. Thus, the Lensahn structure shows high adaptivity. This adaptivity can be maintained for a long time, possibly even indefinitely, if traditional silvicultural dogma is ignored. Adaptivity and structural diversity are complementary features of the CCF approach practiced in Lensahn.

CCF forest Lensahn in Northern Germany (Photo: Klaus von Gadow)

5 CCF and Near-Natural Forest Management

One of the current trends in forestry is increased adoption of near-natural or close-to-nature management principles so as to maintain the biological diversity of forests. The rationale behind this trend is that since natural forests and natural forest dynamics are able to maintain viable populations of all forest-dwelling species, forest management that follows the natural forest dynamics should also be good for biodiversity maintenance.

Natural forests are characterized by their specific species succession and disturbance dynamics. In near-natural management, silvicultural treatments should mimic the species succession and disturbances that occur in natural forests during different stages of stand development. Oliver and Larson (1996) separate four distinct stages of stand development after a major disturbance: (1) stand initiation stage, (2) stem exclusion stage, (3) understorey reinitiation stage, and (4) old-growth stage. As stated earlier in this chapter, RFM is characterized by three phases: planting, thinning and clearfelling. Table 1.1 shows how these phases correspond to the different stages of natural stand development. It can be seen that two stages, namely the understorey reinitiation stage and the old-growth stage are omitted in RFM. As a result, managed even-aged forests lack these stages making forested

Table 1.1 Different stages of stand development in natural stand dynamics after a major disturbance, and in plantation forestry

Natural dynamics	Plantation forestry
Stand initiation stage	Planting
Stem exclusion stage	Low thinning
Understory reinitiation stage	–
Old-growth stage	–
Major disturbance	Clearfelling

landscapes simpler as compared to natural forests, and offering smaller number of different habitat types for forest-dwelling species. As a consequence, the structural diversity of forests is reduced and many species dependent on habitats representing the undestorey reinitiation and old-growth stages are in danger. Using seed tree and shelterwood cuttings with natural regeneration, instead on clearfelling and planting, is an improvement since the former regeneration methods mimic the understory reinitiation stage. However, management that corresponds to the old-growth stage is still missing.

The smallest and most suppressed trees die during the stem exclusion stage. Low thinnings correspond to this stage. Later on, during and after the understorey reinitiation stage, the situation is reversed and the largest trees are now the ones which die most frequently. Therefore, after the stem exclusion stage, thinnings should be conducted as high thinning. In the old-growth stage, individual large trees or groups of trees die creating canopy gaps of different sizes. The stand is uneven-aged and uneven-sized, and often spatially heterogeneous. The stand often turns from a pure stand of a pioneer species into a mixed stand of several species when shade-tolerant species appear under the canopies of shade-intolerant pioneer species.

In the boreal forests of Europe, pine, aspen and birch often occupy the site after a major disturbance, and spruce gradually enters the stand as an understory. Planting spruce on clear-felled areas is widely practiced, but it is clearly against the natural forest dynamics. Since some pioneer species such as pine may survive for 500 years, and hardwoods frequently regenerate in canopy gaps, the old-growth forest often appears as a mixed forest. However, on the poorest sites, where only pine thrives well, the understory and the dominant canopy layers may both consist of only pines.

In the Nordic countries, foresters have been educated to think that the current RFM quite well follows the natural disturbance dynamics of forest. It was assumed that, after the stem exclusion stage, when the trees get older, the stand becomes vulnerable and very sensitive to fire and wind damages, which sooner or later destroy the whole stand. However, more recent and careful analyses have revealed that this is not usually the case (Keto-Tokoi and Kuuluvainen 2010). A major disturbance after the stem exclusion stage is just one possibility, but it is more likely that, after sufficient natural thinning, the stand passes to the understorey reinitiation stage and then to the old-growth stage. The old-growth stage often lasts for 200–300 years (Keto-Tokoi and Kuuluvainen 2010).

The erroneous assumption about shorter intervals between major disturbances is partly due to the fact that, when historical fire events have been analysed,

Undestorey reinitiation and old-growth stages are not recognized in RFM (Photo: Marcus Walsh (*top*) and Tommi Taipale (*bottom*)) human-induced forest fires have not been separated from natural fires. Natural fires represent only a small fraction of all fires. They are lit by strikes of lightning, which are associated with thunderstorms and heavy rains that soon stop the fire. Therefore, natural wildfires are not as devastating as human-induced fires. In many cases a significant part of trees survives the fire. Large pines are the best survivors, which means that natural forests may contain very large and very old pines. Compared to the clearfellings of the current RFM, the major disturbances occurring in natural forests are less frequent and less severe.

In near-natural forestry, the type, geographical scale and interval between cuttings should all resemble the natural disturbance dynamics. Since disturbances vary in severity, interval and area, all these variations should also be integrated into the forest management systems. This calls for parallel use of several silvicultural systems. For instance, if the interval between major disturbances varies from 100 to 500 years (Keto-Tokoi and Kuuluvainen 2010), the interval between heavy "final" fellings should also vary from 100 to 500 years. Some stands may be cut after 100 years, but most stands should be left to continue growing. The severity and size of cuttings should correspond to those of natural disturbances, which means that some large clearfellings should be conducted, but most cutting operations should be restricted to smaller areas and they should remove only a part of the trees.

CCF offers the possibility to convert the current RFM into more near-natural forestry. This transformation can be achieved as follows. The even-aged young stand obtained in a regeneration area is first subjected to low thinnings that mimic the stem exclusion stage. Subsequent thinnings are conducted more and more from above, by removing mainly financially mature trees. In addition to producing income, the purpose of thinnings is to promote undestorey development. The overstorey trees are gradually removed, except that some large trees are left as retention trees. The undestorey, which is uneven-sized and often also uneven-aged, is subjected to repeated high thinnings. If the gradual regeneration slows down or stops, larger canopy gaps are opened to encourage the regeneration of shade-intolerant species, under which shade-tolerant species start to enter again. Finally, the stand may be regenerated with the shelterwood method and in some cases also with heavier fellings.

This kind of management, which need not necessarily be similar in all stands, is often more profitable than RFM since the intervals between expensive artificial regeneration efforts are longer, and harvests mainly remove financially mature large trees having high opportunity cost (stumpage value) but low relative value increment. Smaller trees with low opportunity cost and high relative value increment are left to continue growing. The management is ecologically preferable since it follows the natural forest dynamics. However, to offer habitats for all species, care should be taken to leave some large trees to die and become coarse woody debris. Uneven-sized mixed stands are better protected against several biotic and abiotic risks than the even-aged RFM monocultures. Management is also less risky against economic fluctuations since the intensity, timing and assortment distribution of harvests can be flexibly adjusted to the market situation. In even-aged plantation forestry, the time and assortment distribution of cuttings are largely dictated by the stage of stand development reducing the potential gains that may be obtained by adaptive forest management.

6 Economic Aspects of CCF

The profitability of a forest management system is most conveniently measured by the sum of the net present values (NPV) of all future incomes and costs associated with the management. This sum is called as land expectation value (LEV) or soil expectation value (SEV). A logical reference point of the calculation in RFM is a clear-felled area, i.e. bare land; if the forest landowner plans to invest to a new tree generation, she wants to evaluate the profitability of this investment done at this time point. The NPV calculated for the investment and the subsequent management schedule tells the forest owner, which amount of money, on a savings account, is able to produce the same amounts of net incomes as the proposed forest management schedule.

The main economic problem of RFM is the long time interval between the investment and the major incomes. When discounting is used, the NPV of the whole management schedule easily turns negative if the discount rate is not very low (Fig. 1.9). As a result, even-aged plantation forestry is seldom profitable with slow-growing species involving long rotations.

In uneven-aged management the investment is equal to the value of the trees of the residual stand. The investment may also be calculated as the reduction of the final felling income when only a part of trees are cut instead of all trees. The investment may be larger or smaller than in RFM, depending on the harvesting costs and roadside values of different tree sizes. The investment may be even zero when the residual trees are so small that their roadside value is smaller than their harvesting cost (the stumpage value of the residual trees is zero).

The advantage of uneven-aged management is that the investment produces incomes sooner than in RFM. Therefore, the NPV may be higher than in RFM even if the non-discounted incomes are smaller (Chap 5). As a consequence, uneven-aged management usually becomes more profitable than RFM when the discount rate increases.

Another advantage on uneven-aged management is that the initial investment, i.e., the value of the residual tree stock can be flexibly adjusted on the basis of discount rate. In an extreme case, with very high discount rates, only non-marketable small trees may be left in the residual stand, resulting in no monetary loss and zero investment. If the cutting produces positive net incomes, the NPV of the management schedule is positive irrespective of the discount rate. On the other hand, plantation forestry cannot omit the stand establishment costs and, as a consequence, the NPV of the whole management system inevitably turns negative with high discount rates. Therefore, optimal CCF becomes more profitable than even-aged plantation forestry with increasing discount rate. RFM may be more profitable at low

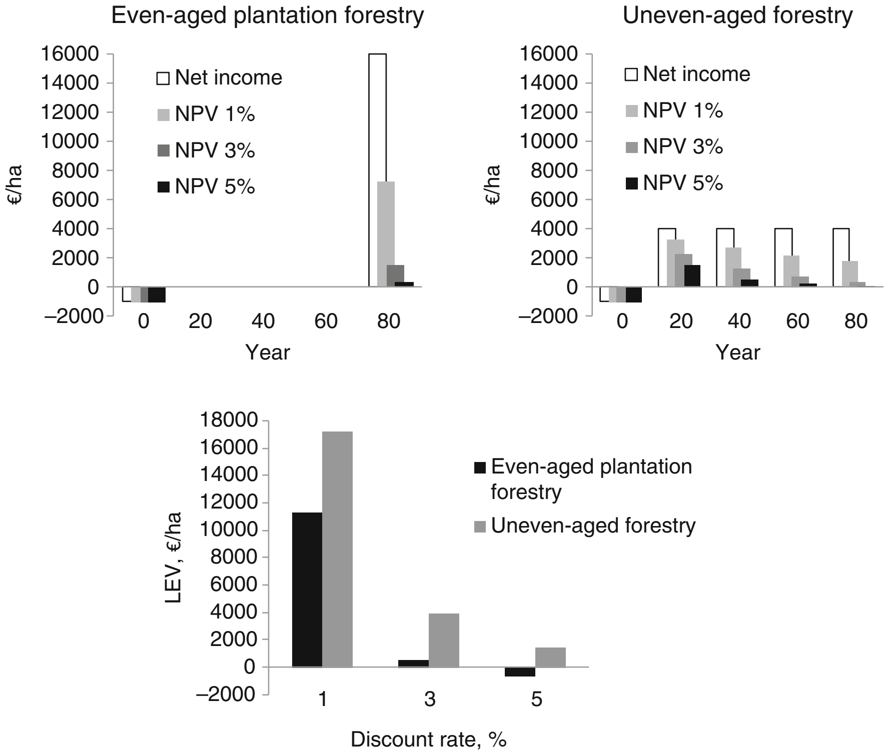


Fig. 1.9 Investments (negative net incomes in year 0) and incomes in RFM without thinnings (*top left*) and steady-stage uneven-aged management (*top right*). The NPVs indicate the discounted values of the investments and incomes. The *lower* diagram shows the land expectation value with different discount rate for the even- and uneven-aged management schedules depicted in the *upper* diagrams. Increasing discount rate improves the relative performance of uneven-aged management

rates, especially if the stand establishment costs are small compared to harvesting incomes, rotation is short, and economies of scale are significant in harvesting.

Another financial benefit of uneven-aged management and several other forms of CCF is that trees can be removed immediately once they become financially mature (Fig. 1.10). This gives the forest owner the possibility to maximize the sum of profits generated by the forest and other investments. This kind of "precision-silviculture" is not possible in even-aged forestry where all trees are removed simultaneously. If the relative value increment is not exactly the same in all trees, clearfelling means cutting some trees too early and some trees too late as compared to the optimal timing (Chap. 5). If harvesting costs are ignored, the profitability of a clearfelling system is inevitably lower than in gradual selective cuttings. The way in which harvesting costs depend on the removed volume and size of removed trees determines whether the economies of scale associated with clearfelling compensate for the inoptimality losses associated with a simultaneous cutting of all trees.

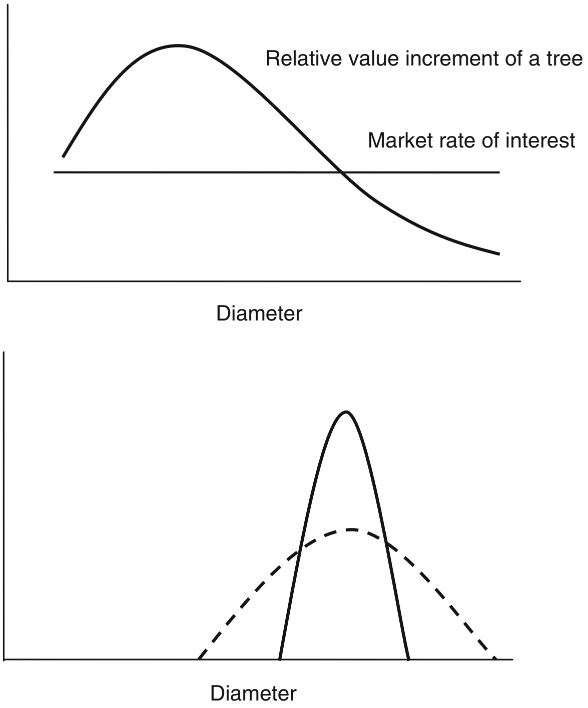


Fig. 1.10 The optimal time to harvest a tree is when its relative value increment falls below the rate of interest in financial market (or slightly earlier since the opportunity cost of land and the effect of removal on the growth of remaining trees should be taken into account). The inoptimality losses due to clearcutting increase with increasing variation in tree size; in the *lower* diagram, the benefit of conducting a high thinning instead of clearfelling is higher in the stand represented by the *wider* diameter distribution (*dashed line*)

It is noteworthy that the optimal management and rotation length of an even-aged stand depend on the size distribution of the trees, not only on stand density and age. If the distribution is narrow, all trees may be cut simultaneously when an average tree reaches the financial maturity. However, if the distribution is wide, optimal management involves one or several high thinnings that remove only the mature trees. The final felling, removing the last trees, occurs later and is determined by the smallest trees of the stand. If partial cuttings are excluded, a wider distribution shortens the optimal rotation length, which largely depends on the financial maturity of the largest and most valuable trees of the stand.

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Chapter 2
Forest Structure and Diversity*

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1 Introduction

Structure is a fundamental notion referring to patterns and relationships within a more or less well-defined system. We recognise structural attributes of buildings, crystals and proteins. Computer scientists design data structures and mathematicians

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*__Author contributions__ Lead author, text and calculations (unless indicated otherwise); co-authors, design of the chapter and contribution to data sets.

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_2,

analyse algebraic structures. All these tend to be rigid and complete. There are also open and dynamic structures. Human societies tend to generate hierarchical or networking structures as they evolve and adjust to meet a variety of challenges (Luhmann 1995). Flocks of birds, insect swarms and herds of buffalo exhibit specific patterns as they move. Structure may develop as a result of a planned design, or through a process of self-organization. Structure is an open-ended theme offering itself to interpretation within many disciplines of the sciences, arts and humanities (Pullan and Bhadeshia 2000).

Important theoretical concepts relating to biological structures include *self-organisation*, *structure/property relations* and *pattern recognition*. *Self-organisation* involves competition and a variety of interactions between individual trees. Selective harvesting of trees in continuous cover forest (CCF) management modifies growing spaces and spatial niches. Physics and materials science have greatly contributed to structural research, e.g. through investigations of *structure–property relationships* (Torquato 2002). Biological processes not only leave traces in the form of spatial patterns, but the spatial structure of a forest ecosystem also determines to a large degree the properties of the system as a whole. Forest management influences tree size distributions, spatial mingling of tree species and natural regeneration. Forest structure affects a range of properties, including total biomass productions, biodiversity and habitat functions, and thus the quality of ecosystem services. The interpretation of tree diameter distributions is an example of *pattern recognition* often used by foresters to describe a particular forest type or silvicultural treatment.

"Forest structure" usually refers to the way in which the attributes of trees are distributed within a forest ecosystem. Trees are sessile, but they are living things that propagate, grow and die. The production and dispersal of seeds and the associated processes of germination, seedling establishment and survival are important factors of plant population dynamics and structuring (Harper 1977). Trees compete for essential resources, and tree growth and mortality are also important structuring processes. Forest regeneration, growth and mortality generate very specific structures. Thus, structure and processes are not independent. Specific structures generate particular processes of growth and regeneration. These processes in turn produce particular structural arrangements. Associated with a specific forest structure is some degree of heterogeneity or richness which we call diversity. In a forest ecosystem, diversity does, however, not only refer to species richness, but to a range of phenomena that determine the heterogeneity within a community of trees, including the diversity of tree sizes.

Data which are collected in forest ecosystems have not only a temporal but also a spatial dimension. Tree growth and the interactions between trees depend, to a large degree, on the structure of the forest. New analytical tools in the research areas of *geostatistics*, *point process statistics*, and *random set statistics* allow more detailed research of the interaction between spatial patterns and biological processes. Some data are continuous, like wind, temperature and precipitation. They are measured at discrete sample points and continuous information is obtained by spatial interpolation, for example by using kriging techniques (Pommerening 2008).

Where objects of interest can be conveniently described as single points, e.g. tree locations or bird nests, methods of point process statistics are useful. Of particular interest are the *second-order statistics*, also known as *second-order characteristics* (we are using the abbreviation SOC in this chapter) which were developed within the theoretical framework of mathematical statistics and then applied in various fields of research, including forestry (Møller and Waagepetersen 2007; Illian et al. 2008). Examples of SOCs in forestry applications are Ripley's K and Besag's L function, pair and mark correlation functions and mark variograms. SOCs describe the variability and correlations in marked and non-marked point processes. Functional second-order characteristics depend on a distance variable r and quantify correlations between all pairs of points with a distance of approximately r between them. This allows them to be related to various ecological scales and also, to a certain degree, to account for long-range point interactions (Pommerening 2002).

In landscape ecology researchers may wish to analyse the spatial distribution of certain vegetation types in the landscape. Here single trees are often not of interest, but rather the distribution of pixels that fall inside or outside of forest land. Such point sets may be analysed using methods of *random set statistics*.

Structure and diversity are important features which characterise a forest ecosystem. Complex spatial structures are more difficult to describe than simple ones based on frequency distributions. The scientific literature abounds with studies of diameter distributions of even-aged monocultures. Other structural characteristics of a forest which are important for analysing disturbances are a group which Pommerening (2008) calls *nearest neighbor summary statistics* (NNSS). In this contribution we are using the term *nearest neighbor statistics* (abbreviated to NNS). NNS methods assume that the spatial structure of a forest is largely determined by the relationship within neighborhood groups of trees. These methods have important advantages over classical spatial statistics, including low cost field assessment and cohort-specific structural analysis (a cohort refers to a group of reference trees that share a common species and size class, examples are presented in this chapter). Neighborhood groups may be homogenous consisting of trees that belong to the same species and size class, or inhomogenous. Greater inhomogeneity of species and size within close-range neighborhoods indicates greater structural diversity.

The evaluation of forest structure thus informs us about the distribution of tree attributes, including the spatial distribution of tree species and their dimensions, crown lengths and leaf areas. The assessment of these attributes facilitates a comparison between a managed and an unmanaged forest ecosystem. Structural data also provide an essential basis for the analysis of ecosystem disturbance, including harvest events.

The structure of a forest is the result of natural processes and human disturbance. Important natural processes are species-specific tree growth, mortality and recruitment and natural disturbances such as fire, wind or snow damage. In addition, human disturbance in the form of clearfellings, plantings or selective tree removal has a major structuring effect. The condition of the majority of forest ecosystems today is the result of human use (Sanderson et al. 2002; Kareiva et al. 2007). The degradation or invasion of natural ecosystems often results in the formation of so-called novel ecosystems with new species combinations and the potential for change in ecosystem functioning. These ecosystems are the result of deliberate or inadvertent human activity. As more of the Earth's land surface becomes transformed by human use, novel ecosystems are increasing in importance (Hobbs et al. 2006). Natural ecosystems are disappearing or are being modified by human use. Thus, forest structure is not only the outcome of natural processes, but is determined to a considerable extent by silviculture.

Structure is not only the result of past activity, but also the starting point and cause for specific future developments. The three-dimensional geometry of a forest is naturally of interest to silviculturalists who study the spatial and temporal evolution of forest structures (McComb et al. 1993; Jaehne and Dohrenbusch 1997; Kint et al. 2003). Such analyses may form the basis for silvicultural strategies. Important ecosystem functions, and the potential and limitations of human use are defined by the existing forest structures. Spatial patterns affect the competition status, seedling growth and survival and crown formation of forest trees (Moeur 1993; Pretzsch 1995). The vertical and horizontal distributions of tree sizes determine the distribution of micro-climatic conditions, the availability of resources and the formation of habitat niches and thus, directly or indirectly, the biological diversity within a forest community. Thus, information about forest structure contributes to improved understanding of the history, functions and future development potential of a particular forest ecosystem (Harmon et al. 1986; Ruggiero et al. 1991; Spies 1997; Franklin et al. 2002).

Forest structure is not only of interest to students of ecology, but has also economic implications. Simple bioeconomic models have been criticised due to their lack of realism focusing on even-aged monocultures and disregarding natural hazards and risk. Based on published studies, Knoke and Seifert (2008) evaluated the influence of the tree species mixture on forest stand resistance against natural hazards, productivity and timber quality using Monte Carlo simulations in mixed forests of Norway spruce and European beech. They assumed site conditions and risks typical of southern Germany and found superior financial returns of mixed stand variants, mainly due to significantly reduced risks.

The management of uneven-aged forests requires not only a basic understanding of the species-specific responses to shading and competition on different growing sites, but also more sophisticated methods of sustainable harvest planning. A selective harvest event in an uneven-aged forest, involving removal of a variety of tree sizes within each of several species, is much more difficult to quantify and prescribe than standard descriptors of harvest events used in rotation management systems, such as moderate high thinning or clearfelling.

Mortality, recruitment and growth, following a harvest event, are more difficult to estimate in an uneven-aged multi-species hardwood forest than in an even-aged pine plantation. The dynamics of a pine plantation, including survival and maximum density, is easier to estimate because its structure is much simpler than the structure of an uneven-aged multi-species forest. Thus, a better understanding of forest structure is a key to improved definition of harvest events and to the modeling of forest dynamics following that event. A forest ecosystem develops

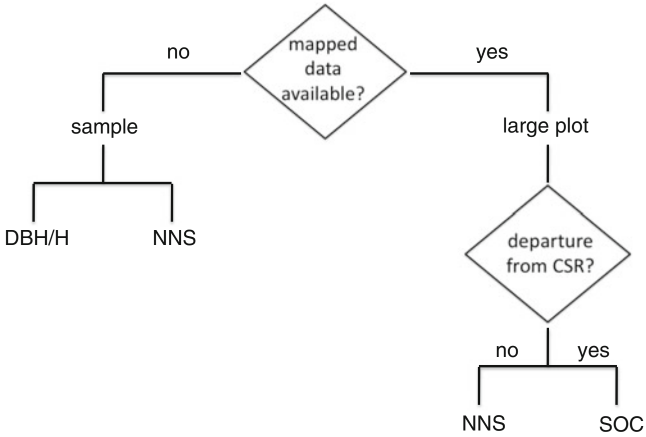


Fig. 2.1 Simplified decision tree indicating methods that may be used depending on available data. The abbreviation *DBH/H* refers to diameter distributions and diameter height relations which are frequently used to reveal structure. In addition to second order statistics (*SOCs*), nearest neighbor statistics (*NNS*) are particularly useful for structural analysis (cf. Pommerening 2008)

through a succession of harvest events. Each harvest event is succeeded by a specific ecosystem response following those disturbances. Improved understanding of forest structure may greatly facilitate estimation of the residual tree community following a particular harvest event, and the subsequent response of that community.

The objective of this contribution is to present methods that can be used to describe and analyse forest structure and diversity with particular reference to CCF methods of ecosystem management. Despite advances in remote sensing and other assessment technologies, mapped tree data are often not available, except in specially designed research plots. Forest inventories tend to provide tree data samples in small observation windows. Thus, in the overwhelming number of cases, the amount of data and their spatial range is too limited to use SOC methods (Fig. 2.1).

Often, tree locations are not measured, but the tree attributes, e.g. species, DBHs, diversity indices and other marks, are established directly in the field. This is a typical case for applying NNS. If, however, mapped data in large observation windows are available, SOCs are often preferred. Such point patterns may depart from the hypothesis of complete spatial randomness (CSR) and marks may be spatially independent or not. Cumulative characteristics such as the L function or the mark-weighted L functions can be used to test such hypotheses. If they are rejected, it makes sense to proceed with an analysis involving SOCs, such as pair correlation functions or mark variograms. If the hypothesis of mark independence is accepted, it suffices to use for example diameter distributions, non-spatial structural indices or NNS.

Because of their practical relevance in CCF, particular emphasis in this chapter will be on analytical tools that do not depend on mapped tree data. Foresters need to be able to analyse the changes in spatial diversity and structure following a harvest event. Their analysis must be based on data that are already available or that can be obtained at low cost.

We will first review non-spatial approaches and then present methods which will facilitate spatially explicit analysis using SOC and NNS, including examples of analysing harvest events and measuring structural differences between forest ecosystems. In some cases implementations of methods are shown using the *Comprehensive R Archive Network* (R Development Core Team 2011).

2 Non-spatial Structure and Diversity

A first impression about forest structure is provided by the frequency distributions of tree sizes and tree species. Accordingly, this section introduces methods that can be used for describing structure based on tree diameters and diameter/height relations. We will also present approaches to defining residual target structures in CCF systems.

2.1 Diameter Distributions

The breast height diameter of a forest tree is easy to measure and a frequently used variable for growth modeling, economic decision-making and silvicultural planning. Frequency distributions of tree diameters measured at breast height (DBH) are often available, providing a useful basis for an initial analysis of forest structure.

2.1.1 Unimodal Diameter Distributions for Even-Aged Forests

Unimodal diameter frequency distributions are often used to describe forest structure. The 2-parameter cumulative Weibull function is defined by the following equation:

$$F(x) = P\,(X \leq x) = \int_{-\infty}^{X} D(X)dX = 1 - e^{-\left(\frac{x}{scale}\right)^{shape}} \tag{2.1}$$

where F(x) is the probability that a randomly selected *DBH X* is smaller or equal to a specified *DBH x*. To obtain maximum likelihood estimates of the Weibull pdf, we can use the function *fitdistr* in package *MASS* of the *Comprehensive R Archive Network* (http://cran.r-project.org/). Using the vectors n.trees and dbh.class, the R code for the 2-parameter Weibull function would be:

```
n.trees<-c(1,4,2,6,6,7,6,4,2,2,0,1)
dbh.class<-c(32,34,36,38,40,42,44,46,48,50,52,54)          # rewriting the data for density function:
tree.d<-NULL; for(i in 1:length(dbh.class)){tree.d<-c(tree.d,rep(dbh.class[i],times=n.trees[i]))}
hist(tree.d,freq=FALSE,ylim=c(0,0.09))                          # plotting the histogram
library(MASS)                                                   # loading package MASS
w<-fitdistr(tree.d,"weibull")                                   # fitting weibull pdf parameters
lines(32:54,dweibull(32:54,w$e[1],scale=w$e[2]),col=2)
```

The Weibull function may be inverted which is useful for simulating a variety of forest structures characterised by the Weibull parameters:

$$F(x) = 1 - e^{-\left(\frac{x-a}{b}\right)^c} \quad \text{inverting gives } x = a + b \cdot [-\ln(1 - F(x))]^{\frac{1}{c}} \tag{2.2}$$

where $\mathrm{P}(X > x) = 1 - F(x)$ is the probability that a randomly selected DBH X is greater than a random number distributed in the interval [0, 1], assuming parameter values $a = 30$, $b = 13.7$ and $c = 2.6$. The following kind of question can be answered when using the inverted Weibull function: what is the *DBH* of a tree if 50% of the trees have a bigger *DBH*? Using our dataset, the answer would be: $x = 30 + 13.4 \cdot [-\ln(0.5)]^{\frac{1}{2.6}} = 41.6$ cm. Thus, we can simulate a diameter distribution by generating random numbers in the interval 0 and 1, and calculating the associated DBH's using Eq. 2.2.

2.1.2 Bimodal Diameter Distributions

There is a great variety of empirical forest structures that can be described using a theoretical diameter distribution. DBH frequencies may occur as bimodal distributions which represent more irregular forest structures. Examples are presented by Puumalainen (1996), Wenk (1996) and Condés (1997). Hessenmöller and Gadow (2001) found the bimodal DBH distribution useful for describing the diameter structures of beech forests, which are often characterised by two distinct subpopulations, fully developed canopy trees and suppressed but shade-tolerant understory trees. They used the general form $f(x) = g \cdot f_u(x) + (1 - g) \cdot f_o(x)$ where $f_u(x)$ and $f_o(x)$ refer to the functions for the suppressed and dominant trees, respectively and g is an additional parameter which links the two parts.

Figure 2.2 shows an example of the bimodal Weibull fitted to a beech forest near Göttingen in Germany. The distribution of the shade tolerant beech (*Fagus sylvatica*) typically shows two subpopulations of trees over 7 cm diameter DBH. In Fig. 2.2, the population of dominant canopy trees is represented by a rather wide range of diameters (18–46 cm), whereas the subpopulation of small (and often old) suppressed and trees has a much narrower range of DBHs. The two subpopulations are usually quite distinct. However, the proportions of trees that belong to either the suppressed or the dominant group may differ, depending mainly on the silvicultural treatment history (Fig. 2.2).

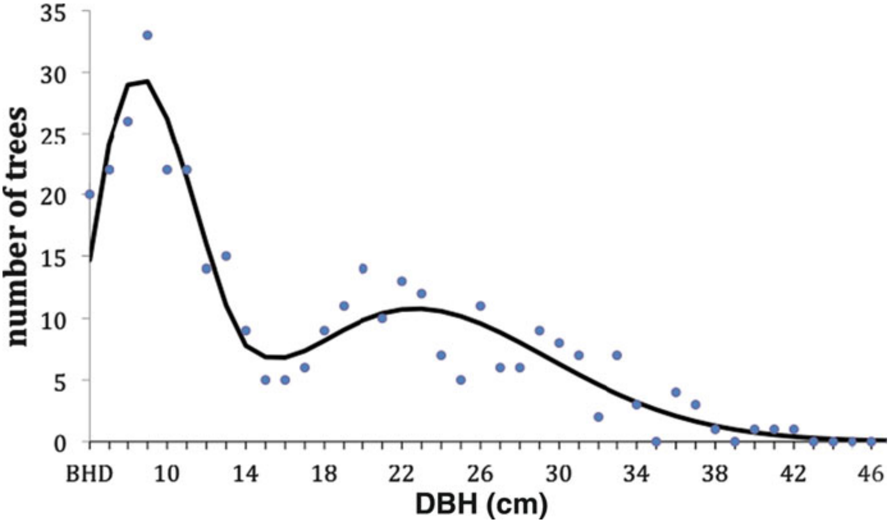


Fig. 2.2 Example of a bimodal Weibull distribution fitted to a beech forest near Göttingen in Germany

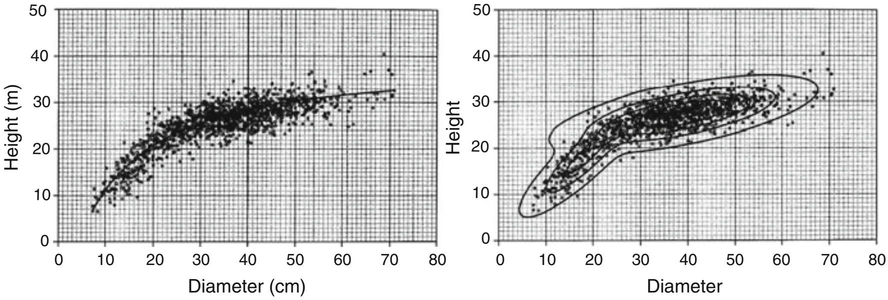


Fig. 2.3 Simple height-diameter relation (*left*) and contour plot of the fitted density function (After Zucchini et al. 2001)

The Weibull model may also be used to describe the diameter distributions in forests with two dominant species (Chung 1996; Liu et al. 2002), using a mix of species-specific *DBH* distributions. The diameter structure, however, is increasingly difficult to interpret as the number of tree species increases. This is one of the reasons why so much work has been done on the structure of monocultures. They are easy to describe using straightforward methods.

2.2 Diameter-Height Relations

One of the most important elements of forest structure is the relationship between tree diameters and heights. Information about size-class distributions of the trees

Table 2.1 Parameter estimates for Eq. 2.3 (After Temesgen and Gadow 2003)

	a				b	c	
Tree species	a_1	a_2	a_3	a_4	b	a_5	a_6
Aspen	20.655			0.08724	0.01509	1.399	
Cedar	17.947		−0.0009	0.14087	0.03497	1.304	
Paper birch	20.446		−0.0007	0.13355	0.03576	1.262	
Douglas-fir	32.037	−0.3504	−0.0007	0.18308	0.01797	1.093	0.00802
Larch	41.792				0.01709	1.118	0.00404
Lodgepole pine	20.852	0.3168	−0.0004	0.23962	0.03184	1.087	−0.0014
Ponderosa pine	32.208				0.01738	1.107	
Spruce	17.080	0.0932		0.34276	0.01073	1.462	

within a forest stand is important for estimating product yields. The size-class distribution influences the growth potential and hence the current and future economic value of a forest (Knoebel and Burkhart 1991). Height distributions may be quantified using a discrete frequency distribution or a density function, in the same way as a diameter distribution. However, despite greatly improved measurement technology, height measurements in the field are considerably more time consuming than DBH measurements. For this reason, generalised height-DBH relations are being developed, which permit height estimates for given tree diameters under varying forest conditions (Kramer and Akça 1995, p. 138).

2.2.1 Generalised DBH-Height Relations for Multi-species Forests in North America

Generalised DBH-height relations for uneven-aged multi-species forests in Interior British Columbia, Canada, were developed by Temesgen and Gadow (2003). The analysis was based on permanent research plots. Eight tree species were included in the study: Aspen (*Populus tremuloides* Michx.), Western Red Cedar (*Thuja plicata* Donn.), Paper Birch (*Betula papyrifera* March.), Douglas-Fir (*Pseudotsuga menziesii* (Mirb.) Franco), Larch (*Larix occidentalis* Nutt.), Lodgepole Pine (*Pinus contorta* Dougl.), Ponderosa Pine (*Pinus ponderosa* Laws.) and Spruce (*Picea engelmanii* Parry × *Picea glauca (Monench) Voss*). Five models for estimating tree height as a function of tree diameter and several plot attributes were evaluated. The following model was found to be the best one:

$$\hat{H} = 1.3 + a\left(1 - e^{b \cdot DBH^{c}}\right) \tag{2.3}$$

with $a = a_1 + a_2 \cdot BAL + a_3 \cdot N + a_4 \cdot G$ and $c = a_5 + a_6 \cdot BAL$. BAL is the basal area of the larger trees (m²/ha); G is the plot basal area (m²/ha); N is the number of trees per ha; $a_1 \ \ldots \ a_7$, b and c are species-specific coefficients listed in Table 2.1.

A generalised DBH-height relation such as the one described above, may considerably improve the accuracy of height estimates in multi-species natural forests. In a managed forest, however, the parameter estimates should be independent of *G* and *BAL* because density changes at each harvest event. This affects the parameter estimates, which is not logical.

2.2.2 Mixed DBH-Height Distributions

Unmanaged forests are used as a standard for comparison of different types of managed stands. There are numerous examples showing that virgin beech forests exhibit structures which include more than one layer of tree heights (Korpel 1992). In a managed beech forest, the vertical structure depends on the type of thinning that is applied. In a high thinning, which is generally practised in Germany, only bigger trees are removed while the smaller ones may survive for a very long time, resulting in a typical pattern with two subpopulations with different diameter-height relations. Thus, the population of trees is composed of a mixture of two subpopulations having different diameter-height distributions. Zucchini et al. (2001) presented a model for the diameter-height distribution that is specifically designed to describe such populations.

A mixture of two bivariate normal distributions was fitted to the diameter-height observations of 1,242 beech trees of a DBH greater than 7 cm in the protected forest Dreyberg, located in the Solling region in Lower Saxony, Germany. The dominant species is beech and the stand is very close to the potentially natural stage. All parameters have familiar interpretations. Let $f(d,h)$ denote the bivariate probability density function of diameter and height. The proposed model then is

$$f\,(d,h) = \alpha \cdot n_1\,(d,h) + (1-\alpha) \cdot n_1\,(d,h) \tag{2.4}$$

where α, a parameter in the interval (0, 1), determines the proportion of trees belonging to each of the two component bivariate normal distributions $n_1\,(d,h)$ and $n_2\,(d,h)$. The parameters of $n_j\,(d,h)$ are the expectations u_{dj}, u_{hj}; the variances σ^2_{dj} and σ^2_{hj}, and the correlation coefficients, $\rho_j, j = 1,2$. A simple height-diameter curve, as shown in Fig. 2.3 (left), describes how the mean height varies with diameter at breast height, but it does not quantify the complete distribution of heights for each diameter.

The larger subpopulation (approximately 80% of the entire population) comprises dominant trees in which the slope of the height-diameter regression is less steep than that for the smaller subpopulation (approximately 20% of the population). This can be seen more distinctly in Fig. 2.4 (right). Mixed *DBH*-height distributions may be fitted using the “flexmix” package (Leisch 2004) of the *Comprehensive R Archive Network.*

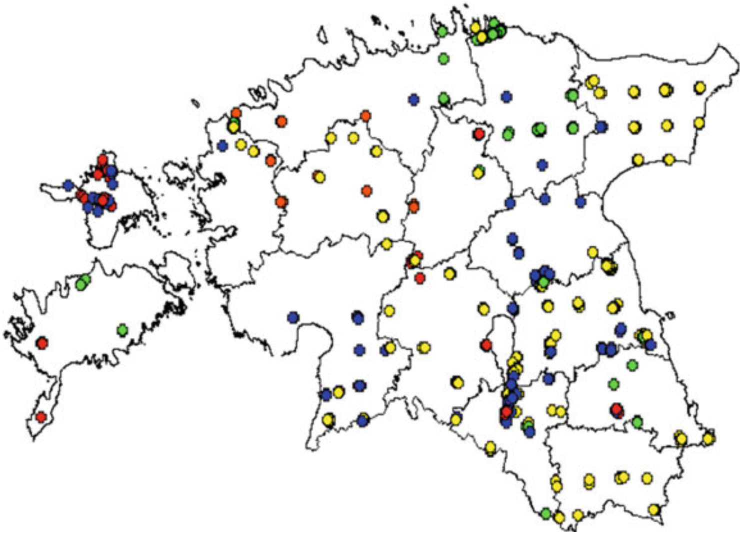


Fig. 2.4 Geographical distribution of permanent plots in the Estonian Permanent Forest Research Plot Network

2.2.3 Generalised DBH-Height Relations in a Spatial Context

Schmidt et al. (2011, in press) present an approach to modeling individual tree height-diameter relationships for Scots pine (*Pinus sylvestris*) in multi-size and mixed-species stands in Estonia. The dataset includes 22,347 trees from the Estonian Permanent Forest Research Plot Network (Kiviste et al. 2007). The distribution of the research plots is shown in Fig. 2.4.

The following model which is known in Germany as the "Petterson function" (Kramer and Akça 1995) and in Scandinavia as the "Näslund function" (Kangas and Maltamo 2002) was used:

$$h_{ijk} = \left(\frac{d_{ijk}}{\alpha + \beta \cdot d_{ijk}} \right)^{\gamma} + 1.3 \tag{2.5}$$

where h_{ijk} and d_{ijk} are the total height (m) and breast height diameter (cm) of the *k*th tree on the *i*th plot at the *j*th measurement occasion, respectively; α, β and γ are empirical parameters. Models are often linearized with the aim to apply a mixed model approach (Lappi 1997; Mehtätalo 2004; Kinnunen et al. 2007). In addition, the use of generalized additive models (*GAM*) requires specification of a linear combination of (nonlinear) predictor effects. The Näslund function can be linearized by setting the exponent γ constant (in our case $\gamma = 3$, see Kramer and Akça 1995):

$$y_{ijk} = \frac{d_{ijk}}{\left(h_{ijk} - 1.3\right)^{1/3}} = \alpha + \beta \cdot d_{ijk} \tag{2.6}$$

A two-level mixed model was fitted, with random effects on stand and measurement occasion levels. Thus, it was possible to quantify the between-plot variability as well as the measurement occasion variability. The coefficients of the diameter-height relation could be predicted not only using the quadratic mean diameter, but also the Estonian plot geographic coordinates x and y. The problem of spatially correlated random effects was solved by applying the specific methodology of Wood (2006) for 2-dimensional surface fitting. Thus, the main focus of this study was to use an approach which is spatially explicit allowing for high accuracy prediction from a minimum set of predictor variables. Model bias was small, despite the somewhat irregular distribution of experimental areas.

3 Analysing Unmarked and Marked Patterns

This section presents methods for analysing unmarked and marked patterns of forest structure and diversity. We show examples of measuring differences between patterns and of reconstructing forests from samples. The emphasis is on nearest neighbor statistics which can be easily integrated in CCF management because they do not require large mapped plots.

3.1 Unmarked Patterns

The two-dimensional arrangement of tree locations within an observation window may be seen as a realisation of a point process. In a point process, the location of each individual tree, i, can be understood as a point or event defined by Cartesian coordinates $\{x_i, y_i\}$. This section deals with unmarked point patterns, using methods for mapped and unmapped tree data. The required code of the *Comprehensive R Archive Network* is also presented to make it easier for potential users to apply the methods.

3.1.1 Mapped Tree Data Available

Second-order characteristics (SOCs) were developed within the theoretical framework of mathematical statistics and then applied in various fields of natural sciences including forestry (Illian et al. 2008; Møller and Waagepetersen 2007). They describe the variability and correlations in marked and non-marked point processes. In contrast to nearest neighbour statistics (NNS), SOCs depend on a distance variable r and quantify correlations between all pairs of points with a distance of approximately r between them. This allows them to be related to various ecological scales and also, to a certain degree, to account for long-range point interactions

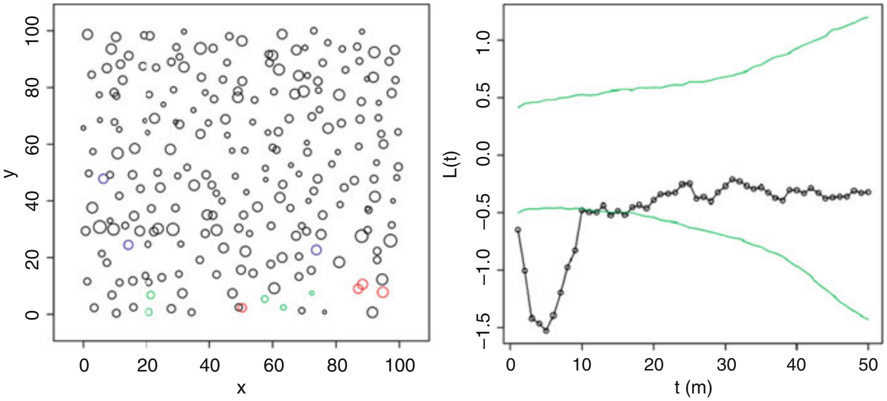
(Pommerening 2002). SOCs may be employed when mapped data from large observation windows are available. Examples are given in this section, but more can be found in the cited literature.

In the statistical analysis, it is often assumed that the underlying point process is *homogeneous* (or *stationary*) and *isotropic*, i.e. the corresponding probability distributions are invariant to translations and rotations (Diggle 2003; Illian et al. 2008). Although methods have also been developed for inhomogeneous point processes (see e.g. Møller and Waagepetersen 2007; Law et al. 2009), patterns are preferred in the analysis for which the stationarity and isotropy assumption holds. These simplify the approach and allow a focused analysis of interactions between trees by ruling out additional factors such as, for example, varying site conditions. In this context the choice of size and location of an observation window is crucial.

A rather popular second-order characteristic is Ripley's K-function (Ripley 1977). $\lambda K(r)$ denotes the mean number of points in a disc of radius r centred at the typical point i (which is not counted) of the point pattern where λ is the *intensity*, i.e. the mean density in the observation window. This function is a cumulative function. Besag (1977) suggested transforming the K-function by dividing it by π and by taking the square root of the quotient, which yields the L-function with both statistical and graphical advantages over the K-function. The L-function is often used for testing the complete spatial randomness (CSR) hypothesis, e.g. in Ripley's L-test (Stoyan and Stoyan 1994; Dixon 2002). For example, using the libraries *spatstat* and *spatial* of the *Comprehensive R Archive Network*, the following function *Lfn* (provided by ChunYu Zhang) calculates the L function and the 99% confidence limits for a random distribution.

```
library (spatstat)
library (spatial)
# Lfn calculates the L-function
Lfn=function(data,xu,yu,intervallength){    # intervallength=1 interval between distances, scales.
ppregion(xl=0, xu=xu, yl=0, yu=yu)           # Sets the rectangular domain (xl= min, xu= max) of plot
a=Kfn(data,min(xu,yu)/2,min(xu,yu)/2/intervallength) # K-Function in spatial package Ripley
Lfn=a$y-a$x; return(data.frame(t=a$x,Lfn=Lfn)) }      # L-Function;  t is the scale
#Critical values can be computed using Monte Carlo simulation.
# The following function Lenv calculates the envelope for L(t),
Lenv=function (data,xu,yu,intervallength){
x1=Sys.time(); ppregion(xl=0, xu=xu, yl=0, yu=yu); b=matrix(0, min(xu,yu)/2,1)
for (i in 1:10000) {a=Kfn(Psim(length(data$x)),min(xu,yu)/2,min(xu,yu)/2/intervallength)
 Lfn=a$y-a$x; Lfn=matrix(Lfn,,1); b=cbind(b,Lfn)}; low0.01=numeric(); up0.01=numeric()
 for (j in 1:dim(b)[1]){low0.01[j]=sort(b[j,])[51]; up0.01[j]=sort(b[j,])[9951] }
 x1=Sys.time()-x1; cat("run time:",x1, "\n")
return(data.frame(t=a$x,low0.01=low0.01,up0.01=up0.01))}
```

The above functions are implemented in a field experiment where the dominant species is beech (*Fagus sylvatica*). The plot Myklush Zavadivske covering 1 ha, is located near Lviv in Western Ukraine. The following R code generates the graphics presented in Fig. 2.5.



Mapped field plot Myklush Zavadivske

Corresponding *L(r),* and 99% confidence limits

Fig. 2.5 The beech trees in the almost pure beech plot Myklush Zavadivske, show significant regular distribution up to distances of 10 m (*right*)

```
attach(dat)                                              # dataset used
pts<-list(x=dat$x, y=dat$y)                              # all coordinates
Lfn1=Lfn(pts,60,100,1); Lenv1=Lenv(pts,60,100,1)    # Lenv1 takes about 2-4 minutes
plot(Lfn1$t,Lfn1$Lfn,ylim=c(min( range(Lenv1$ low0.01),range(Lenv1$
up0.01),range(Lfn1$Lfn)),max( range(Lenv1$
low0.01),range(Lenv1$ up0.01),range(Lfn1$Lfn))),xlab="t (m)",ylab="L(t)")
lines(Lfn1$t,Lfn1$Lfn); lines(Lenv1$low0.01,col="green"); lines(Lenv1$up0.01,col="green")
cbind(Lfn1,low0.01=Lenv1$low0.01,up0.01=Lenv1$up0.01)
```

The beech trees in the experimental plot "Myklush_Zavadivske", show significant regular distribution up to distances of 10 m, which can be seen by the values of the *L* function well below the lower bound envelope.

Cumulative functions are not always easy to interpret and for detailed structural analysis functions of the nature of derivations are preferred. One of these is the pair correlation function, $g(r)$, which is related to the first derivative of the *K* function according to the interpoint distance *r* (Eq. 2.7; see Illian et al. 2008):

$$g(r) = \frac{K'(r)}{2\pi r}. \tag{2.7}$$

For a heuristic definition consider $P(r)$ as the probability of one point of the point process being at *r* and another one at the origin *o*. Let *dF* denote the area of infinitesimally small circles around *o* and *r* and $\lambda\ dF$ the probability that there is a point of the point process in a circle of area *dF*. Then

$$P(r) = \lambda dF\,\lambda dF\;g(r). \tag{2.8}$$

In this approach, the pair correlation function $g(r)$ acts as a correction factor. For Poisson processes and for large distances when the point distributions are

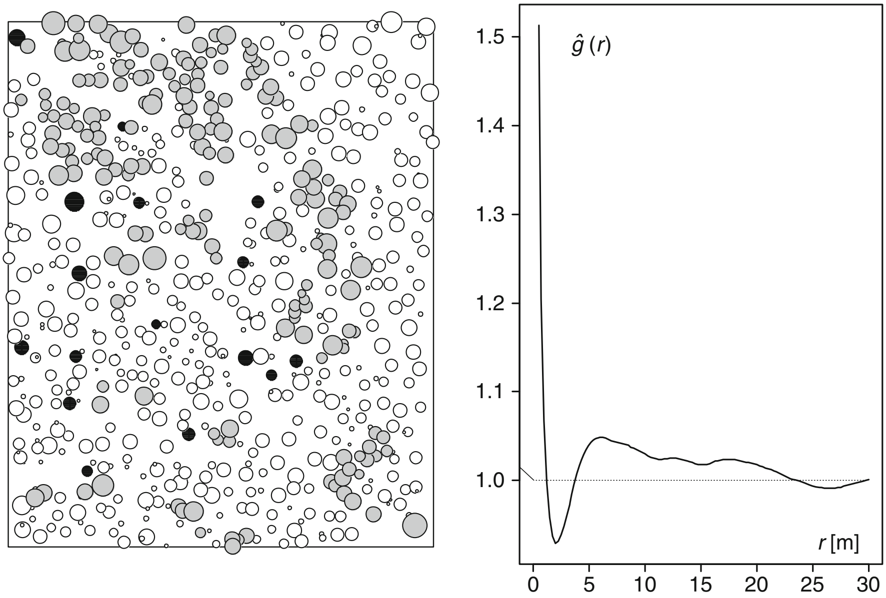


Fig. 2.6 *Left*: Mixed beech (*white*) – Norway spruce (*gray*) – oak (*black*) forest Walsdorf. The observation window is 101 × 125 m. *Right*: The pair correlation function, $\hat{g}(r)$, for all tree locations regardless of species (r is the intertree distance)

stochastically independent $g(r) = 1$. In the case of attraction of points $g(r) > 1$ (leading to a cluster processes) and for inhibition between points $g(r) < 1$ (resulting in regular patterns).

Figure 2.6 shows an example from the Walsdorf forest in Germany (provided by Arne Pommerening) with beech, Norway spruce and oak. The pair correlation function, $\hat{g}\ (r)$, indicates strong clustering of trees at short distances up to approximately 1.5 m. This is followed by a deficit of pairs of trees occurring at distances between 1.5 and 4 m. The most frequent larger intertree distance is 5 m. There are spatial correlations between trees up to 30 m (correlation range). The partial pair correlation function $g_{ij}\,(r)$, which by analogy to $K_{ij}\,(r)$ (Lotwick and Silverman 1982) can also be referred to as the intertype pair correlation function, is a tool for investigating multivariate point patterns. Its interpretation is similar to that of $g(r)$. We will not present examples here, mainly because the focus of this section is on unmarked processes. The reader may refer to Penttinen et al. (1992), Stoyan and Penttinen (2000), Illian et al. (2008), Kint et al. (2004) and Suzuki et al. (2008).

3.1.2 Mapped Tree Data Not Available

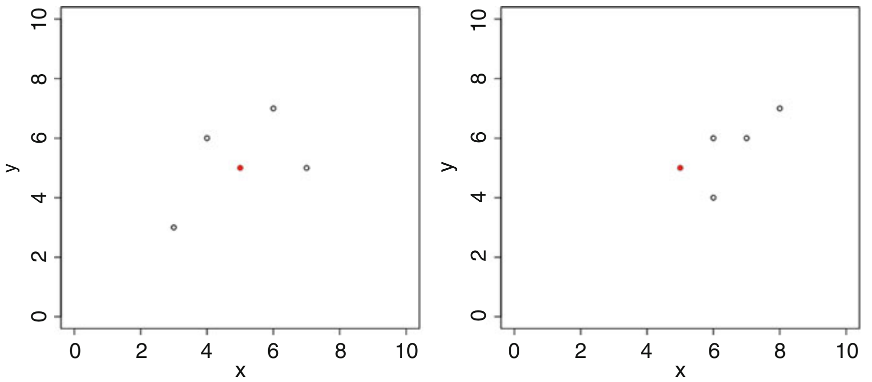
Staupendahl and Zucchini (2006) present a brief review of a variety of forest structure indices which were specially developed or adapted for forestry use (cf. Upton and Fingleton 1989, 1990; Biber 1997; Gleichmar and Gerold 1998; Smaltschinski 1998; Gadow 1999). For practical forestry purposes, where complete mappings of populations are normally not feasible, the distance methods are particularly useful. These methods, which are also used for density estimation, are based on nearest-neighbour distances. The measurements made are of two basic types: distances from a sample point to a tree or from a tree to a tree. Again from these methods the *T* square sampling (Besag and Glaves 1973) and their modifications (Hines and Hines 1979) are particularly useful (Diggle et al. 1976; Byth and Ripley 1980), especially regarding the test of complete spatial randomness.

Assunção (1994) describes the spatial characteristics of a population of forest trees based on the angle between the vectors joining a particular sample point to its two nearest neighbouring trees. Unaware of Assunção's work, but with the same intention, Gadow et al. (1998) also evaluated the angles between neighbouring trees. They did not use a reference point but a reference tree, thus allowing cohort-specific structural analysis. Cohort-specific structure refers to the specific structure in the vicinity of a cohort of reference trees, for example the structure in the vicinity of a given species, or in the vicinity of very large trees. They also considered more than two neighbors, thus extending the neighborhood range. Finally, they proposed that the angles should not be measured exactly, which would be time consuming during field assessment, but classified. They defined a standard angle, for example 72° in the case of four neighbors (Hui and Gadow 2002). The number of observed angles between two neighboring vectors which are smaller than the standard angle, are added up and divided by the total number of angles, as follows:

$$W_i = \frac{1}{n} \sum_{j=1}^{n} v_{jk} \quad \text{with} \quad v_{jk} = \begin{cases} 1, \ \alpha_{jk} < \alpha_0 \\ 0, \ \text{otherwise} \end{cases} \quad \text{and } 0 \leq W_i \leq 1 \tag{2.9}$$

With four neighbors, W_i can assume five values: 0.0; 0.25; 0.5; 0.75 and 1.0. The following R-function calculates the uniform angle index:

```
# input: x0;y0 (ref. tree coordinates), st_angle (standard angle)
# n=number of neighbors; x, y (vectors of neighbor coordinates)
uai<-function (x0,y0,x,y, n, st_angle)
{
  x1<-x-x0; y1<-y-y0                                      # array of vectors for all neighbors
  c1<-sqrt(y1*y1 + x1*x1)                                 # calc the lengths of the bevel edges
  phi<-acos(y1/c1)* sign(x1) * 180/pi + 360               # calc the angles
  for (i in 1:n) if (phi[i]>=360) phi[i]<- phi[i]-360
  sa<- order(phi,x1,y1,x,y); sa<-rbind(phi,x1,y1,x,y)[,sa]  # sort neighbors
  phi<-sa[1,]; x1<-sa[2,]; y1<- sa[3,]; x<-sa[4,]; y<-sa[5,]
  count<-0; if ((phi[4]- phi[1])< st_angle) count<-count+1
  for (i in n:2) if ((phi[i]- phi[i-1])< st_angle) count<-count+1
  w<-count/length(x); return(w)
} # end of function uai
```



Neighbor coordinates: *x*<-c(7,4,6,3);*y*<-c(5,6,7,3)   Neighbor coordinates: *x*<-c(6,6,7,8); *y*<-c(6,4,6,7)

Fig. 2.7 Structural constellations for two reference trees located in the centre of the *squares*

The constellations for two reference trees, located in the centre of the square, are shown in Fig. 2.7. The following R code generates the plots for the two groups and calculates the uniform angle indices, using the function *uai*:

```
x0=5; y0=5; x<-c(6,6,7,8); y<-c(6,4,6,7); stangle<-72   # stangle is the standard angle
plot(x,y, xlim=c(0,10),ylim=c(0,10))                    # plot them
points(x0, y0, pch=19, cex=1, col = "dark red")         # pch=19: solid circle!!
w<-uai(x0,y0,x,y,stangle); w # call the function
```

Staupendahl and Zucchini (2006) studied an index considering the three trees nearest to a reference point. Figure 2.8 schematically illustrates the three types of angle-based approach. All three indices measure the angle αij which is the smaller one of the two possible angles formed by the vectors $\mathbf{v}_{ij}$ and $\mathbf{v}_{ij+1}$ joining the reference point (a, c) or reference tree (b) to tree j and tree $j+1$ respectively, with $j=1$ (a), $j=1\ldots4$ (b) and $j=1\ldots3$ (c). In (b) and (c) $\mathbf{v}_{ij}$ is sorted by azimuth into ascending order. To avoid edge effects sample points are chosen from the unshaded subregion with measurements allowed to trees also in the shaded buffer zone.

A structure unit thus consists of a reference tree, or a reference point, and the n nearest neighbors in the vicinity of the tree or point. A reference tree-based structure unit will reveal spatial patterns in the vicinity of a particular tree cohort (species or size class), which allows more meaningful interpretations.

The treatment of edge trees can affect the estimation of structural indices since they can involve off-plot neighbours. Pommerening and Stoyan (2006) investigated in what circumstances edge-correction methods are necessary, and evaluated different approaches. They proposed a variable buffer zone around the edge of the plot. Only such trees were selected as reference trees, which were located further away from the plot edge than the distance to their n^{th} nearest neighbour.

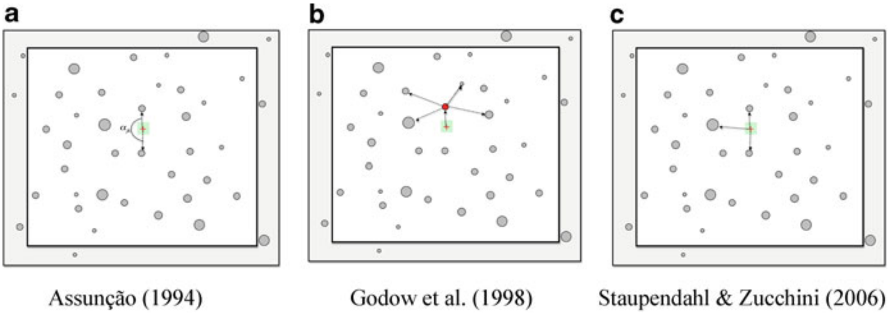


Fig. 2.8 Schematic representation of the sample unit of three types of angle-based measures of spatial pattern at sample point i within a hypothetical forest: (**a**) angle test using the two nearest trees to sample point, (**b**) uniform angle index W using the four nearest trees to a reference tree which is the nearest tree to the sample point and (**c**) index W_p using the three trees nearest to the sample point

The expected value of the uniform angle index is the average of an infinite number of realisations of W obtained by repeatedly randomizing orientations of the sample grid:

$$\mathrm{E}(\hat{W}) = \frac{1}{m} \cdot \sum_{r=1}^{m} \hat{W}_r \quad \text{for } m \to \infty \tag{2.10}$$

where $\hat{W}$ is the estimate of W in the rth simulation. Staupendahl and Zucchini (2006) used sampling simulations in three hypothetical forests with a regular, random and clustered pattern and sample sizes varying between 5 and 50, with 1,000 replications each. They show that the point-based criterion $\hat{W}_p$ is virtually unbiased for the entire range of sample sizes and for different spatial patterns. They conclude that the performance of the neighborhood-based method is comparable to alternative methods that are more costly to implement, and recommend a sample size of 20–30 sample points per compartment. Corral-Rivas et al. (2010) found that nearest neighbour-based indices enable a categorisation of a spatial pattern of forests with a sensitivity comparable to that of Ripley's $L(r)$ test, at finer scales. The assessment of the angles can be integrated into routine forest surveys, virtually without additional cost.

3.2 *Marked Patterns*

We believe that one of the basic requirements of sustainable CCF management is the ability to assess complex forest structure at affordable cost. Therefore, this section presents a brief introduction to methods that can be used to incorporate structural assessment in routine forest inventories. We then give examples of measuring tree size diversity. Finally, we deal with species diversity, again in a spatial context.

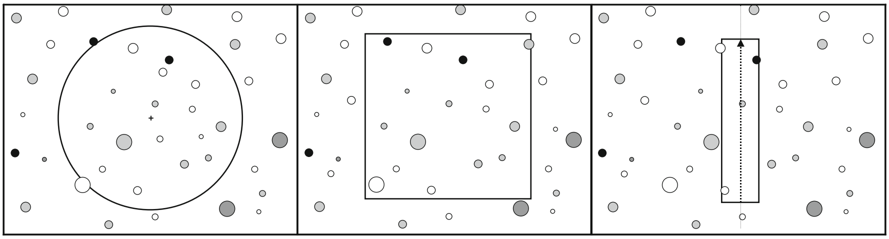


Fig. 2.9 *Circular*, *square* and *rectangular* sample plots for assessing area-based attributes, distributions and relationships between variables

3.2.1 Assessment

Traditionally, a forest ecosystem is characterised by area-based attributes (basal area, biomass, number of trees per hectare), mean values and distributions (diameters), and relationships (DBH-height regression). These classical attributes of a forest ecosystem are usually assessed in the field by sampling in field plots of specified shape and size (Kleinn et al. 2010). Fixed area plots of circular, square or rectangular shape (Fig. 2.9) and sample points using the angle count method are most common. For details refer to standard forest mensuration textbooks (e.g. Van Laar and Akça 2007).

Variables which characterise the spatial structure and diversity of an ecosystem are either included in a rather rudimentary way, or not considered at all, in standard forest sampling schemes. Forest spatial structure is characterised by nearest neighbor statistics (*NNS*) which include the variables *aggregation*, *species mingling* and *size differentiation*. *Aggregation* refers to the regularity of tree positions. High aggregation is often described as “clumped”. *Species mingling* defines the degree of spatial segregation of the tree species in a forest. Low species mingling means high segregation, no matter how many species occur in the forest. *Size differentiation* measures the degree by which trees of different sizes are spatially mingled. High size differentiation implies that trees of varying size occur in close vicinity of each other. These relationships are explained by three simple diagrams in Fig. 2.10.

Traditional forest sampling concentrates on assessment of forest density, volume and timber products. For characterising spatial structure and diversity in uneven-aged multi-species forest, we require information about the distributions of aggregation, species mingling and size differentiation. A convenient sampling scheme for assessing these variables is distance sampling, which is popular among ecologists despite the potential for bias (Krebs 1999; Nothdurft et al. 2010). A convenient sampling unit is the *n-tree-structure-unit* (NSU), which consists of a sample point, or a reference tree which is located closest to the sample point, and its *n* nearest neighbors (Fig. 2.11). The *n-tree-structure-unit* is not to be confused with the *n-tree-sample* where the plot radius is equal to the distance to the nth tree, as proposed by Prodan (1966). In spatial statistics the terms point-related and test-location-related summary characteristics are sometimes used (Pommerening

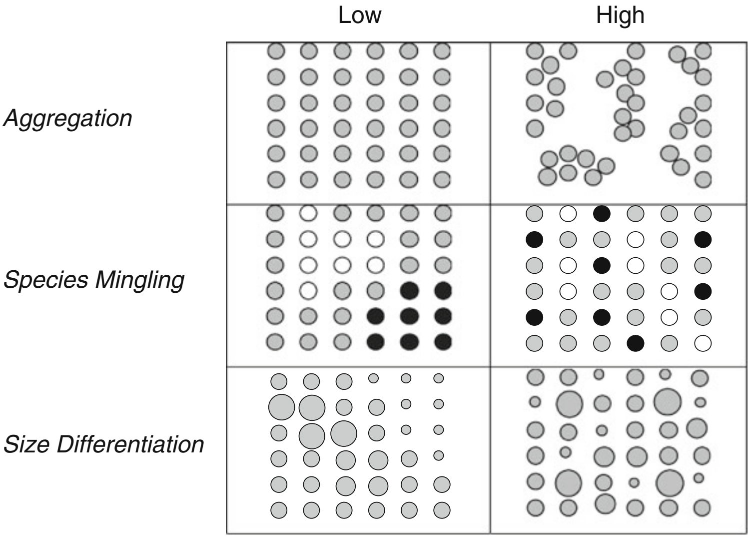


Fig. 2.10 Schematic representation of the three variables aggregation, species mingling and size differentiation which are used to describe forest structure and diversity

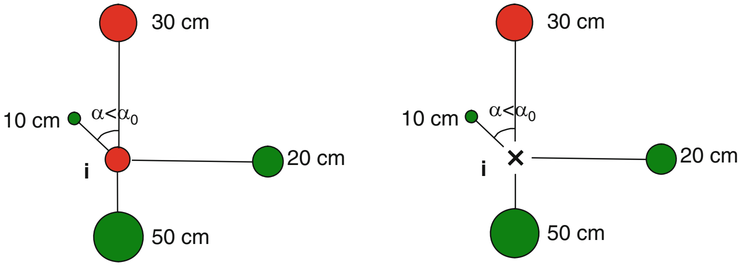


Fig. 2.11 Two neighborhood units for assessing the distributions of aggregation, species mingling and size differentiation in a forest. *Left*: tree-based unit, *right*: point-based unit. Numbers represent the tree DBHs

2008; Illian et al. 2008). In a circular plot sampling design where the azimuth and distance to the plot centre are known for each tree, the attributes of an *n-tree-structure-unit* can be calculated using the methods explained in other sections of this chapter. In sampling designs where tree coordinates are not known, the NSU can simply be integrated into the design. Measuring between tree distances is costly and unnecessary. Unbiased estimates of basal areas may be obtained using the angle count method. Thus, using the NSU, it will be possible to assess forest structure and diversity plus all the other traditional variables, without having to measure tree-to-tree distances or tree coordinates.

Examples of point-based structure variables are shown below:

Aggregation (W_P)	*Species Mingling* (M_P)	*Size Differentiation* (T_P)
Proportion of angles $\alpha_{i1}, \ldots, \alpha_{in}$ between the vectors from reference point i towards the *n* nearest neighbors (clockwise), which are smaller than some standard angle α_0.	Number of tree species of the *n* nearest neighbors as a proportion of the maximum possible number of species (*n*).	DBH coefficient of variation of the *n* nearest neighbors around a reference point *i*.

Examples of tree-based structure variables are shown below:

Aggregation (W_T)	*Species Mingling* (M_T)	*Dominance* (U_T)
Proportion of angles $\alpha_{i1}, \ldots, \alpha_{in}$ between the vectors from reference tree i towards the *n* nearest neighbors (clockwise), which are smaller than some standard angle α_0.	Proportion of species of the *n* nearest neighbors which do not belong to the species of the reference tree.	Proportion of *n* nearest neighbors with a DBH smaller than the DBH of the reference tree.

Figure 2.11 shows two neighborhood units, a tree-based one and a point-based one. The four nearest neighbours around the reference point, or the reference tree, are shown. Both units feature the same two species *A* (one or two trees) and *B* (three trees). Numbers represent the tree DBHs.

Only one angle α_{ik} is smaller than the standard angle α_0 (which happens to be 72°). The numbers in the diagrams represent each tree's DBH. The following structure variables may be calculated for a tree-based and a point-based unit respectively:

Tree-based unit	Point-based unit
$W_T = \frac{0+0+0+1}{4} = 0.25$	$W_P = \frac{0+0+0+1}{4} = 0.25$
$M_T = \frac{0+1+1+1}{4} = 0.75$	$M_P = \frac{2}{4} = 0.5$
$U_T = \frac{0+0+0+1}{4} = 0.25$	$T_P = \frac{\sqrt{\{(50-27.5)^2+(10-27.5)^2+(30-27.5)^2+(20-27.5)^2\}/3}}{(50+10+30+20)/4} = 0.65$

Hui and Albert (2004) proposed a slightly modified sampling design. Their sampling unit consists of the four trees closest to a sample point. Each of these four trees is a reference tree. This gives four *n-tree-structure-units*, instead of just one. Thus, the structure of a forest is characterised by the distributions of the variables assessed in all the field sample points. An advantage of the tree-based approach is the fact that forest structure may be analysed cohort-specific (structure in the vicinity of trees of species *X* with a DBH greater than 50 cm, for example).

Motz et al. (2010) carried out a study to investigate how tree diversity measures may be estimated as extensions of existing forest resource inventories. They compared the precision of angle count and fixed radius plot sampling with respect to nine representative diversity indices in three different forest types at stand, enterprises and national forest level. Their results indicated that most of the spatially explicit indices are more precisely estimated by fixed radius plots. The superiority of fixed radius plot sampling to angle count sampling increased significantly with increasing diameter differentiation of forests. Basal areas were estimated by angle count sampling with at least the same precision as from fixed radius plots.

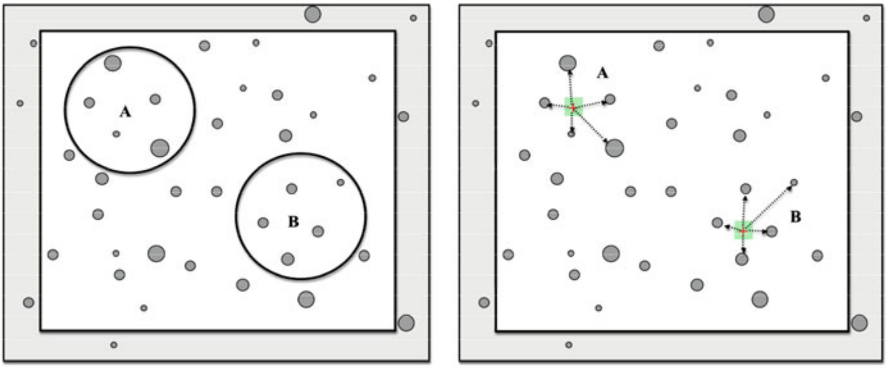
The design of a forest inventory system is an optimisation problem (Staupendahl and Gadow 2008; Kleinn et al. 2010). It should be possible to estimate the target variables with high accuracy, subject to the constraint of limited resources. In an attempt to respond to the need for more flexible forest inventory designs, Gadow and Schmidt (1998) proposed that forest sampling should take place after the trees had been marked for a thinning and before they are harvested. Thus, a *harvest event assessment* delivers information about the products that will be removed, the structural changes caused by the harvest, and the condition of the forest remaining after the harvest event. The emphasis is on the timing of the inventory. Instead of assessing the entire resource at fixed time intervals, information is gathered where it is needed most. Because of the specific timing, *Harvest event assessment* is particularly well suited to management control in complex forest structures, such as found in CCF (Puumalainen 1998).

3.2.2 Tree Size Diversity

The examples in the previous sections have shown how the structure of a forest may be defined by the distribution of tree sizes and by the particular relationship between different size variables (DBH-height; DBH-crown width). Such distributions, however, do not reveal the particular spatial arrangement of these variables (Schütz 2002). It requires little imagination to realize that a variety of spatial patterns of tree diameters may be possible for any particular diameter frequency distribution. One might refer to a high degree of "size mingling" when large and small trees occur in close vicinity of each other, or to low size mingling when large and small trees are spatially segregated. Such spatial arrangements may be assessed in the field using sample plots (Fig. 2.13, left) or neighborhood groups of trees (Fig. 2.12, right).

There are two main categories of edge-correction: *plus-sampling* and *minus-sampling* (Illian et al. 2008). Plus-sampling makes full use of all data within the observation window either by recording objects outside the observation window or by simulating them. Minus-sampling (also referred to as border method) only makes use of an inner sub-set of the observation window. Pommerening (2008) summarised edge-correction methods according to these two categories.

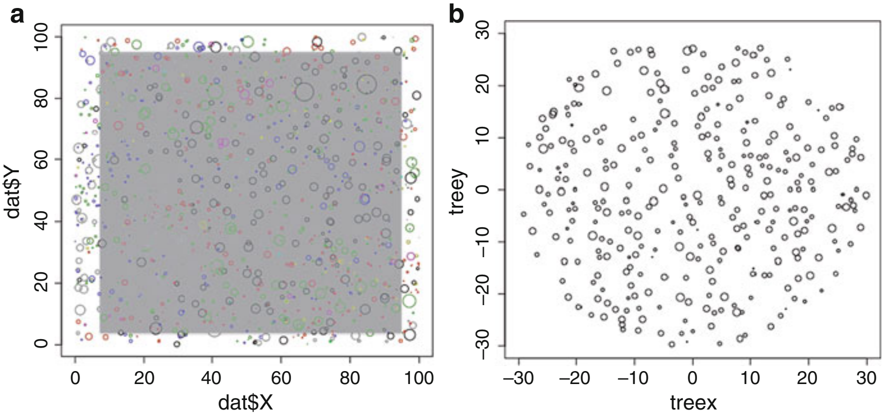
We define the trees in such a group as a "structure unit". Fixed-area sample plots are preferred for unbiased assessments of area-based variables, e.g. number of trees or basal area per ha. Neighborhood groups are useful if the target information should



Fixed-radius plot

Neighborhood group

Fig. 2.12 DBH mingling may be assessed in the field using sample plots or neighborhood groups of trees. In order to avoid edge effects, sample points are chosen from the unshaded subregion. References are allowed to trees which are located in the shaded buffer zone, this is known as plus-sampling. Another method to avoid edge effects, known as minus-sampling, considers only those trees as reference trees in the analysis which are located further away from the plot edge than the distance to their nth nearest neighbor (Pommerening and Stoyan 2006)



Square plot Jiaohe1

Circular plot 1080

Fig. 2.13 Spatial arrangement of tree diameters in the (**a**) square field plot Jiaohe1 and the (**b**) circular plot 1080

be independent of density, as in the case of spatial mingling of tree diameters. Each structure unit has certain attributes. The distribution of the DBH mingling values of the sampled structure units will reveal the particular spatial arrangement of tree sizes in a larger forest area. Within a given diameter distribution, greater homogeneity of structure units reveals spatial segregation: trees of the same size are found in close proximity to each other.

Table 2.2 Four heterogeneity indices (H', CV_{DBH}, *Gini*, *Skew*) and two neighborhood indices (T_i, U_i) which may be used for describing DBH mingling within a structure unit (After Sterba and Zingg 2006)

Formula/symbol	Explanation	Reference
$H' = -\sum_{i=1}^{S} p_i \ln(p_i)$	$S =$ number of DBH classes; $p_i = n_i / N$ with $n_i =$ number of trees in DBH class i; $N =$ all trees	Shannon and Weaver (1949)
$CV_{DBH} = \frac{SDD}{D_m}$	$CV_{DBH} =$ DBH coefficient of variation; $SDD =$ standard deviation of DBHs (cm); $D_m =$ mean DBH (cm)	Any textbook of statistics
$Gini = \frac{0.5 - L_A}{0.5}$	$L_A =$ area under the Lorenz curve (obtained by plotting the cumulated basal areas over the cumulated tree numbers)	Stöcker (2002)
$Skew = \frac{\sum (D - D_m)^3 / (n-1)}{SDD^3}$	$Skew =$ skewness of DBH distribution; $SDD =$ standard deviation of DBHs (cm); $D_m =$ mean DBH (cm)	Sterba and Zingg (2006)
$T_i = 1 - \frac{1}{n} \cdot \sum_{i=1}^{n} \frac{\min(D_i, D_j)}{\max(D_i, D_j)}$	$T_i =$ DBH differentiation; n = number of neighbors of reference tree i; D_i, $D_j =$ DBH (cm) of reference tree i and neighbor j	Füldner (1995)
U_i	$U_i =$ dominance, the proportion of neighboring trees which are smaller than the reference tree in a structure unit	Hui et al. (1998)

Examples of a segregation of tree sizes may be found in a forest with dense groups of saplings within gaps adjoining areas of mature trees. On the other hand, a high degree of spatial mingling may be found in a forest of shade tolerant species where all tree sizes occur in close proximity to each other. Both forests may have the same diameter distribution, but completely different diameter mingling patterns. DBH mingling within a structure unit may be described using a variety of indices (Kint et al. 2003). Six examples of indices calculated for neighborhood groups are listed in Table 2.2. The distribution of each index may facilitate the analysis of the structural composition of the entire forest and the reader is referred to reviews by Staudhammer and LeMay and McElhinny et al. (2005).

Sterba and Zingg (2006) found close correlations and specific relationships between the four heterogeneity indices (H', CV_{DBH}, *Gini*, *Skew*) and the two neighborhood indices (T_i, U_i), based on data from a great variety of forest types, including coppice forests of different age, even-aged and uneven-aged forests. The four heterogeneity indices can be used to evaluate the evenness of tree sizes within a particular structure unit (a sample plot or a neighborhood group). The two neighborhood indices can be used to evaluate the structural attributes of a cohort, i.e. a specific subpopulation of trees that are similar in some way.

Hyytiäinen and Haigth (2011) employ a specific form of the Shannon index (S) to evaluate habitat quality. They describe the simultaneous species and size richness of a forest as follows:

$$S = -w_{sp}\sum_{u=1}^{y}\frac{B_u}{B}\ln\left(\frac{B_u}{B}\right) - w_{size}\sum_{v=1}^{z}\frac{B_v}{B}\ln\left(\frac{B_v}{B}\right) \qquad (2.11)$$

The weights of species and size diversity are denoted by w_{sp} and w_{size}, respectively. B is the total basal area, B_u is the total basal area of trees which belong to species u, and B_v is the basal area of trees in diameter class v. The number of species is y, the number of diameter classes is z. The following R code groups the basal areas of all trees in the plot *Ulaschkiwski29(6)* which had been measured in 2009 in the Carpatian mountains, Western Ukraine. The dataframe is Korol_dat and the basal areas are *Korol_dat$B:*

```
BA <- hist(Korol_dat$B)                                  # group basal areas
BA$mids # 250  750 1250 1750 2250 2750 3250              # BA-class midpoints
BA$counts # 23 32 26 28 18 9  2                          # BA-class frequencies
```

The following code calculates the total basal area of a particular species, for example for all pine trees:

```
Pine<-subset(Korol_dat, Species=="Kiefer"); PineBA<-sum(Pine$B)       # = 109522.6
```

The same is done for all the other three species (oak=48483.41, beech=17110.87 and spruce = 5255.89). The total basal area is 180372.7 cm^2. The first and 2nd term of Eq. 11 are calculated as follows, assuming weights of 0.6 and 0.4 respectively:

```
TB<-sum(B)                                                    # total basal area
BuB<-c(109522.6/TB, 5255.9/TB, 17110.9/TB, 48483.4/TB)        # species relative basal areas
k<-0; for (i in 1:4) {k=k+BuB[i]*log(BuB[i])}; sp<--0.6*k     # the first term of Eq. 11 gives 0.59
Sizes<-(BA$mids*BA$counts)/TB                                 # basal area relative frequencies
k<-0; for (i in 1:7) {k=k+Sizes[i]* log(Sizes[i])}; size<--0.4*k  # the 2nd term of Eq. 11 gives 0.71
```

There is no straight-forward approach to constraining the calculations such that (a) S assumes values between 0 and 1 and (b) S allows a comparison between two arbitrary ecosystems. The forest with the maximum number of species and the greatest diameter variances is not known. However, a practical solution might be to use as reference a fixed number of basal area classes (for example 7, as in our example) and a representative number of species occurring within a region. A negative characteristic of the Shannon-Weaver index is the fact that the value increases with increasing evenness of the relative frequencies. However, rare species are often considered to contribute more to diversity than common species (see Hui et al. 2011).

3.2.3 Cohort-Specific Structure

Nearest neighbor statistics are particularly useful in the study of cohort-specific structural attributes. We are using the data of two experimental field plots to illustrate the analysis of the diversity of tree sizes in the neighborhood of all trees which belong to a specific cohort of reference trees:

1. Plot Jiaohe1 is located in the Jiaohe forest in Jilin Province of North-Eastern China. The forest is managed by selective harvesting. The plot area is one hectare. The DBHs range from 0.4 to 79.2 cm. The following tree species occur in the plot: *Betula costata, Carpinus cordata*, *Fraxinus mandshurica*, *Acer mandshurica*, *Tilia amurensis*, *Acer mono*, *Syringa reticulata var. mandshurica*, *Juglans mandshurica*, *Abies holophylla*, *Pinus koraiensis*, *Ulmus laciniata*, *Juglans mandshurica*, *Phellodendron amurense and Ramus davurica.* The spatial arrangement of species and tree sizes, and the 5 m buffer zone outside the shaded area, is shown in Fig. 2.13a.
2. Plot 1080 is located in the south-central part of Estonia. The plot shape is circular, the plot radius being 30 m. Three tree species occur in the plot: *Pinus sylvestris*, *Betula pendula and Picea abies.* The spatial arrangement of species and tree sizes is shown in Fig. 2.13b. Plot 1080 has no fixed buffer zone. Instead, only those trees are considered in the analysis, which are located further away from the plot perimeter than the distance to their third nearest neighbor (see Pommerening and Stoyan 2006).

The following R code was used to identify the neighbor indices of a specific cohort of trees (*tree0*) inside the buffer zone (*inside*) using the *nnwhich()* function of the *spatstat* library. The dataframe *dat* refers to all trees including those located in the buffer zone; dat$X and dat$Y are their coordinates:

```
tree0<-subset(inside, inside$d_cm>1) # specify all reference trees for a dbh>1cm
nb<-4                                # number of nearest neighbors
width<-100; leng<-100                # size of plot (m)
library(spatstat)                    # specifics may change
allxy<-ppp(dat$X,dat$Y, c(0,width), c(0,leng)) # create planar point pattern dataset
m4<-nnwhich(allxy, k=1:nb)           # all nb nearest neighbors
ref<-m4[tree0$Tree,]                 # neighbor indices of cohort
```

The diameter coefficient of variation (cvd) of the four nearest neighbors increases with decreasing, as well as increasing, size of reference tree (Fig. 2.14).

The lowest cvd values in the neighborhood group are found between DBHs of 10 and 25 cm of the reference tree. It is important to note that this particular analysis does not require mapped datasets. The important advantage of using neighborhood groups is the fact that data can be assessed in routine forest surveys at practically no additional cost. More results of the analysis of the two plots are presented at the end of this section.

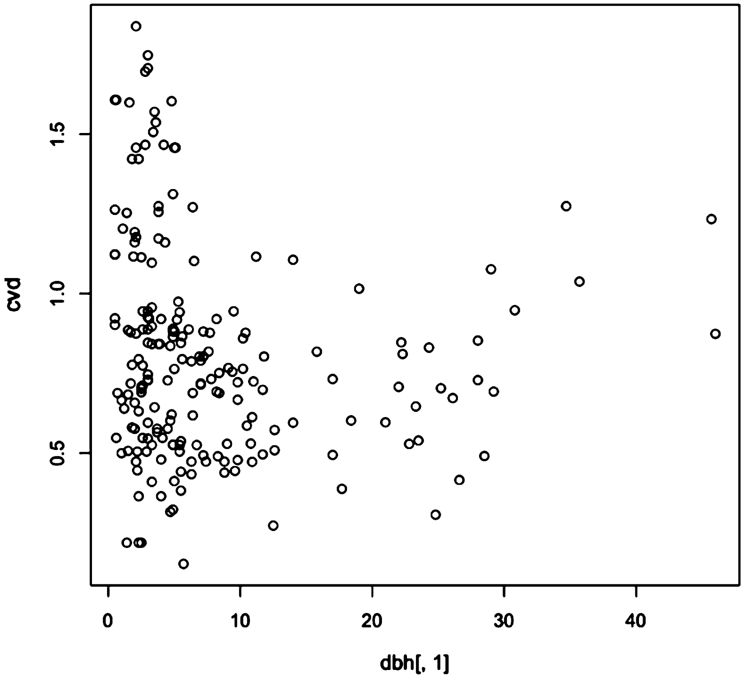


Fig. 2.14 Relationship between the dbh of the reference tree and the diameter coefficient of variation in the respective structure unit in the field plot Jiaohe1, for all trees of the species *Acer mono*

3.2.4 Tree Species Diversity

The main structural feature of a forest which only includes one single species, is the distribution and spatial mingling of tree sizes. A multi-species forest is additionally characterized by tree species richness and spatial mingling of tree species. A direct consequence of the large-scale forest destruction, especially since the second half of the twentieth century, is a serious depletion of tree species diversity. Many species have already become extinct or are threatened by extinction. Realising these threats, scientists have been increasing their activities in the area of biodiversity research.

Species Richness

Scientists began to use measures of biodiversity during the early years of the twentieth century. Among the first to propose indices of diversity were Fisher et al. (1943) and Simpson (1949). The Shannon-Weaver index was first applied in studies on community species diversity by Margalef (1957). Whittaker (1972) divided the diversity indices into four spatial scales, the α, β, γ and δ diversity. The literature on biodiversity seems endless and new ideas continue to emerge. Ganeshaiah and Shaanker (2000, 2003) for example, developed the *Avalanche* index which is based on taxonomic differences between species. Scholes and Biggs (2005) proposed a biodiversity *intactness* index which measures the decline of populations relative to their presumed pre-modern levels. Recent reviews of biodiversity indices include those by Ferris and Humphrey (1999) and Spanos and Feest (2007). One of the results of this research is the establishment of relationships between the size of sample areas and the number of species.

As expected, the number of tree species increases with increasing assessment area. Hubbell (2001) has shown empirical and theoretical relationships between area size and species number. His theoretical three-phase-curve of species diversity shows that at the local level the number of species increases rapidly with increasing area. At the regional level, the cumulative increase in the number of species is not influenced so much by the relative species frequency, but more by the balance between species formation, spatial distribution and extinction. The continental and intercontinental biogeographic scale produces spatially segregated evolutionary developments. The consequence is another accelerated increase in the species number with increasing area.

The increase of the number of species with increasing sample area is known as the species–area relationship. Lawton (1999) has referred to this relationship as one of the few fundamental laws in ecology. The difference in micro-site conditions generally increases with area and accordingly, the variety of species that can be supported generally increases with area (Williams 1964; Barkman 1989). The species–area relationship is more suitable for assessment of diversity than the mere number of species (Lepě and Stursa 1989). Several models have been proposed to describe this relationship (Monod 1950; de Caprariis et al. 1976; Gitay et al. 1991; Buys et al. 1994; Williams 1995; Tjørve 2003).

These models allow us to determine the *minimum species areal*, i.e. the smallest area which is required to capture all the species present within a given contiguous region. Using relatively large contiguous sample areas with known tree positions, assessed in different climatic regions with varying tree species abundances, Gadow and Hui (2007) could establish a specific relationship between the maximum number of tree species within a forest (S_{max}) and the minimum area required to capture all of them. This minimum species areal (A_{min}) can be directly estimated using a power function (Fig. 2.15). The analysis has shown that, for contiguous forest areas, the form of the species–area relationship is directly defined by the species abundance, the maximum number of tree species. This result confirms assumptions made by Preston (1962), May (1975) and Hubbell (2001).

Near-natural forest management systems with selective harvesting require undisturbed reference areas which are sufficiently large to include the essential features of an unmanaged forest. One of the key features of an unmanaged forest is the tree species distribution. Thus, A_{min}, the smallest area which represents the species richness of the entire population, is of great practical relevance. The minimum tree species areal may provide an important scientific basis for types of selective forest management which attempt to mimic natural processes of forest dynamics.

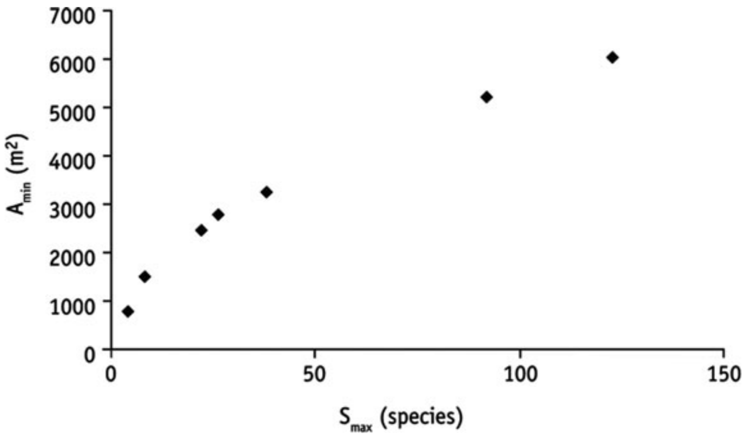


Fig. 2.15 Relationship between the minimum areal A_{min} and the estimated maximum number of tree species S_{max}. Gadow and Hui (2007) expressed the relationship by $A_{\min} = 487.8 \cdot S_{\max}^{0.524}$

Furthermore, if it is possible to estimate the minimum species area, then by analogy Wehenkel et al. (2011) propose to estimate the balanced structure area (BSA), which has been defined by Koop (1981) as the minimum contiguous area that includes all tree developmental stages. In the study presented by Wehenkel et al. (2011), the BSA is the minimum area required for sustainable management in a Mexican multi-sized selection forest. Their analysis has shown that a multi-sized forest represents a balanced structural unit if a specific relationship between harvest and growth can be maintained, using a defined target diameter distribution and disregarding major natural disturbances. Thus, using the BSA as an indicator of demographic sustainability, a range of goods and services may be consistently produced over time, thus realising the vision proposed by Nyland (2002).

Jenssen and Hofmann (2002) have shown relationships between different successional stages of a beech ecosystem and the plant species richness. Using plots of the same size, the average number of species in that particular investigation was increasing from the dense sapling stage to a stage dominated by mature trees. The highest number of plant species was observed in the senescent stage of the trees where the small-scale change between gaps and shaded areas produced a variety of environmental niches. The species diversity of a managed forest is thus greatly influenced by the type of silviculture.

Species Spatial Mingling

A useful measure of tree species diversity not only reflects the species richness of a community, but also the particular spatial structure. An important advantage of any index of forest diversity would be easy integration in routine management surveys to facilitate its application in land-use planning and monitoring. To develop more effective variables which reflect not only species richness and evenness, but also include a spatially relevant structural component, is an important challenge for science (Pretzsch 2003; Xia 2007).

The literature on biodiversity is extensive and new ideas continue to emerge (for an exhaustive review of biodiversity indices refer to Ferris and Humphrey (1999) and Spanos and Feest (2007)). Statistical methods for analysing beta diversity include principle component analysis and redundancy analysis (Legendre and Gallagher 2001). Some studies presented sophisticated and advanced evaluations of beta diversity, even in mega-diverse tropical forests (e.g. Condit et al. 2002). Many of these approaches require tree coordinates which are normally not available, thus presenting a major limitation regarding practical use. If costly assessment of tree coordinates is not required for diversity assessment, spatially relevant biodiversity analyses could be possible, based on data from routine forest surveys.

Hui and Albert (2004) defined the spatial relationship between a particular reference tree and its *n* nearest neighboring trees as the forest spatial structure unit. Theoretically, *n* could be any reasonable number. However, based on a series of field studies, they found that the optimum group size of such a spatial structure unit consisted of a reference tree and its four nearest neighboring trees. Pommerening (2006) found that the optimum number of nearest neighbours is that which allows the best spatial reconstruction of a given forest from sampled NNS (see details below).

The species spatial mingling within a structure unit is equal to the proportion of neighbors which do not belong to the same species as the reference tree. The previous work only considered the spatial mingling of the reference tree by calculating the proportion of neighbors which do not belong to the same species as the reference tree. The spatial mingling index, which was described by Gadow (1993) and used by Füldner (1995) and Pommerening (2002), is defined as follows:

$$M_i = \frac{1}{n} \sum_{j=1}^{n} v_{ij} \tag{2.12}$$

where n is the number of nearest neighbors considered, $v_{ij} = 1$ if the *j*th neighboring tree is not of the same species as the *i-th* reference tree and $v_{ij} = 0$ otherwise. The distribution of the M_i values, in conjunction with the species proportions within a given tree population, allows a detailed study of the spatial diversity within a forest. However, the number of different tree species in the structure unit was not taken into account, and this was a shortcoming of the original mingling index.

A logical improvement of the mingling index would be to consider in Eq. 3.6 not only the spatial mingling, but also the number of tree species. This can be achieved by multiplying M_i with $S_i / n_{\max}$ where S_i is the number of tree species in the neighborhood of reference tree *i*, including tree *i*, and $n_{\max}$ is the maximum number of species in this structure unit *i*. In our special case $n_{\max} = 5$, which was found to represent a good compromise between scientific assertion and practicability. The

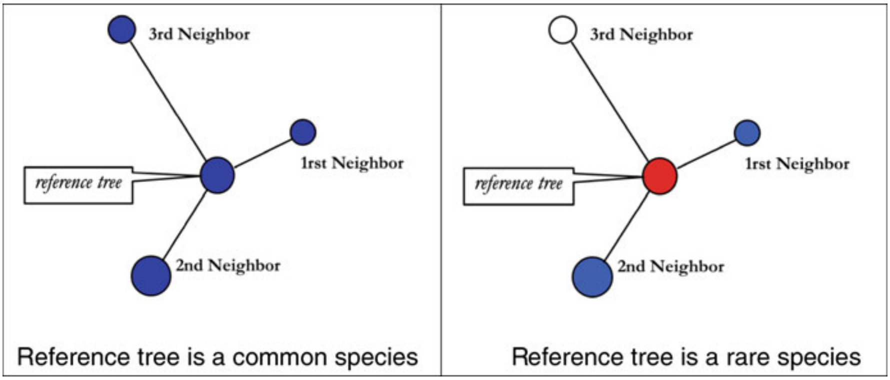


Fig. 2.16 Reference tree of a common species (*left*) and a rare species (*right*). A rare species is likely to produce high MS_i values

reasons for $n_{max} = 5$ were presented in several previous studies which evaluated different numbers of neighbors under field conditions (Gadow and Hui 2002; Hui and Albert 2004; Hui et al. 2007). A greater number of neighbors allows a more detailed and differentiated analysis. The smaller the number of neighbors, the lower is the cost of assessment. The most convenient number of neighbors from the practical point of view would be 1, because the one nearest neighbor is very easy to identify in the field. The assessment effort increases with an increasing number of neighbors. The cost of identifying a fifth neigbour increases sharply (because of the need to evaluate many tree-to-tree distances) and the cited field studies have shown that four neighbors ($n_{max} = 5$) represents the best compromise.

The spatial diversity status (MS_i) of a particular tree species is determined by the relative species richness within the structure unit *i* and the degree of mingling of the reference tree, and may be expressed as follows:

$$MS_i = \frac{S_i}{n_{\max}} \cdot M_i \tag{2.13}$$

where the M_i are the species mingling values, as defined above (refer to previous work, e.g. Füldner 1995; Pommerening 2002) and $n_{\max}$ is the maximum number of species in the structure unit. Equation 2.13 thus measures the tree species richness as well as an important spatial characteristic of a structure unit. A reference tree of a common species is more likely to have neighbors of the same species, which is reflected by low MS_i values (Fig. 2.16, left). On the other hand, a rare species is likely to produce a high proportion of high MS_i values (Fig. 2.16, right). Thus, MS_i is especially sensitive to rare tree species. Again, an important practical advantage, considering the assessment effort, is the fact that it is not necessary to measure tree coordinates in the field.

A useful statistic reflecting the status of each individual tree species in the community is the species average spatial status (MS_{sp}), which is the average value of the MS_i for each tree species in the community:

$$MS_{sp} = \frac{1}{5N_{sp}} \sum_{i=1}^{N_{sp}} \left(M_i \cdot S_j\right) \tag{2.14}$$

where N_{sp} is the number of trees of species *sp* in the community. The commonly used diversity indices are represented by a single statistic which combines the species richness and evenness (the ratio of observed to maximum richness). Assuming additivity, the tree species spatial diversity of a tree population may be expressed as the sum of the average spatial diversity states of the different tree species. This sum is conveniently expressed by the *TSS* criterion proposed by Hui et al. (2011):

$$TSS = MS_{sp1} + MS_{sp2} + \cdots + MS_{spn} = \sum_{sp=1}^{s} \left[\frac{1}{n_{\max} \cdot N_{sp}} \sum_{i=1}^{N_{sp}} M_i \cdot S_i \right] \tag{2.15}$$

where *s* is the number of tree species. *TSS* assumes a maximum value of 1 when each species is represented by one tree, in which case all MS_{sp} are equal to 1. When there is only one species of *N* trees in the community, the species richness is a minimum, and the *TSS* value is zero. The *TSS* variable thus represents the sum of the average spatial mingling values of all tree species in the community. It is a measure, not only of tree species richness, but also of tree species spatial diversity within a given ecosystem. Table 2.3 presents the MS_{sp} values for the 21 tree species in *Jiaohe1* and the three species in plot 1008.

The average MS_{sp} value for the 17 tree species in Jiaohe1 is 0.66, and the *TSS* value is 11.22. The average MS_{sp} value for the 3 tree species in 1080 is 0.31. Due to its absolute dominance, *Pinus sylvestris* in plot 1080 has a very low MS_{sp} value. The TSS value in 1080 is only 0.92, which is the result of the comparatively small number of species. The average diameter coefficient of variation is 0.75 in Jiaohe1 and 0.33 in 1080. Both, the *TSS* value and the average diameter coefficient of variation indicate a high spatial diversity in Jiaohe1. The spatial diversity of tree species and sizes is much lower in 1080.

TSS is sensitive to rare species and to variations in community structure, including species spatial isolation and spatial mingling. For these reasons, the *TSS* criterion is more effective in measuring tree species diversity than the commonly used indices. It allows detailed interpretation of forest spatial diversity and of forest structural modifications following selective thinnings in CCF systems. A particular advantage of the *TSS* index is the fact that its assessment, which is based on neighborhood relations, can be easily integrated in routine forest management surveys at practically no additional cost.

Table 2.3 Average MS_{sp} values and diameter coefficients of variation for the different tree species in plots Jiaohe1 and 1080

Plot	Species	Number of trees	MS_{sp}	Average cvd
Jiaohe1	*Acer mandshurica*	214	0.54	0.84
	Acer mono	202	0.50	0.81
	Ulmus propinqua	123	0.55	0.76
	Juglans mandshurica	112	0.63	0.74
	Fraxinus mandshurica	110	0.60	0.76
	Ulmus laciniata	65	0.66	0.81
	Syringa reticulata	60	0.63	0.70
	Carpinus cordata	55	0.67	0.81
	Tilia amurensis	38	0.70	0.89
	Pinus koraiensis	35	0.63	0.90
	Abies holophylla	16	0.68	0.90
	Betula costata	15	0.71	0.66
	Tilia mandschurica	8	0.71	0.78
	Phellodendron amurense	6	0.70	0.60
	Acer tegmentosum	5	0.84	0.74
	Prunus padus	3	0.67	0.68
	Qercus mongolica	3	0.80	0.88
1080	*Pinus sylvestris*	244	0.02	0.26
	Betula pendula	7	0.46	0.28
	Picea abies	7	0.44	0.45

Expected Values of NNS

Expected values of spatial indices may be of interest to students involved in comparative analysis of forest structures. According to Lewandowski and Pommerening (1997) expected mingling (**E**M), can be calculated as

$$\mathrm{E}M = \sum_{i=1}^{s} \frac{p_i\,(p - p_i)}{p\,(p-1)} \tag{2.16}$$

where s is the number of species, p is the number of trees in the observation window and p_i refers to the number of trees of species i.

Expected mark diameter differentiation is not as straightforward as expected mark mingling. Using tree diameters, DBH, as example marks Pommerening (1997, p. 18) proposed to sort DBH in ascending order, i.e. $i < j \Rightarrow DBH_i \le DBH_j$. As a result, the index set J of a given forest is obtained. Then the auxiliary measure R is defined as

$$R_i = \begin{cases} 0 & \text{for } i = 1 \\ \sum_{j=1}^{i-1} DBH_j & \text{for } i = 2, \ldots, p \end{cases} \tag{2.17}$$

Jiaohe forest in north-eastern China (Photo: Klaus von Gadow)

The expected mark differentiation for tree DBH, **E***T*, may now be calculated as

$$\mathrm{E}T = 1 - \frac{2}{p\,(p-1)} \sum_{j=1}^{p} \frac{R_j}{DBH_j}. \tag{2.18}$$

Details about the derivation of Eqs. 2.17 and 2.18 can be found in Pommerening (1997).

3.3 Assessing Differences Between Ecosystems

CCF management involves regular modification of forest structure through harvest events. Foresters need to assess the impact of ecosystem modification caused by a particular harvest event (Puumalainen 1998). To be able to do this, they need to describe the structural changes caused by the tree removals. Harvest events modify the tree diameter distributions, the species distributions and spatial patterns. A major objective of many *CCF* systems is to mimic natural ecosystem dynamics. Therefore,

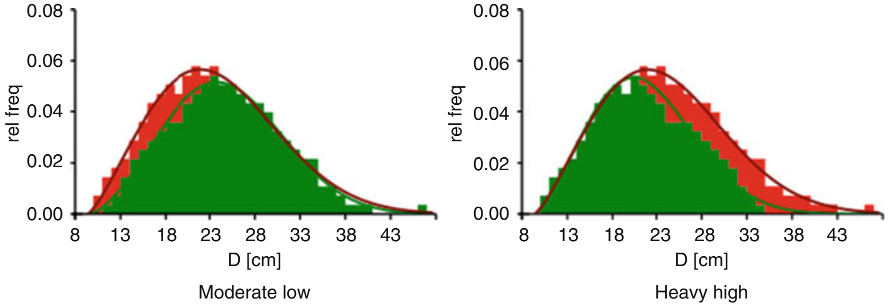


Fig. 2.17 Expected change of forest structure based on a combination of two adjectives describing a specific harvest event. The remaining and removed parts may be described by Weibull parameters

it is often desirable to compare the current state of a particular ecosystem with some ideal state, such as an unmanaged virgin forest. Accordingly, this chapter introduces methods to quantify differences between forest structures.

3.3.1 Describing Harvest Events with Linguistic Variables

A harvest event is a silvicultural activity which modifies ecosystem structure. A selective thinning, typical in CCF systems, is usually described by foresters using some adjective which defines the weight and the type of ecosystem modification. The European forestry literature abounds with definitions of specific harvest events (Kramer 1988, p. 180). A "moderate low thinning" removes the understorey trees, and the adjective "moderate" refers to the amount removed (Fig. 2.17).

A "heavy high thinning" involves the removal of all competitors of some specially identified trees which are believed to produce future value (Schober 1991). The expected change of forest structure based on a combination of two adjectives is shown in Fig. 2.17. Harvest events are described by a great variety of linguistic variables, including "selective thinning" (German: *Auslesedurchforstung*; see Schädelin 1942; Leibundgut 1978, p. 116; Abetz 1976; Johann 1982), "plenter thinning" (German: *Plenterdurchforstung*; see Borggreve 1891; Schütz 1989), "qualitative group selection" (German: *qualitative Gruppendurchforstung*; see Kato and Mülder 1983), "structural thinning" (German: *Strukturierende Durchforstung*; see Reininger 1987, p. 142). The "variable retention systems" In North America are defined by expressions like "strip shelterwood", "irregular shelterwood", "group retention" or "group selection" (Maguire et al. 2006).

The use of simple verbal expressions for describing complex structural modifications creates confusion, especially in multi-species forests. The linguistic variables *plenter thinning* and *selective thinning*, for example, may refer to virtually identical harvest events and ecosystem modification. On the other hand, one particular linguistic variable, like *plenter thinning*, may be interpreted in completely different ways. For a more detailed analysis of the confusion created by silvicultural

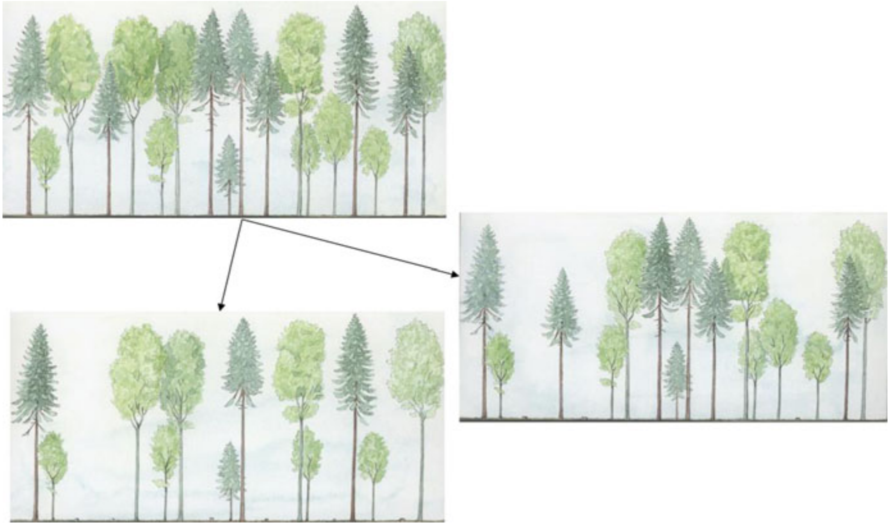


Fig. 2.18 Two hypothetical harvest events causing different structural changes in an uneven-aged forest

terminology refer to Füldner and Gadow (1994). To describe the structural changes caused by different harvest events is particularly challenging in uneven-aged multi-species forests which are selectively managed in a CCF system. Figure 2.18 presents an example of two harvest events and the corresponding structural changes. The example demonstrates that one would need many adjectives to differentiate between the two harvest events.

3.3.2 Simple Numerical Variables

Johann (1982) proposed the A-*thinning index* (Eq. 2.19) which defines a critical distance cd_{ij} between tree i and a given neighbour j. This critical distance is defined by the thinning intensity parameter A. Any neighbouring tree j which is located closer to tree i than the critical distance cd_{ij} is removed. Equation 2.19 shows that, apart from the thinning intensity parameter A the index uses the height diameter ratio of tree i and the diameter of the neighbouring tree j. The A-*thinning index* is thus sensitive to the h/d ratio of tree i: Trees with a greater h/d ratio are more heavily released than those with a lower h/d ratio. The A-values may range from 4 to 8. Higher values indicate decreasing thinning intensity. Johann (1982) recommended values of 4, 5 and 6 for even-aged pure Norway spruce forests which he considered to be synonymous with heavy, moderate and light release. A-values of 4 and 6 are frequently used values in thinning experiments (Hasenauer et al. 1996; Pretzsch 2002).

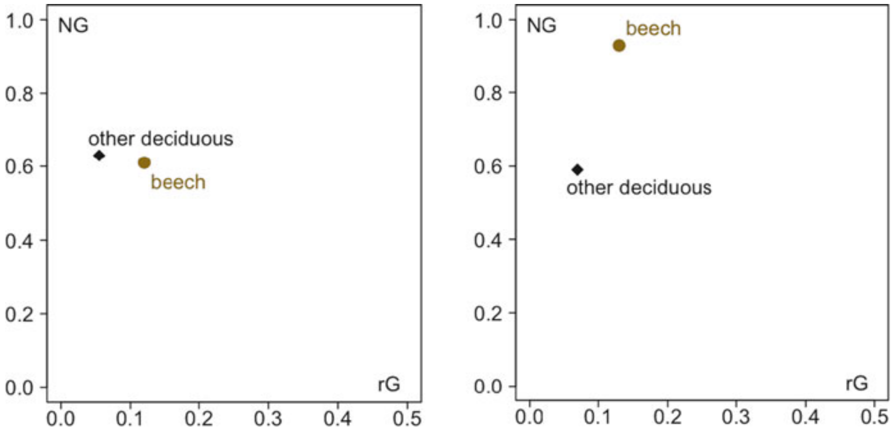


Fig. 2.19 Combinations of *rG* and *NG* values for beech (Fagus sylvatica) and other deciduous species in the experimental field plot Lensahn, for thinnings in 1999 and 2004. The values refer to total basal areas and stem numbers

$$dist_{ij} = \frac{h_i}{A} \cdot \frac{DBH_j}{DBH_i} \quad \text{or} \quad A = \frac{h_i}{dist_{ij}} \cdot \frac{DBH_j}{DBH_i} \tag{2.19}$$

The A-index has been used to simulate thinnings in spruce monocultures where silviculture is geared to clearfelling. The index may be difficult to adapt to realistic situations in CCF.

A practical variable for defining the weight of a thinning in CCF is the portion of the basal area (m^2/ha) removed during a harvest event. We denote this quantity as *rG* (Murray and Gadow 1993):

$$rG = \frac{removed\ basal\ area}{total\ basal\ area} \tag{2.20}$$

Correspondingly, the removed portion of the number of trees per ha may be denoted as *rN*. The ratio of these two quantities, *NG-ratio*, describes the type of thinning:

$$NG = \frac{rN}{rG} \tag{2.21}$$

Both *rG* and *NG* can be related to the change of the parameters of the Weibull distribution (Staupendahl and Puumalainen 2000). In an uneven-aged multi-species forest, the thinning weight (*rG*) and type (*NG*) may be calculated separately for each species or species group. Figure 2.19 presents an example.

The harvest event in 1999 removed more beech trees (slightly more than 10% of total basal area) than other deciduous species (5% of total basal area) while the *NG* ratios of about 0.6 indicate a high thinning (removal of predominantly bigger trees) in both groups. The structural modification caused by the 2004 harvest event was similar regarding the "other deciduous" group, but different in beech where the *NG* ratio was approaching unity, thus indicating removal of smaller beech trees, on average, when compared with the 1999 event. Only 16% of the basal area was removed in 1999, and 20% in 2004.

3.3.3 Removal Preferences in a Spatial Context

Sometimes we wish to know the spatial context of the trees that were removed during a harvest event: did the harvest event preferably target the dominant or the suppressed trees in the vicinity of their immediate neighbors? Where trees selected for removal located preferably in groups composed of one species or in mixed groups? To be able to do this, we can describe the neighborhood constellations of all removed trees and compare that with the neighborhoods of all trees before the harvest. An example of a variable that can be used is the species *mingling* (the proportion of the n nearest neighbors of a particular reference tree that are not of the same species as the reference tree). Another example is the *Dominance* (the proportion of the n nearest neighbors of a particular reference tree that are smaller than the reference tree). Considering four neighbors, each of these two variables can assume five values. The relative proportion of the removed reference trees divided by the proportion of all reference trees (before the harvest) within a structural class i and j is a measure of the removal preference (Pr_{ij}) within a given combination of structural classes:

$$Pr_{ij} = \frac{\text{Proportion of trees removed in structural class } ij}{\text{Proportion of trees existing before harvest in structural class } ij} \tag{2.22}$$

Table 2.4 presents the removal preferences in the *Lensahn* experiment during the 2004 harvest event, using the criteria Mingling and Dominance for beech and other deciduous species. The removed beech trees had occurred within a broad array of spatial constellations. They had been suppressed as well as dominant individuals. The highest removal preference ($Pr_{ij} = 6.37$) refers to neighborhood groups in which the removed beech was (a) the smallest tree among its four nearest neighbors ($U = 0.00$) and (b) surrounded by three neighbors that were not beech trees ($M = 0.75$). The other deciduous species were removed with high preference if they were co-dominant ($U = 0.75$) and surrounded by three out of four neighbors of a different species ($M = 0.75$).

3.3.4 Spatially Explicit Simulation of Harvest Events

Numerous tools have been developed to facilitate the prediction of tree growth in uneven-aged multi-species forests. Fairly advanced growth models are available

Table 2.4 Removal preferences Pr_{ij} for 5 classes of spatial mingling (M) and dominance (U), of beech and other deciduous species for the thinning in 2004 (numbers in bold show high preference)

	Mingling									
	Beech					Other deciduous				
Pr_{ij}	0.00	0.25	0.50	0.75	1.00	0.00	0.25	0.50	0.75	1.00
Dominance										
0.00	0.34	0.00	0.00	**6.37**	0.00	0.00	0.00	0.00	0.00	0.00
0.25	0.40	0.00	0.37	0.00	0.00	0.00	0.00	0.00	1.61	0.00
0.50	1.06	1.33	0.58	2.12	0.00	0.00	0.00	0.00	1.07	0.00
0.75	1.18	1.89	2.83	0.00	0.00	0.00	0.00	0.00	**2.15**	1.61
1.00	2.50	1.06	1.27	0.00	0.00	0.00	0.00	0.00	0.64	1.03

in many regions where forest ecosystems are selectively harvested. To be able to evaluate the specific dynamics of a selectively managed ecosystem, harvest event models are indispensible. In our experience, developing a model that predicts the structural modification caused by a harvest event is a rather challenging task, requiring effective algorithms that translate silvicultural prescriptions into tree selection algorithms resulting in complex forest structural modifications (Albert 1999; Hessenmöller 2002). It is not always possible to describe a harvest event using an index or a combination of indices that describe the weight and the type of tree removals. Continuous cover forestry is characterised by a wide range of specific harvesting types, including gap removals and *Z-tree* release thinnings. A *Z tree* in Germany is known as a *frame tree* in the UK, in reference to their role as representing the basic frame of a stand of trees. Frame trees are selected for their outstanding vitality, stem quality, stability and crown morphology. They are released from competition during a thinning by removal of their immediate competitors. *Z-tree* release thinning is widely practiced in Germany and simulation allows a more detailed and meaningful quantitative analysis of a harvest event. Examples were presented by Albert (1999, 2001; Fig. 2.20).

Albert (2001) presented a simulation of silvicultural alternatives to support the decision process in management planning. Management alternatives were generated by applying different thinning concepts, thinning intervals and changing the silvicultural objectives. The decision to identify a tree as a frame tree was based on the attributes crown vitality, stem quality, and distance to the nearest frame tree. The simulations demonstrated a realistic approach to harvest event prediction.

In his PhD thesis, Hessenmöller (2002) simulated the structural modifications caused by different harvest events. Each harvest event was described by a set of rules published in silvicultural textbooks. The selective thinnings were also characterised by different removal intensities to see if the removal probabilities would be affected by the thinning weight. Figure 2.21 shows a simulated selective and gap thinning for two types of spatial distribution.

A harvest event in *CCF* management modifies the tree size distribution, the species distribution and the spatial pattern. Foresters, who need to assess the impact

Fig. 2.20 Harvest event simulation on the experimental plot Frankenberg 131a. Identification of frame trees (•) and removed trees (+) based on the rule "remove the two most competitive neighbors of each frame tree" (Albert 1999)

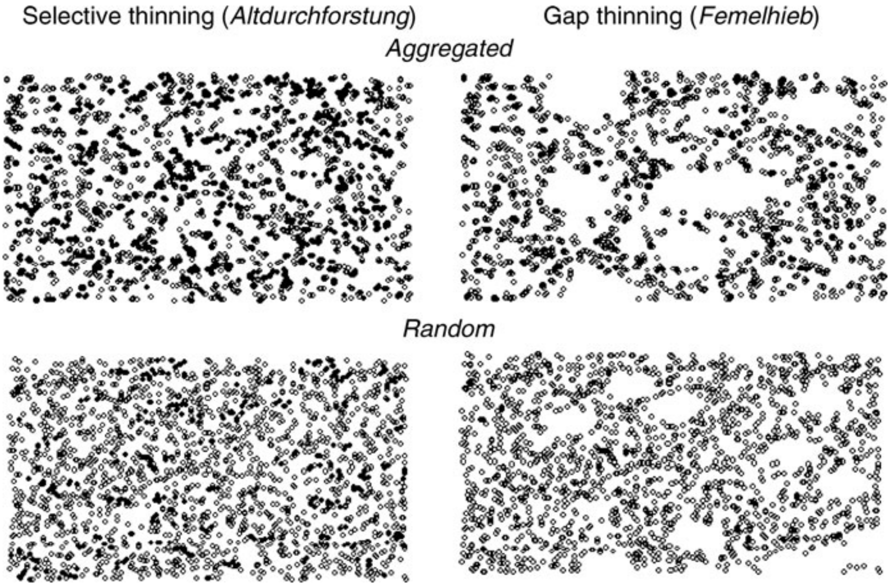


Fig. 2.21 Four examples of harvest event simulations presented by Hessenmöller (2002) for an aggregated and a random forest. The *left column* shows the removed (*black dots*) and remaining trees (*circles*). The *right column* just shows the remaining trees

$$G = \frac{\text{Area between Lorenz curve and line of perfect equality (C)}}{\text{Total area below the line of perfect equality (0BA)}}$$

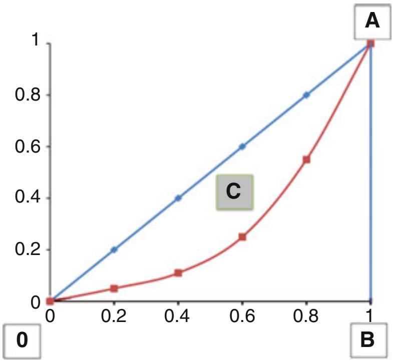


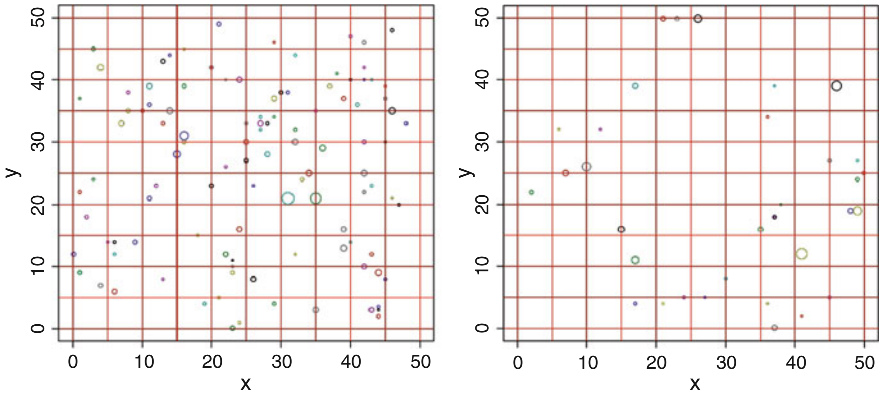
Fig. 2.22 Calculating the Gini Coefficient which is equal to the area C divided by the area of the triangle 0AB

of ecosystem modification caused by a particular harvest event, have to be able to describe structural changes caused by the tree removals. Mason et al. (2005) have shown that 2nd order statistics may also be usefully employed for analysing harvest events.

3.3.5 Comparing Ecosystems

As mentioned before, major objective of many CCF systems is to mimic natural ecosystem dynamics. Therefore, it is often desirable to compare the current state of a particular ecosystem with some ideal state, such as an unmanaged virgin forest. Accordingly, this section very briefly introduces a method based on the Gini coefficient (G), which permits a quantitative analysis of differences between forest structures. The Gini measures the area between the Lorenz curve (red colored in Fig. 2.22) and a hypothetical line of absolute homogeneity, expressed as a percentage of the maximum area under the line. The *Gini* coefficient is the ratio of the area between the *Lorenz* Curve and the line of absolute equality (numerator) and the whole area under the line of absolute equality (denominator). Based on Fig. 2.22, the *Gini* Coefficient is thus equal to the area C divided by the area of the triangle 0AB.

It is possible to use the *Gini* coefficient to measure the degree of structural regularity, i.e. the homogeneity of tree frequencies in the m x m cells. In a situation of perfect homogeneity, the Lorenz curve would overlap the line of perfect equality and the *Gini* coefficient would be equal to zero. In the theoretical situation of one cell containing all the trees, the Lorenz curve would coincide with the axes and the *Gini* coefficient would equal unity.



Plot 1 ($G = 0.5496$)

Plot 3 ($G = 0.7883$)

Fig. 2.23 The two Mexican plots used in the calculation of the *Gini* index, showing tree positions and superimposed cells

Two 50 × 50 m experimental plots located in Mexico are used to demonstrate the method. The study areas are located in the high range of the Sierra Madre Occidental, in the north-eastern region of the state of Durango, within the geographical coordinates 24º 30′ 59″ – 25º 30′ 20″ N and 106º 25′ 00″ – 105º 52′ 21″ W. Uneven-aged and highly semi-natural pine-oak stands are the predominant forest types, often mixed with *Pseudotsuga*, *Arbutus* and *Juniperus* (Wehenkel et al. 2011). The forest covering about 30,350 ha in total is owned and managed by a community (locally known as *ejido*). The ejido "San Diego de Tezains" belongs to the municipality of Santiago. The forest is mainly managed by selective removals, i.e. as a *CCF* system, according to the *Método Mexicano de Ordenación de Bosques Irregulares* (*MMOBI*; see Torres 2000). The following R code subdivides the area into one hundred 5 × 5 m subplots and counts the number of trees in each cell (Fig. 2.23).

```
attach(Durango1)      # the dataframe with xy tree locations
M<-50                  # Length of side of square plot
n<-length(x)           # Number of trees in plot
#set.seed(1); x<-runif(n,0,M); y<-runif(n,0,M) # Generate random trees if needed
m<-10                 # Number of row/column partitions
counts<-matrix(0,m,m) # Set up matrix for counts
# Fill the matrix
for  (i in 1:n){
    # find row cell
    ir<-ceiling(m*y[i]/M)
    ic<-ceiling(m*x[i]/M)
    counts[ir,ic]<-counts[ir,ic]+1}
result<-counts[m:1,] # result; sum(result)
```

The *Gini* coefficient is calculated using the library *reldist* and the following code:

```
vec<-c(result) #generate the vector of cell frequencies
library(reldist)
G<-gini(vec) # compute Gini coefficient
G # returns 0.5496 for plot 1; 0.7883 for plot 3
```

The following code generates a plot of the trees in the cell grids (Fig. 2.23):

```
plot(x,y,xlim=c(0,M), ylim=c(0,M), cex= BHD*0.05, col=sp.tab$col.sp)
abline(v=seq(0,M,length=m+1), col="red"); abline(h=seq(0,M,length=m+1), col="red")
```

This approach may be useful if we are interested in measuring the difference in structural homogeneity between two forests, e.g. a virgin forest and a managed one.

As mentioned before, the spatial statistics literature abounds with descriptions of point processes. But many of these are merely of academic interest. In the real world, despite advances in remote sensing (especially laser scanning) mapped tree coordinates are just not available. Thus it is reasonable to suggest that in the study of managed forests, which represent almost all that is left of the original "virgin" ecosystems, the most significant challenge is to develop methods for analysing spatial structures for situations where mapped tree data are not available. Forest structures tend to be complex in CCF systems. Meaningful structural analysis can be performed at affordable cost using the neighborhood methods described in several sections of this chapter.

3.4 Reconstruction and Simulation

Several authors have proposed methods which can be used to reconstruct an entire forest based on the structural information of a sample (Pretzsch 1997; Lewandowski and Gadow 1997; Hui et al. 2003; Pommerening 2006; Pommerening and Stoyan 2008). The objective of *reconstructing* a forest in this way may be to improve monitoring of silvicultural operations, to create realistic ecosystems as a basis for spatially explicit analysis and visualisation, or to use more advanced growth models based on known neighborhood constellations and competition status. A reconstruction is considered successful if there is a close resemblance between the real forest and the artificial one. It is usually impossible in practice to measure all the tree positions, the reproduction must thus be achieved with limited information. The information typically available to foresters involves distributions of neighborhood relations obtained during a routine forest inventory. A precondition to obtaining such information is the application of assessment techniques, such as described above. The simulation generates a reproduction of a real forest, based on a sample of the structural variables described in the previous section.

Pretzsch (1997) presented a method for producing a spatially explicit forest based on information about the manner in which two different tree species mingle (e.g., in clusters, rows or groups). He used empirical functions to estimate the distance to the nearest neighbor and a set of probability functions combining an inhomogenous Poisson process and hardcore process. Tree size mingling was not considered and the species are limited to two.

According to Lewandowski and Gadow (1997), a reproduction of a forest is considered to be perfect, if each tree in the real stand has a counterpart in the reproduced one, with exactly the same distances to its three nearest neighbors, and if all the species-mingling values and all size-differentiation values in the reproduction occur with the same frequencies as those in the real forest. Their simulation consisted of four separate phases, during which tree positions are generated and shifted and position attributes exchanged, until the distributions of the structural variables of the reproduction are close enough to those of the real stand. The initial positioning of the trees is done in phase 0. The positions may be allocated either at random or in some predefined manner. The denser the stand, the less important is the manner in which initial coordinates are generated. Phase 1 consists of 3 cyclic subphases: optimizing the distances to the first, the second and the third neighbor. The subphases are repeated until all the distances agree with those of the sample from the real forest. The purpose of phase 2 is to optimize the *mingling*. This is achieved by successively exchanging two trees of different species (swapping trees of the same species would not change the mingling value).[1] The swap is retained if it reduces the difference between the real and the simulated distributions of the species-mingling values, otherwise it is repealed. Phase 2 terminates when the difference between the two distributions of the mingling values cannot be further reduced. The *diameter differentiation* was optimized in phase 3. Again, a pairwise swapping of trees of the *same species* was done with the aim of attaining a distribution of the size-differentiation values, which closely resembles that of the original forest. The algorithm was later refined by Pommerening (2006) who reversed the analysis and thus enabled the synthesis of forest structure from the indices derived. He investigated this idea with a simulation model that uses the concept of cellular automata. The rules according to which the spatial pattern of tree positions developed in the stand matrix were deduced directly from the distributions of the structural indices of the input data. Pommerening used different combinations of indices to assess and simulate the structure of four sample stands. Simulations using species specific distributions of indices and limiting the number of neighbours to three or four were most successful at reconstructing the original stand structure.

---

[1] A total enumeration would be a fairly hopeless endeavour. For example, in a forest containing 47 trees (with 20 trees of species 1, 15 of species 2 and 12 of species 3), there are

$$\binom{47}{20} * \binom{27}{15} * \binom{12}{12} = 1.697 * 10^{20}$$

different ways in which the species can be assigned to the available positions.

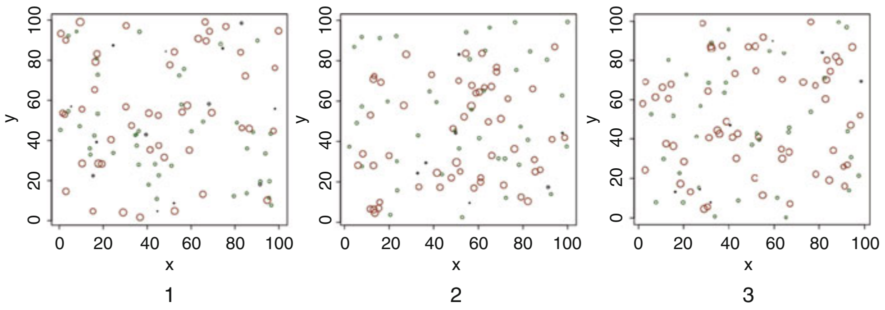


Fig. 2.24 Three random forests using the same distribution of tree species and tree diameters

Hui et al. (2003) proposed a method of reconstruction based on the *uniform angle index* (UAI) described in a previous section of this chapter. They generated tree coordinates for random, clumped and regular distributions, based on the sampled UAI.

Pommerening and Stoyan (2008) presented a method for synthesizing spatial point patterns from nearest neighbor summary statistics sampled in small circular subwindows. They used a stochstic optimisation technique based on simulated annealing and conditional simulation. The success of their reconstruction was tested by comparing tree point patterns reconstructed from sample data with the known patterns in three structurally different forests. Their validation has shown that it is possible to successfully reconstruct complex forest structures from *NNS* sample data.

Simulation is a useful technique for generating and analysing particular spatial patterns. The following R code generates a random forest, with 100 tree positions in a 100 × 100 m square plot:

```
x<-runif(100,0,100); y<-runif(100,0,100)              # Generate tree positions
sp<-sample(1:3,100, replace=TRUE,prob=c(10,50,40))   # Generate 3 species, in a proportion 1:5:4
ind1 <- sp==1; ind2 <- sp==2; ind3 <- sp==3           # Determine which trees of species 1, 2, 3
DBH <- rep(NA,100)                                     # Set up an empty vector for the DBH‘s
DBH[ind1] <- rweibull(sum(ind1),2.5,12)               # Fill in the DBH‘s for species 1 (Weibull)
DBH[ind2] <- rnorm(sum(ind2),40,3)                    # Fill in the DBH‘s for species 2 (Normal)
DBH[ind3] <- rnorm(sum(ind3),20,1)                     # Fill in the DBH‘s for species 3 (Normal)
plot(x,y,cex=DBH/20,col=sp)                            # Plot with point size and col=species
forest<-data.frame(sp,DBH,x,y)                        # use the data for further analysis
```

The number of trees in this example has been specified as a constant. Alternatively, the number of points may be specified as a realisation of a Poisson-distributed random variable with the parameters $\mu = \lambda \times a \times b$, where λ is the point density and a and b are the sides of a rectangular observation window ($x \leq$ random number $\times\, a$; $y \leq$ random number $\times\, b$). The results of three separate runs using the above code are shown in Fig. 2.24.

If the tree positions are regarded as constant, tree attributes (marks) can be assigned to them at random. To generate a random forest, the tree attributes are *permuted*, i.e. randomly reassigned to the observed tree locations (Hartung 1985, p. 96). There are $N!$ ways in which the observed values of one attribute can be assigned to the N tree positions. It is assumed that each of these $N!$ assignments has the same probability $1/N!$. Since the tree attributes are not modified, the distributions of tree species and diameters in the random forest are identical to that of the real forest. Lewandowski and Pommerening (1997) applied this method to generate 1,000 random forests with the associated 1,000 spatial patterns using random permutation. Using the same approach, Schröder (1998) found a surprising agreement in the mean observed and expected DBH differentiation (T_i) of different tree species cohorts in the *Knysna* natural forest in South Africa, using only the nearest neighbor.

There are many ways to simulate clustered forests. The following code (provided by Walter Zucchini) is useful because of its intuitive logic and simplicity:

```
Nc<-20                                    # 1. Specify  the number of clusters (Nc)
xc<-runif(Nc,0,100); yc<-runif(Nc,0,100)  # 2. Generate the cluster centers (e.g. uniformly)
lambda<-5                                 # 3. Specify the number of trees per cluster
Nt <-rpois(Nc,lambda)                    # 4. number of trees in each cluster (e.g. Poisson
distributed)
theta<-0.3                               # 5. parameter for the (x,y) distance from cluster center
# 6. For each tree, find the x and x distance to it's cluster center (e.g. exponential) and hence the
position
N<-sum(Nt)                           # total number of trees
x<-y<-rep(0,N)                       # create vectors for the positions
k<-0                                # tree "number"
for (i in 1:Nc)                     # loop over clusters
{
   for (j in 1: Nt[i])              # loop over trees within each cluster
  {
   k<-k+1                            # tree "number"
   x[k]<-xc[i]+rexp(1,theta)        # compute the x-position of this tree
   y[k]<-yc[i]+rexp(1,theta)        # compute the y-position of this tree
   }
}
```

Labelling the positions by generating species, independently and in a given proportion, tree diameters and other attributes, may be done using the code shown above (Fig. 2.24).

The simulation technique enables us to generate random structures which can be used as a basic reference. Some CCF management approaches claim to mimic developments in a natural ecosystem. Deviations from randomness may be helpful when analysing virgin or random forests in relation to managed forests.

4 Discussion

The objective of this contribution was to present methods that can be used to analyse forest structure and diversity with particular reference to CCF methods of ecosystem management. Trees are sessile and once established, their locations are fixed. Each population of trees exhibits a very specific structure and diversity at a given point of time. However, this condition is not static. Tree growth, regeneration and mortality represent important structuring processes. Specific structures generate particular processes of growth and regeneration. These processes in turn produce particular patterns and constellations. Thus, structure and processes are mutually dependent. Associated with a specific forest structure is some degree of heterogeneity or richness. In a forest ecosystem, such diversity does however not only refer to species richness, but to a range of phenomena that determine the heterogeneity within a community of trees. Most important on the macro scale is the diversity of tree dimensions.

Despite advances in remote sensing, mapped tree data are usually not available in the practice of CCF management. Forest inventories tend to provide tree data samples in small observation windows. Thus, in the overwhelming number of cases, nearest neighbor statistics (NNS) are a realistic choice. For this reason, a variety of NNS methods are presented in this chapter, including unmarked as well as market patterns and their use in structural analysis and description. NNS methods assume that the spatial structure of a forest is largely determined by the relationship within neighborhood groups of trees. These methods have important advantages over classical spatial statistics, including low cost field assessment and cohort-specific structural analysis.

Trees are objects which can be conveniently described by their locations and attributes. Thus, methods of point process statistics are useful when mapped data are available. Especially important are the *second-order statistics* (SOCs), which were developed within the theoretical framework of mathematical statistics and then applied in various fields of research, including forestry. Examples of *SOC*s in forestry applications are Ripley's K and Besag's L function, pair correlation functions and mark variograms. SOCs describe the variability and correlations in marked and non-marked point processes. Functional second-order characteristics depend on a distance variable r and quantify correlations between all pairs of points with a distance of approximately r between them. This allows them to be related to various ecological scales and also, to a certain degree, to account for long-range point interactions.

The analysis of forest structure informs us about the distribution of tree attributes, including the spatial distribution of tree species and their dimensions, crown lengths and leaf areas. The assessment of these attributes facilitates a comparison between a managed and an unmanaged forest ecosystem. In CCF it is especially important to evaluate structural modifications caused by harvest events. For this reason, analytical tools that describe structure and diversity are needed. The structure of a forest is the result of natural processes and human disturbance. Human disturbance in the form of selective tree removal in CCF has a major effect and forest structure is determined to a considerable extent by silviculture.

Important ecosystem functions, and the potential and limitations of human use are defined by the existing forest structures. The vertical and horizontal distributions of tree sizes determine the distribution of micro-climatic conditions, the availability of resources and the formation of habitat niches and thus, directly or indirectly, the biological diversity within a forest community.

Forest structure is not only of interest to students of ecology, but has also economic implications. Uneven-aged multi-species forests were found to be more resistant against natural hazards, and sometimes may generate superior financial returns. However, the management of uneven-aged forests requires not only a basic understanding of the species-specific responses to shading and competition on different growing sites, but also more sophisticated methods of sustainable harvest planning to ensure long-term productivity. A selective harvest event in an uneven-aged forest, involving removal of a variety of tree sizes within each of several species, is much more difficult to quantify and prescribe than standard descriptors of harvest events used in rotation management systems. Thus, the methods presented in this chapter may be helpful in assisting foresters, not only in analysing complex structures, but also in predicting the dynamics of an uneven-aged multi-species forest. A better understanding of *forest structure* is a key to improved definition of harvest events and to the modeling tree growth and recruitment following that event.

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Chapter 3
Continuous Cover Forestry in Finland – Recent Research Results

Timo Pukkala, Erkki Lähde, and Olavi Laiho

1 Introduction

In the past, Finnish landowners managed their forests using various types of selective cuttings. Only large trees were merchantable and therefore they were the only ones that were cut. Some smaller trees were used for household consumption. As a consequence, almost all forests were uneven-aged and uneven-sized (Lähde et al. 1991, 1992b). Most of the mature stands of Finnish forests are still rather uneven-sized despite the fact that even-aged management with clear felling and low thinning has been recommended for several decades (Pukkala et al. 2011a). This is mainly because of the gradual regeneration and ingrowth processes, which tend to convert even-aged stands into two- and multi-layered stands.

The traditional uneven-aged management with selective cuttings did not provide enough cheap raw materials for the expanding pulp and paper industry in the middle of the 1900s. Therefore, a group of leading forestry authorities started to promote even-aged rotation forestry (also called rotation forest management, RFM) with low thinning, clear felling, and planting. This type of silviculture was preferred for the production of small-sized wood required by the pulp and paper mills (Pukkala et al. 2011a). One outcome of this mission was the Declaration Against Uneven-aged Management, issued in 1948. Together with other initiatives, it led to the gradual

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_3,

abandonment of uneven-aged forest management in Finland (Siiskonen 2007). Also natural regeneration with seed tree and shelter wood methods was discouraged. As a result, more and more forest areas were established by planting, using selected seed of mainly Scots pine at first and Norway spruce more recently.

Many landowners realized that RFM with clear felling and planting would not maximize their benefits, and they wanted to continue practicing selective cuttings. However, since the traditional uneven-aged management was now considered unacceptable and interpreted even illegal, these landowners were punished, and timber sales from their forests were barred. They had to switch to clear-felling, planting, and low thinning if they wanted to get any income from their forests (Pukkala et al. 2011a).

There was no research available that would have proven the superiority of the new silvicultural paradigm. Not only the use but also the research of uneven-aged management was discouraged for several decades and no objective comparisons of alternative management systems were done. Until the 1980s, no experiments were established for comparing the yield and profitability of uneven-aged vs. even-aged management. It is evident that RFM provides more conifer pulpwood, but its profitability to the forest landowner was never analyzed. The harmful impacts of clear felling on visual landscapes, forest diversity, erosion, nutrient leaking, soil carbon balance, and various non-timber products and services were ignored when RFM was advertised.

At the moment, uneven-aged management is gaining popularity in Finland. The old practices are becoming acceptable again. They have been modernized and diversified, and are called as “continuous cover forestry” (CCF). All management systems that do not use clear-felling and planting are regarded as different variants of CCF. The increasing interest towards CCF is for several reasons. One is the decreasing importance of pulp and paper industries in the Finnish economy. Instead, nature tourism is a growing economy. Paper consumption is decreasing in countries that import Finnish paper. The high energy consumption of mechanical mass industry with harmful climatic impacts is another reason for the gradual decline of this industry, reducing the need for spruce pulpwood. The harmful impacts of RFM on the recreational value and biodiversity of forests have been increasingly emphasized. According to recent surveys (Valkeapää et al. 2009), only 5% of Finns and half of forest landowners approve clear-felling without any reservations.

One reason for the incipient new era of increased freedom and variability in forest management are the encouraging results on the yield and profitability of CCF (Pukkala et al. 2010; Tahvonen et al. 2010). Although investigations on CCF were actively constrained for several decades, some researchers managed to set up silvicultural experiments in which various forms of CCF were included. These experiments were established during the 1980s with the aim of comparing alternative silvicultural systems (Lähde et al. 1992a). The experiments have already provided useful empirical data for growth and yield analyses (e.g., Lähde et al.

Uneven-aged spruce forest in South Finland (Photo: Timo Pukkala)

1999a). The measurements of the experiments have also been used for growth modeling. The growth models, in turn, have enabled detailed long-term analyses of the yield and profitability of alternative management systems. Several studies on the growth and profitability of CCF have been conducted and reported during the past few years. Some studies also deal with the non-timber benefits provided by different management systems. An indication of the growing interest and activity is that two silvicultural text books on CCF have recently been published in Finland (Valkonen et al. 2010; Pukkala et al. 2011a).

This chapter provides a review on the recent Finnish research related to CCF. The next section describes the experiments established on CCF and the results obtained from them. The third part reviews the models developed for the stand dynamics of uneven-aged forests. Then, results on economically optimal uneven-aged management are summarized. A separate section is devoted to multifunctional forestry. Finally, the last section discusses what research is still necessary for the improved management of Finnish forests and for increasing reliability of the decision support provided by models and calculations.

Uneven-aged pine forest in South Finland (Photo: Jussi Saarinen)

2 Experiences from Silvicultural Experiments

Several experiments on CCF were established in Finland during the 1980s. The best known experiments are those of Honkamäki and Vessari in southern Finland. The forests of these experimental areas were regenerated mainly for spruce by the shelter wood method during the 1940s. The shelter trees were removed in 1957 and 1960. The experiments that compare different silvicultural methods were started in 1986 and 1987. Both experiments consist of tens of plots (92 plots altogether). Four main treatments were included: (1) RFM with different intensities of low thinning and CCF with various types of (2) single-tree selection and (3) diameter-limit cutting; and (4) no cuts at all (control). Tree species composition and post-cutting stand density were varied in low thinning and single-tree selection. Every treatment was represented by several plots, and the treatments of the plots were randomized. The plots have been measured at 3-year intervals, on average. The trees were mapped in 1994. Cuttings have been done from one to four times depending on the treatment, in 1986/1987, 1994, 2003/2004 and 2009.

At the latest measurement, in 2009, the even-aged plots had already reached the rotation age of RFM, and several of them were clear felled in autumn 2009. The other even-aged plots were treated with different variants of regenerative cuttings

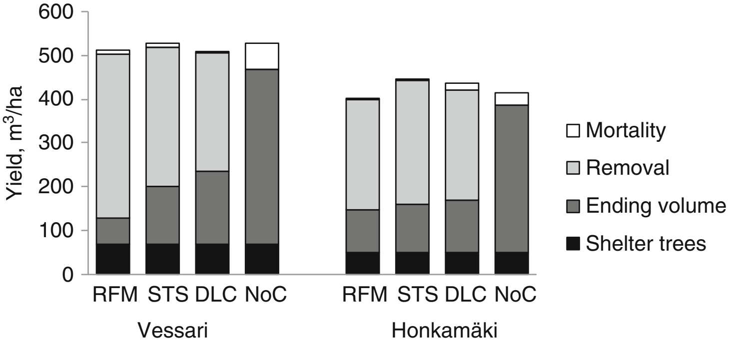


Fig. 3.1 Total volume production in different management systems in the Vessari and Honkamäki experiments during a 75-year period (typical rotation length in even-aged forestry). "Shelter trees" refers to the volume increment of shelter trees during the first years of the rotation (shelter tree cut – removal of shelter trees). *RFM* rotation forest management (even-aged management); *STS* single-tree selection; *DLC* diameter-limit cutting; *NoC* no cuttings

(seed tree and shelter wood cuttings using different intensities, and conducted as high, uniform or low thinning). Therefore, the experiments already provide growth and yield information from a period that corresponds to one full rotation (75 years) of even-aged management. In Fig. 3.1 the total yield equals the sum of ending volume, harvested volume, volume of mortality, and the volume increment of the initial shelter trees. The growth of the shelter trees during 1945–1957/1960 has been estimated at 70 m^3 ha^{-1} in Vessari and 50 m^3 ha^{-1} in Honkamäki.

According to the measurements, the total wood production has been 3% (Vessari) or 11% (Honkamäki) higher in single-tree selection than in RFM. If mortality is included in the yield, control (no cuttings) is the second best in Vessari and the third in Honkamäki. In Vessari, the diameter-limit cutting removed all trees larger than 9 cm in 1986. This kind of uneven-aged management has commonly been regarded as exploitive and destructive. However, since the spruce under-storey was well developed, the total production of diameter-limit cutting was the same as in even-aged management. In the Honkamäki experiment, diameter-limit cuttings were carried out in 1987 and 2004, both removing all trees larger than 16 cm in dbh. Diameter-limit cutting has been the second best in total wood production in Honkamäki.

Assuming that the harvested timber was sold with the current stumpage prices, the net present values of different management systems, discounted to 1986/1987, are as shown in Fig. 3.2. The net present value (NPV) of the ending forest (predicted net present value of cuttings after 2009, to infinity) has been calculated with the models of Pukkala (2005), and discounted to 1986/1987. The most profitable management option in 1986/1987 had been diameter-limit cutting, both in Vessari and Honkamäki. RFM with low thinning had been the least profitable option in both experiments.

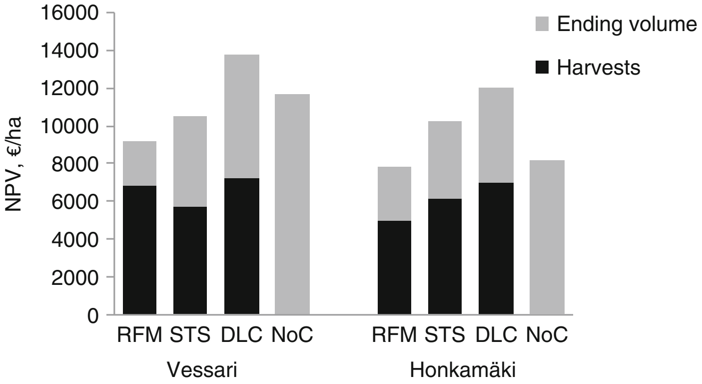


Fig. 3.2 Discounted net revenue (with 3% rate) in alternative management systems in the Vessari and Honkamäki experiments. All net revenues have been discounted to 1986 or 1987 when the stands were 40 years old and the experiments were started. The net present value (NPV) due to future cuttings (NPV of ending volume) has been estimated with the models of Pukkala (2005). Some of the even-aged plots were clear felled and planted with Norway spruce in 2009 and some were left for natural regeneration without any site preparation. *RFM* rotation forest management; *STS* single-tree selection; *DLC* diameter-limit cutting; *NoC* no cuttings

Many other experiments were also established in different parts of southern and northern Finland during the 1980s, mainly for Scots pine in the north and Norway spruce in the south. The total number of plots in these scattered smaller experiments is about 150 in southern Finland. These plots have been monitored less systematically than the Vessari and Honkamäki experiments. The measurement interval has typically been 5–10 years. The drawback in these experiments is that trees have not been numbered and their coordinates have not been measured. Therefore, the growth and survival of individual trees are not known. Yet, the plots provide valuable information for growth and yield studies (e.g., Pukkala et al. 2011b) and for validating the results based on other materials. An advantage is that the plots are reasonably large (40 m by 40 m) and each plot is surrounded by forest with similar management.

Several research articles have been published based on these experiments. For instance, regeneration in uneven-aged stands was studied in the beginning of 1990s (Lähde 1992a, b). Plots treated with commercial selective fellings 10–30 years earlier and those treated with single tree selection 4–7 years ago produced, in general, rich regeneration. In both materials, regeneration decreased with increasing stand volumes. During the first monitoring period of 11 years, the CCF-plots had almost 20% faster volume increment than the low-thinning plots of RFM (Lähde et al. 2002a; Saksa et al. 2003). Another study compared single-tree selection, low thinning and diameter limit cutting (Lähde et al. 2001) and found single-tree selection to be the most productive.

In the experiment established in an old-growth Norway spruce stand in eastern Finland and treated with various modifications of CCF-management, the structural diversity index (Lähde et al. 1999b) was clearly the highest in untreated plots (Lähde et al. 2002b). Single-tree selection was the second best whereas even-aged management resulted in the lowest structural diversity.

Recently, new large silvicultural experiments have been established in state-owned forests. They aim at mimicking the natural disturbance dynamics occurring in forests (University of Helsinki 2010). Two large forest areas (1,200 and 650 ha) have been assigned for the studies. The idea is to analyze, not only even-aged and uneven-aged management, but also management systems that are between these extremes. An example of the in-between systems is regeneration by small canopy gaps. The purpose is to compare cuttings that mimic small disturbances (individual trees and tree groups removed), medium disturbances (many trees removed and gaps created), or heavy disturbances (large-scale clear fellings). The experiments are useful but since the study was initiated in 2009, it will take many years until major results can be expected.

Measurements of national forest inventory plots have also been used to compare even- and uneven-sized forests (e.g., Lähde et al. 1994a, b, 2002c). These data sources have been used to analyze the structure of the forests (Pukkala et al. 2011a) and to examine regeneration (Lähde et al. 1999c) and tree growth in different stand types (Pukkala et al. 2009). National forest inventory data cover all growing sites and stand types and represent the whole surface area of Finland. The drawback is that the management history of stands is not well known and only periodical growth information is available. Yet, national forest inventory data can be used for modeling the dynamics of uneven-aged forests. These models may then be used to analyze the wood production and profitability of alternative management systems.

3 Growth and Yield Modeling

The first growth and yield models developed specifically for uneven-aged forests in Finland were transition matrix models (Pukkala and Kolström 1988; Kolström 1993). A transition matrix model predicts the probability of a tree of certain size to stay in the same diameter class, or move to the next higher class (Chap. 6). Mortality is also accounted for since, in these models, trees that do not stay in the same class nor move to another class are taken as mortality. Ingrowth is also incorporated into the transition matrix models.

The above models have been used very little for analyzing the yield and profitability of uneven-aged management because the modeling data did not include stands that are known to have been treated with repeated selective high thinnings. A more recent transition matrix model was presented by Tahvonen et al. (2010). This model is based on data collected in the Vessari and Honkamäki experiments, where more than half of the plots represent uneven-aged management. However, this new transition matrix model is only locally valid since the modeling data came from two experiments only, located close to each other.

Individual-tree models are another type of growth models suitable for simulating the dynamics of uneven-aged stands. Management planning in Finnish forestry has a long tradition of employing individual-tree models, which means that this model type is familiar to forest planners and people who develop decision support systems for forestry decision making. The first individual-tree models specifically targeted to uneven-aged stands are those of Pukkala et al. (2009). They were based on three data sets with a total of 50,000 diameter growth measurements: (1) the experiments of Honkamäki and Vessari, (2) sample trees of the third national forest inventory (1951–1953), and (3) a set of plots measured in irregular conifer stands in Eastern Finland (158 plots). The majority of the data comes from uneven-sized stands since most forests were still uneven-sized during the third national forest inventory, and half of the plots of Vessari and Honkamäki were managed using CCF.

In the individual-tree approach the minimum set of models required for simulating stand dynamics consists of

- individual-tree diameter increment model;
- individual-tree survival model; and
- ingrowth or regeneration model.

In addition, height, volume and biomass models are usually employed in calculations, but they are not necessary for simulating the stand dynamics. Pukkala et al. (2009) fitted ingrowth models using 5 cm dbh as the ingrowth limit; the models predict the number of trees that pass the 5-cm limit during the coming 5-year period. However, since these ingrowth models were based on a rather small data set (ingrowth was unknown for the national forest inventory plots) and the model behavior was not always logical, Pukkala et al. (2011b) fitted new models based on 140 plots measured two to four times in various silvicultural experiments in southern and central Finland. The ingrowth diameter limit was now 0.5 cm. Therefore, the best set of models currently available for simulating the dynamics of uneven-aged stands with the individual-tree approach consists of the diameter increment and survival models of Pukkala et al. (2009) and the ingrowth models of Pukkala et al. (2011b). In this model set, the diameter increment models are as follows:

$$\begin{aligned}\textit{Scots pine}\colon i_{\mathrm{d}} = 1.110 \times \exp\big(&-7.758 - 0.0530 BAL_{\mathrm{Others}} - 0.0335 BAL_{\mathrm{Spruce}} \\ &- 0.266\ln(G) + 0.237\sqrt{d} - 0.000901 d^2 - 0.238 MT - 0.333 VT \\ &- 0.612 CT - 1.201 ClT + 1.229\ln(TS)\big)\end{aligned}$$

$$\begin{aligned}\textit{Norway spruce}\colon i_{\mathrm{d}} = 1.124 \times \exp\big(&-5.317 - 0.0106 BAL_{\mathrm{Others}} - 0.0430 BAL_{\mathrm{Spruce}} \\ &- 0.486\ln(G) + 0.4557\sqrt{d} - 0.000927 d^2 - 0.180 MT \\ &- 0.450 VT - 0.929 CT + 0.823\ln(TS)\big)\end{aligned}$$

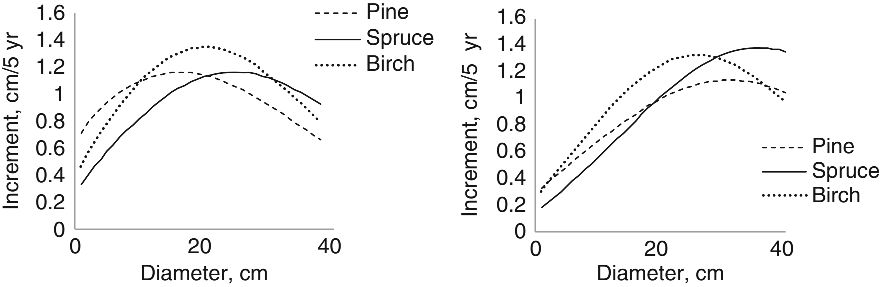


Fig. 3.3 Dependence of diameter increment on diameter in an uneven-aged stand growing on a mesic site in southern Finland when the stand basal area is 25 m^2 ha^{-1}. In the *left* diagram, *BAL* has a fixed value of 10 m^2 ha^{-1}. On the *right*, *BAL* decreases from 25 m^2 ha^{-1} to zero when diameter increases from zero to 40 cm

$$\begin{aligned}\textit{Birch}\colon i_\mathrm{d} = 1.127 \times \exp\big(&-11.873 - 0.0304BAL_\mathrm{Others} - 0.0474BAL_\mathrm{Spruce}\\&- 0.173\ln(G) + 0.446\sqrt{d} - 0.00123d^2 - 0.121MT - 0.227VT\\&-0.524CT + 1.627\ln(TS)\big)\end{aligned}$$

where i_d is diameter increment (cm/5 years), d is dbh (cm), G is stand basal area (m^2 ha^{-1}) and *TS* temperature sum (°C). *TS* is equal to the sum of mean daily temperatures minus 5° of those days on which the mean daily temperature is at least 5°C. *MT*, *VT*, *CT* and *ClT* are indicator variables indicating whether the site is mesic (*MT*), sub-xeric (*VT*), xeric (*CT*) or poorer than xeric (*ClT*). *BAL* is the basal area in trees larger than the subject tree (m^2 ha^{-1}). *BAL* is calculated separately from spruces (BAL_Spruce) and the other tree species (BAL_Others). *BAL* describes the competitive status of the tree within a stand. It is the most significant predictor in the models. The models indicate that small trees grow slowly and large trees grow well in an uneven-aged stand (Fig. 3.3, right). Slow early growth results in good wood quality, and fast growth at older ages maintains high relative value increment. Good diameter growth in large trees is no longer a disadvantage for wood quality.

The survival models are:

$$\textit{Spruce}\colon p = \frac{1}{1 + \exp\left[-\left(4.418 + 1.423\sqrt{d} - 1.046\ln(G) - 0.0954BAL_\mathrm{Spruce}\right)\right]}$$

$$\textit{Pine and birch}\colon p = \frac{1}{1 + \exp\left[-\left(0.496 + 1.649\sqrt{d} - 0.06BAL\right)\right]}$$

where p is the probability that a tree survives during the coming 5 years. According to the models, large trees have a greater chance to survive than small ones. Trees facing much competition (having large *BAL*) die more often. A few percents of

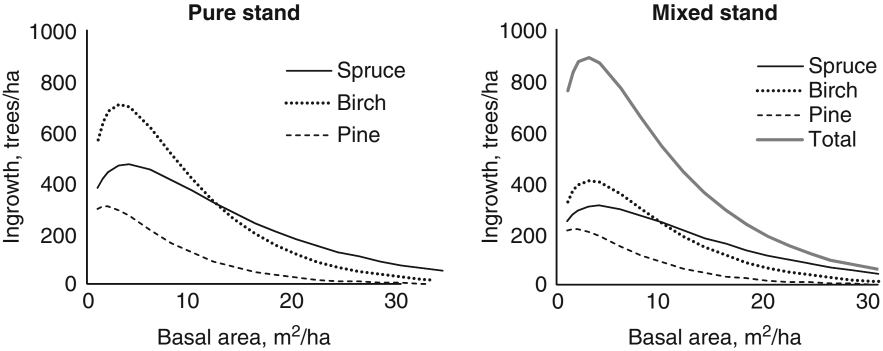


Fig. 3.4 Five-year ingrowth of different species in pure stands (*left*) and in a mixed stand (*right*) on mesic site according to the ingrowth models. The share of each species in the mixed stand is 33% of basal area

trees smaller than 10 cm die during a 5-year period. Trees larger than 10 cm die infrequently. Spruce trees survive better than pines and birches.

The ingrowth models are (Fig. 3.4):

$$\textit{Scots pine}: Ni = \exp(5.547 + 0.309\ln(G_{\text{Pine}}) - 0.141G + 0.356MT^{-})$$

$$\textit{Norway spruce}: Ni = \exp(6.030 + 0.373\ln(G_{\text{Spruce}}) - 0.099G - 0.539MT^{-})$$

$$\textit{Birch}: Ni = \exp(6.500 + 0.496\ln(G_{\text{Birch}}) - 0.161G + 0.433MT^{-})$$

$$\textit{Other hardwoods}: Ni = \exp(5.872 + 0.310\ln(G_{\text{Other}}) - 0.177G - 0.082MT^{-})$$

where Ni is the number of trees that pass the 0.5-cm diameter limit during the coming 5-year period. Indicator variable MT^{-} equals 1 if the site is mesic or poorer, otherwise $MT^{-} = 0$.

The above model set has been used in all the optimizations and calculations discussed in this chapter. By using the models, one 5-year time period of stand development is simulated as follows:

- Calculate the predictors of the models
- Calculate the survival probabilities of trees and multiply the frequencies of trees by their survival probabilities
- Calculate the diameter increments of trees and add them to the current tree diameters
- Calculate ingrowth and add new trees to the tree list
- Calculate tree height, volume, biomass and other characteristics as required

A forest stand may be represented by a list of trees in different size classes or by a plot in which each tree is known and treated individually in the simulation. In the first case, every tree has its own frequency, i.e., it represents a certain number of trees per hectare, and mortality is simulated by reducing the tree's frequency. In the latter case, mortality is simulated by assigning the tree either as a survivor or non-survivor. This can be done by means of random numbers (the predicted survival probability is compared to a uniformly distributed random number) or using a certain limiting survival probability (e.g., 0.5) beyond which a tree is assigned as a survivor. In the analyses of this chapter, a stand was represented by a list of sample trees, and mortality was simulated by multiplying tree frequencies by the trees' survival probabilities. When a list of sample trees corresponding to a certain diameter distribution is generated, sample trees are taken at 0.5- or 1-cm intervals over the range of breast height diameters.

4 Optimization of CCF Management

4.1 Problem Formulations and Optimization Methods

We have used the following simple formula to calculate the net present value for the steady-state management of an uneven-aged forest (Chang 1981):

$$NPV = \frac{N_T}{(1+i)^T - 1} - A_T \tag{3.1}$$

where N_T is the net income obtained regularly at T-year intervals (after T, $2T$, $3T, \ldots$ years), T is the cutting cycle, i is discount rate (percentage divided by 100) and A_T is the stumpage value of the post-thinning stand. The NPV of a certain post-thinning steady state diameter distribution is calculated as follows. (1) Calculate the value of the growing stock (A_T); (2) simulate stand development for T years using the models explained above; (3) reduce the frequencies of diameter classes back to their initial levels and calculate the net income (N_T) from the harvesting costs and roadside values of the removal; and (4) calculate NPV from Eq. 3.1 (see also Chang and Gadow 2010).

If the purpose is not to find the optimal steady-state structure but to optimize the management of an existing stand, one possibility is to simulate the growth and management for a long enough time and calculate the NPV from the simulation results. If the simulation period is short or discounting rate is low, the net present value of the ending growing stock should be taken into account. One way of doing this is to continue simulation until the forest reaches a steady state, then calculate the periodical income of the steady-state forest, and add the NPV of this periodical income to the NPV accrued before the steady state. It is also possible to constrain

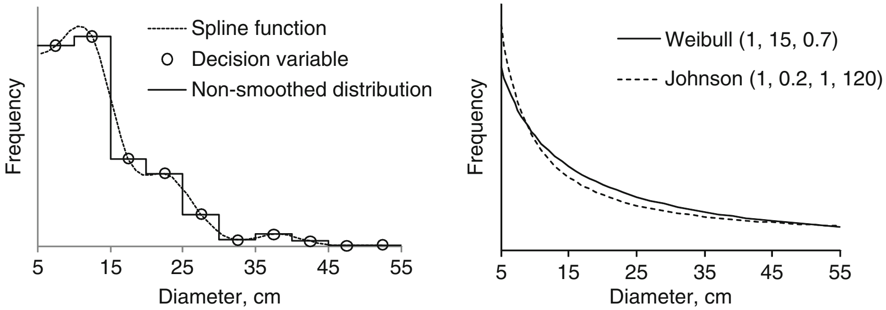


Fig. 3.5 Alternative ways to describe the post-thinning diameter distribution of an uneven-aged stand in optimization. On the *left*, the decision variables are the frequencies of diameter classes, which may be smoothed with a spline function to avoid stepwise distributions. On the *right*, the decision variables are the parameters of the Weibull or Johnson's SB function

the optimization to reach a predefined steady-state uneven-aged structure during a conversion period. In this case, the last simulated cutting, which converts the stand into the steady-state structure, is not optimized.

The most straightforward way to formulate the optimization problem is to use the post-cutting frequencies of different diameter classes as decision variables. This choice has the advantage of being flexible, i.e., the diameter distribution is not constrained to a certain shape (such as the "inverse J" shape). A drawback is that the number of decision variables becomes large, especially if a series of successive transformation cuttings or mixed stands are optimized, in which different tree species have their own, species-specific diameter distributions.

To analyze the effect of detail in describing the stand structure, Pukkala et al. (2010) compared four alternative sets of decision variables in the optimization of the steady-state structure of a spruce stand (Fig. 3.5):

1. Frequencies of 4-cm wide diameter classes
2. Smoothed frequencies of 4-cm wide diameter classes
3. Parameters of the Weibull distribution
4. Parameters of the Johnson's SB distribution

In all cases, the distribution was sampled at 0.5-cm intervals. The result was a set of sample trees that represented the stand in simulation. When the frequencies of 4-cm diameter classes were used as decision variables (option 1), the frequencies of all sample trees falling within a given diameter class were equal (Fig. 3.5). In the case of smoothed frequencies, a cubic spline function was fitted to the decision variables, and the frequencies of sample trees were obtained from the spline function (Fig. 3.5). Therefore, the frequencies of sample trees within a 4-cm diameter class were unequal.

When using the Weibull or Johnson's SB function, the optimized decision variables were the parameters of the function, together with the total number of trees per hectare. The Weibull density function is defined as follows:

$$f(d) = \left(\frac{c}{b}\right)\left(\frac{d-a}{b}\right)^{c-1} \exp\left[-\left(\frac{d-a}{b}\right)^{c}\right]$$

where d is diameter, $f(d)$ is frequency of diameter d, a is minimum diameter, b is a location parameter and c is a shape parameter. The Johnson's SB function is:

$$f(d) = \frac{\delta}{\sqrt{2\pi}} \frac{\lambda}{(d-\xi)(\xi+\lambda-d)} \exp(0.5z^2)$$

with

$$z = \lambda + \delta \ln\left(\frac{d-\xi}{\lambda+\xi-d}\right)$$

where ξ is the minimum and λ the range of the distribution, and γ and δ are shape parameters.

In all four methods to describe the post-thinning diameter distribution, the maximum diameter retained after a thinning was used as an additional decision variable; post-thinning frequencies of all trees larger than this diameter were taken as zero. Therefore, in the case of the Weibull function, the optimized variables were the total number of trees, parameters a, b, and c of the distribution function, and the maximum diameter (5 decision variables). There was one more decision variable when Johnson's SB was used because this function has two shape parameters. When frequencies were used, the number of diameter classes was 10 (with class mid-points of 7, 11, 15, …, 43 cm), which means that the number of optimized decision variables was 11.

The growth of a forest stand was simulated for 20 years ($T = 20$ years), after which a felling was simulated, which reduced the frequencies of diameter classes to their initial levels. In this analysis, N_T was calculated from the stumpage value of the removal and A_T from the stumpage value of the initial stand. A penalty function was used to guarantee that the remaining frequencies of diameter classes, after simulating a cutting at the end of cutting cycle, were not less than the initial frequencies. This requirement may be violated if the ingrowth to a certain class is less than the transition of trees to larger diameter classes. Thus, the objective function, which was maximized in the optimizations, was NPV minus a penalty:

$$OF = NPV - Penalty$$

with

$$Penalty = \sum_{i=1}^{n} \left(200i \left|f_i' - f_i\right|\right)$$

where f_i' is the frequency of trees in diameter class i at the end of cutting cycle, i.e., after simulating stand development for T years and simulating a thinning treatment

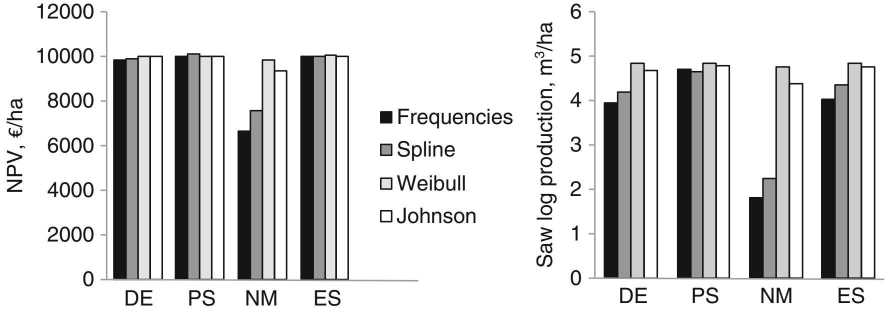


Fig. 3.6 Mean value of objective variable (net present value with 2% discount rate or mean annual saw log production) in 10 repeated optimizations with four optimization methods and four ways to describe the post-thinning diameter distribution in a spruce stand on herb-rich heath in South Finland (*DE* Differential evolution; *PS* Particle Swarm Optimization; *NM* Nelder-Mead method; *ES* Evolution Strategy Optimization)

at the end of the cycle, f_i is the initial frequency of trees in diameter class *i*, and *n* is the number of diameter classes. The multiplier *i* in the formula ensures that a lack of large trees was penalized more than a lack of small trees. Another optimization was made in which the saw log harvest was maximized instead of NPV.

Four different optimization techniques were used to solve the problems for a Norway spruce stand growing on medium site in South Finland. The methods were Differential Evolution, Particle Swarm Optimization, the Nelder-Mead method (also called Amoeba Search or Polytope Search), and Evolution Strategy Optimization (Pukkala 2009).

Based on the results, it was concluded that the Weibull function (together with maximum retained diameter) and Evolution Strategy Optimization was the best combination of decision variables and optimization method (Fig. 3.6). Therefore, the decision variables used in the optimizations described in this chapter use Evolution Strategy Optimization with the following decision variables: parameters *a*, *b*, and *c* of Weibull distribution (*aW*, *bW* and *cW*), number of trees per hectare (*N*), and maximum diameter retained in cutting (*Dmax*). Later on, it was found that assuming *aW* to be equal to zero does not usually deteriorate the result. Therefore, the optimized decision variables have mostly been *bW*, *cW*, *N* and *Dmax* in the calculations reported in this chapter. We recognize that the use of the Weibull distribution function constrains the diameter distribution to be descending or unimodal, but this does not seem to be a problem, at least when steady-state stand structures are optimized.

4.2 *Optimal Steady-State Management*

Tables 3.1 and 3.2 show the optimal steady state post-thinning diameter distributions of spruce and pine stands on different growing sites in South (temperature sum 1,200 d.d.) and North Finland (temperature sum 900 d.d.) when net present value

Table 3.1 Optimal steady state post-thinning stand structure in South Finland when net present value is maximized with 2% discount rate

Variable	Fertile (OMT) Spruce	Mesic (MT) Spruce	Sub-xeric (VT) Pine	Xeric (CT) Pine
Cutting cycle, years	15	15	20	30
Basal area at cutting, m^2 ha^{-1}	21	20	20	21
Remaining basal area, m^2 ha^{-1}	9	8	8	7.5
Remaining volume, m^3 ha^{-1}	59	49	47	37
Remaining no. of trees, ha^{-1}	1,612	1,332	1,403	1,542
• 0–5 cm	784	675	648	791
• 5–10 cm	417	323	364	393
• 10–15 cm	248	199	263	238
• 15–20 cm	131	111	128	120
• 20–25 cm	28	24	–	–
• >25 cm	–	–	–	–
Wood production, m^3 ha^{-1} a^{-1}	6.6	5.2	4.8	3.0
Saw log harvest, m^3 ha^{-1} a^{-1}	4.1	3.3	2.9	1.8
Net income, € ha^{-1} a^{-1}	269	209	177	106
Net present value (2%), € ha^{-1}	10,225	7,912	6,398	3,470

Table 3.2 Optimal steady state post-thinning stand structure in North Finland when net present value is maximized with 2% discount rate

Variable	Mesic (MT) Spruce	Sub-xeric (VT) Pine	Xeric (CT) Pine	Dry (ClT) Pine
Cutting cycle, years	20	30	40	50
Basal area at cutting, m^2 ha^{-1}	19	19	19	16
Remaining basal area, m^2 ha^{-1}	9	7	7.5	7.5
Remaining volume, m^3 ha^{-1}	52	42	37	30
Remaining no. of trees, ha^{-1}	2,137	1,457	1,698	2,379
• 0–5 cm	1,230	751	748	1,568
• 5–10 cm	497	357	395	479
• 10–15 cm	282	237	244	241
• 15–20 cm	128	112	111	92
• 20–25 cm	–	–	–	–
Wood production, m^3 a^{-1}	4.1	3.0	1.9	0.9
Saw log harvest, m^3 ha^{-1} a^{-1}	2.3	1.8	1.1	0.5
Net income, € ha^{-1} a^{-1}	155	106	67	24
Net present value (2%), € ha^{-1}	5,307	3,348	1,783	352

is maximized with a 2% discount rate. The cutting cycle varies from the 15 years of fertile sites in South Finland to 50 years of poor sites in North Finland. When economic profitability is maximized the optimal management involves removing all trees larger than 20–25 cm. This means that all sawlog-sized trees are harvested with the exception that some small log-sized trees are retained in the spruce stands of South Finland.

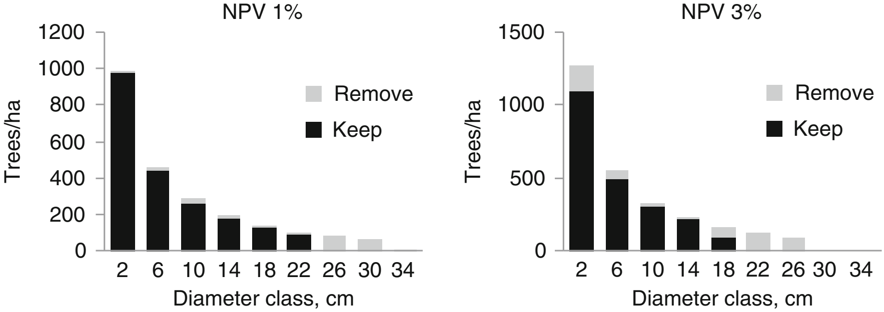


Fig. 3.7 Effect of discount rate on the steady-state structure of a spruce stand growing on fertile herb-rich site in Central Finland

The remaining stand basal areas are low, 7.5–9 m^2 ha^{-1}, and slightly less than the accepted minimum in the current forestry regulations for the thinning of an even-aged stand. However, since the number of trees is high (always more than 1,300 trees taller than 1.3 m), the post-cutting stand is not under-stocked. If the selection cutting is interpreted as a removal of the over-storey trees, then there is no discrepancy between the optimization result and the current regulations. The mean annual wood production ranges from 0.9 m^3 ha^{-1} on the poor sites of North Finland to 6.6 m^3 ha^{-1} on fertile sites in South Finland. If wood production were maximized, instead of profitability, the sustainable yield would be 1.5 to 7.8 m^3 ha^{-1} a^{-1} (Pukkala et al. 2010).

The results of Tables 3.1 and 3.2, as well as all the remaining optimization results discussed in this chapter, have been calculated by using roadside timber prices. The harvesting costs have been calculated with the models of Rummukainen et al. (1995), which take into account, among other things, the forwarding distance, size of harvested trees, and harvested volume per 100 meters of extraction road. Typically, the harvesting cost per cubic meter in steady-state CCF is about the same as in a normal thinning of a mature even-aged stand. Harvesting is more costly than in clear felling, but cheaper than in the first commercial thinning of even-aged management. When taking the whole rotation, there is no major difference between the harvesting costs of CCF and RFM (Pukkala et al. 2011a).

As expected, increasing discount rates lead to decreasing density and stumpage value of the post-cutting residual growing stock (Fig. 3.7). However, due to the requirement for constant regeneration and ingrowth, the basal area of the post-cutting residual stand is never more than 9 m^2 ha^{-1} when NPV is maximized. The removal of pulpwood-sized trees (dbh $<$ 19 cm) is low even with high discounting rates since the relative value increment of these trees is high. Due to these reasons, the post-thinning basal area is not particularly sensitive to the discount rate.

If saw log production or net income is maximized, instead of NPV, the post-cutting stand will have more large trees (Fig. 3.8). However, maximal saw-log production greatly decreases economic profitability (Schulte et al. 1999; Pukkala

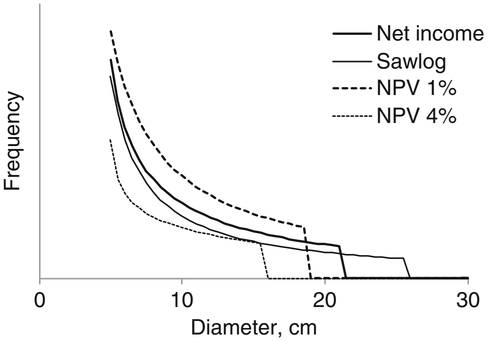


Fig. 3.8 Optimal post-cutting diameter distribution in a herb-rich spruce stand in southern Finland when the objective variable is net income, saw log production, or net present value with 1% or 4% discount rate

et al. 2010). On the other hand, near maximal saw-log yields can be obtained with only 10% decrease in NPV (Pukkala et al. 2011a), which means that fairly good saw-log production and fairly good profitability can be attained simultaneously.

The optimal steady-state stands lack large trees. Therefore, the stand structures may not be the best possible for multiple-use forests. To find management options that are more suitable for recreation, we have also optimized the post-thinning residual stand structure with a penalty for lack of large trees. In these calculations the solution was penalized if the post-thinning stand contained too few large trees. Table 3.3 shows that the requirement for large trees in the post-cutting stand decreases both economic profitability and yield. However, since economic profit is not the only objective in multiple-use or recreational forests, these constrained schedules may be the preferred ones especially in regions where nature tourism is important. It is also possible that the value of large trees increases in the near future, since timber is again favored in construction. Large trees also have the smallest share of pulpwood, and especially spruce pulpwood has a declining price.

When the NPV of the optimal uneven-aged steady-state management has been compared to optimal even-aged management, uneven-aged management has been more profitable in almost all calculations (Pukkala et al. 2010; Tahvonen et al. 2010). The two management systems have been found equally profitable on the best growing sites of South Finland with a low discount rate (1%). In all other cases, uneven-aged management has been more profitable. The relative advantage of uneven-aged management increases with decreasing site quality and timber price, and with increasing discount rate and management costs (Fig. 3.9). The net present value of even-aged management is negative already with a 5% discounting rate (Hyytiäinen and Tahvonen 2003) whereas the optimal uneven-aged management has a positive NPV with all discount rates.

Compared to the currently recommended even-aged management, the optimal even-aged management would have the first commercial thinning later, and the thinnings would be high thinnings instead of the prevailing practice of thinning from below (low thinning). The stand basal area would be gradually decreased towards the end of the rotation, which is opposite to the current official recommendations, which suggest increasing the stand basal area with increasing stand age.

Table 3.3 Effect of the requirement for maintaining large trees in the post-cutting stand on the structure, profitability and yield of a spruce stand in South Finland and a pine stand in North Finland

	Spruce on herb rich site, South Finland			Pine on sub-xeric site, North Finland		
Variable	NPV	Large trees	Very large trees	NPV	Large trees	Very large trees
Cutting cycle, a	15	15	15	30	30	30
Thinned at, $m^2\ ha^{-1}$	21	20	20	19	18	18
Remaining BA, $m^2\ ha^{-1}$	9	10	13	7	8	9.5
Remaining volume, $m^3\ ha^{-1}$	59	64	92	42	51	72
Remaining no. of trees	1,612	1,757	2,119	1,457	1,398	1,164
• 0–5 cm	784	929	1,154	751	764	648
• 5–10 cm	417	506	580	357	330	275
• 10–15 cm	248	198	204	237	180	100
• 15–20 cm	131	63	90	112	73	61
• 20–25 cm	28	38	42	–	29	34
• 25–30 cm	–	14	20	–	13	20
• 30–35 cm	–	9	16	–	8	12
• 35–40 cm	–	–	8	–	–	8
• 40–45 cm	–	–	4	–	–	5
• >45 cm	–	–	1	–	–	1
Yield, $m^3\ ha^{-1}\ a^{-1}$	6.6	5.8	4.5	3.0	2.5	2.1
Saw log, $m^3\ ha^{-1}\ a^{-1}$	4.1	3.3	3.1	1.8	1.3	0.7
Net income, € $ha^{-1}\ a^{-1}$	269	221	190	106	47	46
NPV (2%), € ha^{-1}	10,225	7,805	5,422	3,348	552	−442

"Large trees" means that the post thinning stand must contain at least 20 trees/ha of 20–30 cm in dbh and at least 5 trees larger than 30 cm. "Very large trees" means that there must be at least 20 trees/ha of 30–40 cm and at least 5 trees larger than 40 cm

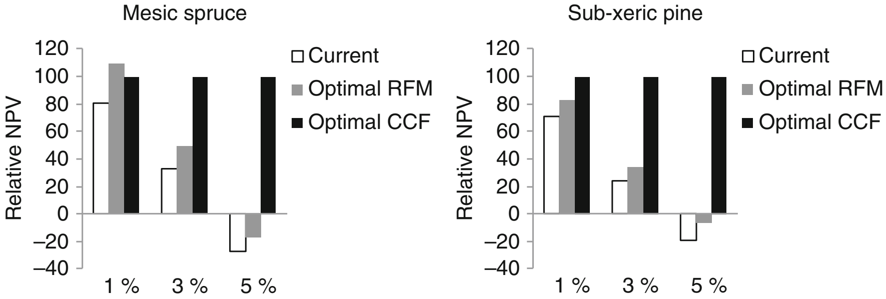


Fig. 3.9 Relative net present value (optimal CCF = 100) of the current even-aged management system (clear-felling, planting, low thinning), optimal even-aged plantation forestry (optimal RFM) (clear-felling, planting, optimal high thinning) and optimal uneven-aged steady-state management (optimal CCF) in Central Finland in a spruce stand on mesic site (*left*) and pine stand on sub-xeric site (*right*) with 1%, 3% and 5% discount rates

The only way to guarantee positive net present values with high discount rates is to minimize costs, i.e., to reduce the intensity of forest management. This is possible also in even-aged management. The NPV of even-aged management is positive, irrespective of discount rate, if the clear-felled area is left to regenerate naturally, without site preparation, sowing, planting and tending operations, and timber is sold only when the stumpage price is positive, i.e., when the road side price is higher than the harvesting cost. In practice, this kind of extensive management would mean natural regeneration with the seed tree or shelter wood methods or using small canopy gaps. Because these methods may be regarded as variants of continuous cover forestry, it may be concluded that if the forest landowner wants to practice forestry that is profitable (without state subsidies) also with high discount rates, on poor growing sites and in northern parts of Finland, her only option is continuous cover management.

4.3 Species Composition in Multi-species Stands

The steady-state structure of a mixed stand with more than one tree species can be optimized by finding the optimal post-thinning diameter distributions separately for the different tree species of the stand. The cuttings return the species-specific frequencies of diameter classes to their optimal initial levels. The solution is penalized if the post-cutting frequency of any diameter class of any species is less than the initial frequency. In the problem formulation adopted in this chapter, the decision variables of the optimization problem for mixed steady-state stands are the species-specific values of Weibull parameters b and c, number of trees per hectare, and maximum diameter retained in thinning.

Realistic species mixtures on the most fertile growing sites of Finland (herb rich, OMT) are those of birch and spruce. On medium sites (mesic site, MT), all mixtures of the main tree species (spruce, pine and birch) are possible. Pine is the best-growing species on sub-xeric site (VT), but pine-spruce mixtures are also feasible since spruce frequently enters sub-xeric pine forests as an under-storey and attains reasonable dimensions. On the very poorest sites, pine is the only economically meaningful option.

Spruce enters birch forests on fertile sites (Photo: Olavi Laiho)

According to the steady-state optimizations for mixed stands, the best species composition on the best growing sites is a pure spruce stand if economic profitability is maximized, even though a small admixture of birch would increase yield. This result is mainly due to the fact that the price of spruce pulpwood is clearly better than that of birch pulpwood. However, increased use of wood as biofuel, and the decreasing demand for spruce pulpwood for mechanical mass may alter the situation making birch mixtures more profitable. An admixture of birch and other hardwoods would also improve the scenic value (Silvennoinen et al. 2001) and structural diversity (Lähde et al. 1999a) of the stand, and enhance regeneration. Therefore, maintaining an admixture of hardwood seems to be worthwhile on the best sites.

On medium sites the most profitable two-species combination is a mixed stand of spruce and pine, spruce being the dominant species. A mixed stand is slightly

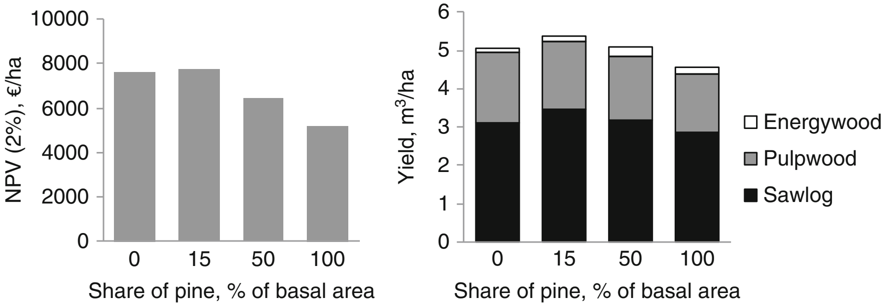
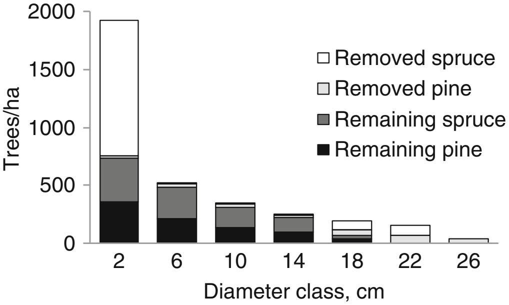


Fig. 3.10 Net present value (*left*) and sustained annual production (*right*) of a pine-spruce stand on mesic site in Central Finland when NPV is maximized with 2% discount rate, and the share of pine in the post-thinning stand is 0%, optimal (15%), 50% or 100% of basal area

Fig. 3.11 Optimal steady-state structure and management of pine and spruce mixture on sub-xeric site in Central Finland with 20-year cutting cycle when NPV with 2% discount rate is maximized



more productive and more profitable than a pure spruce stand (Fig. 3.10). Similarly as on the best growing sites, some birches should be maintained since they enhance regeneration, amenity values, and diversity.

The optimal structure of a sub-xeric site is a pine-spruce mixture, in which pines are larger than spruces (Fig. 3.11). The stand therefore looks like a two-layered stand where the spruce layer consists of smaller trees than the pine layer. However, the pine and spruce layers overlap. Many current forests already show this type of structure, except that there are too few small pines compared to the optimal uneven-aged structure. In many cases, the lower spruce storey is denser than the optimal post-thinning distribution suggests. The challenge of sustainable steady-stage management of a mixed conifer stand is therefore to maintain the regeneration and ingrowth of pine. This can be done by reducing the stand basal area to a lower level than currently used, and thinning the lower storey of spruce. Creating small gaps or very sparse areas may also be necessary to promote the regeneration of pine and hardwoods. Another question is how meaningful it is to maintain a mixed stand in a steady state. It might be easier and more profitable to alternate between periods of pine and spruce dominance on medium sites, and between hardwood and spruce dominance on fertile sites.

Spruce enters pine stands on medium sites (Photo: Timo Pukkala)

4.4 *Optimizing Transformation Cuttings*

Another optimization problem is the conversion of an even-aged stand, or any existing stand, to the optimal steady state structure. As explained earlier in this chapter and in the other chapters of the book, this can be done in two different ways. One is to optimize the steady state diameter distribution simultaneously with the transformation cuttings. A slightly easier way is to optimize a certain number of transformation cuttings with a constraint that the next cutting (after the optimized ones) converts the stand into a predefined steady-state structure. In both cases, the optimized variables are the diameter distributions of several successive post-thinning stands; a sequence of diameter distributions is optimized. The time points of cuttings can be fixed or they can be optimized simultaneously with the distributions.

This section presents optimization results for fertile spruce stands in which the number of transformation cuttings is four and their interval is 15 years (cutting years are 0, 15, 30 and 45). The fifth cutting, after 60 years, converts the stand into a predefined steady state structure. The optimized variables are the post-thinning diameter distributions of the first four cuttings, and each distribution is described by the same variables (*bW*, *cW*, *N* and *Dmax*). The solution is not penalized if there are too few remaining trees in some diameter classes after the four transformation cuttings. However, shortage is not allowed in the fifth cutting, since the treatment from that moment onwards should represent sustainable steady-state management. Because of this, the optimization in fact consists of finding a sequence of optimal cutting limits for the diameter classes; all excessive trees, as compared to the cutting limit, are removed.

In a mature even-aged spruce stand the optimal sequence of cuttings consists of an immediate shelter tree cut, removal of the shelter trees gradually in two cuttings after 15 and 30 years, and using normal uneven-aged management thereafter (Fig. 3.12, left). It is noteworthy that it is not optimal to manage the mature stand with repeated light high thinnings. Instead, it is more profitable to regenerate the stand first. Another important observation is that shelter trees should be selected from among the smallest trees of the stand. These trees have the best relative value increment and the lowest opportunity cost when left to continue growing. Dominated and co-dominant trees also have narrow crowns, and their removal therefore causes less damage than the removal of large trees with wide crowns.

In a two-layered stand the optimal management involves removing the over-storey in one or several cuttings (in two cuttings in the example shown in Fig. 3.12). The removal should be started with the largest trees, and pulp-wood sized trees should be left to continue growing until they are log-sized. By the time when all over-storey trees have been removed, the diameter distribution of the under-storey has already reached a near-optimal post-thinning distribution of a steady-state stand.

In young even-aged stands it is optimal to wait until a fair removal of log-sized trees can be obtained. From this moment onwards, the stand is managed with high thinnings that soon convert the stand into the optimal steady state structure.

Also Tahvonen et al. (2010) optimized the future management of different initial stands. The optimization problem was solved in its most general dynamic form. When NPV was maximized, uneven-aged management was the choice in almost all cases. The only exception was a mature even-aged stand with a low discount rate. In this stand, which had passed the stage of financial maturity, the optimal management involved immediate clear cutting and regeneration. However, the next tree generation should be managed as an uneven-aged forest.

4.5 Temporal Variation of Harvests

Some studies (e.g., Haight et al. 1985) suggest that the profitability of uneven-aged management may improve if the stand density is varied in successive cuttings. There may be a very strong cutting every now and then, resulting in good regeneration, followed by a period of light cuttings with small removals. Then the normal cutting level is used, followed again by a strong regenerative cutting. This kind of periodical uneven-aged management may be optimized by replacing the single steady-state post-thinning diameter distribution by a series of successive distributions, which

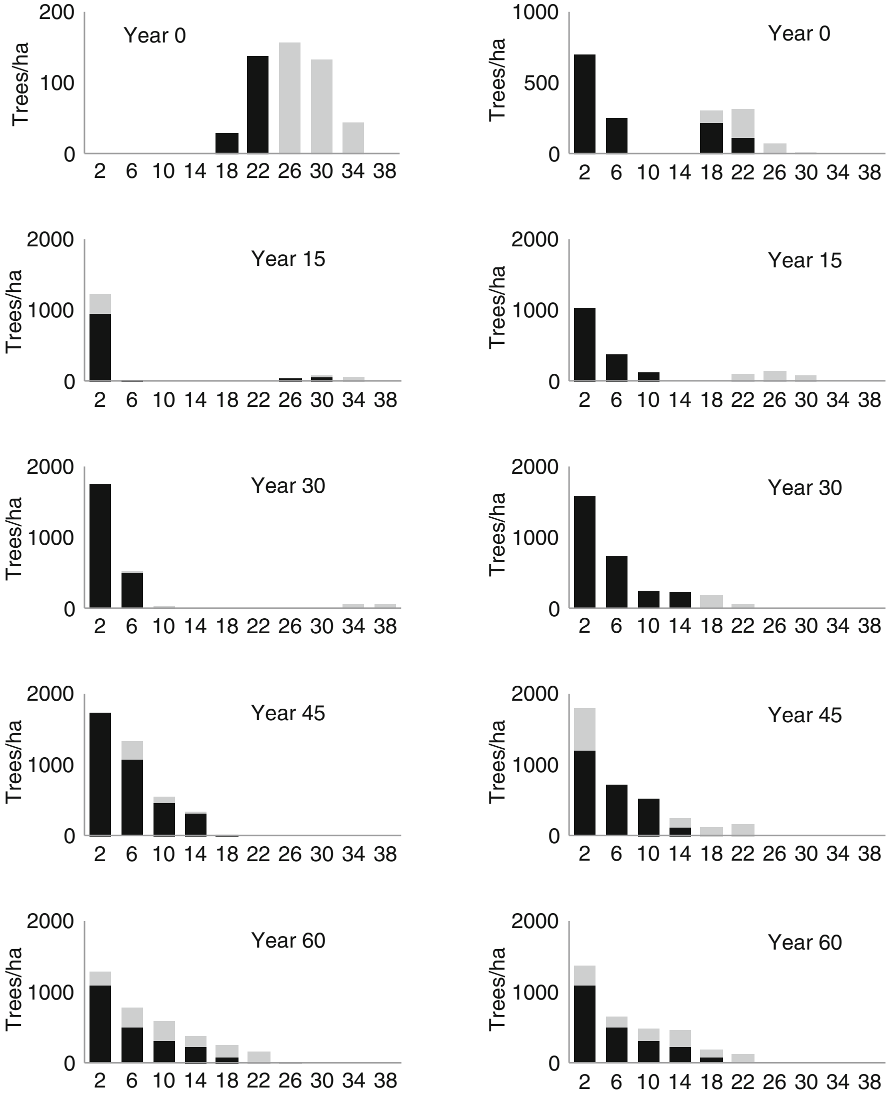


Fig. 3.12 Optimal path of transformation cuttings on a herb-rich site in a mature even-aged spruce stand (*left*) and in two-layered spruce stand (*right*) when the optimal steady-state diameter distribution must be reached in the 5th cutting in year 60. *Black*: residual, *grey*: removed trees

are repeated with a longer cycle. For example, if the normal cutting cycle is 20 years, four successive distributions may be optimized, representing the remaining frequencies of diameter classes in the first, second, third and fourth thinning, conducted after 0, 20, 40 and 60 years. The fifth thinning, after 80 years, returns the first diameter distribution and begins a new 80-year cycle. The initial investment

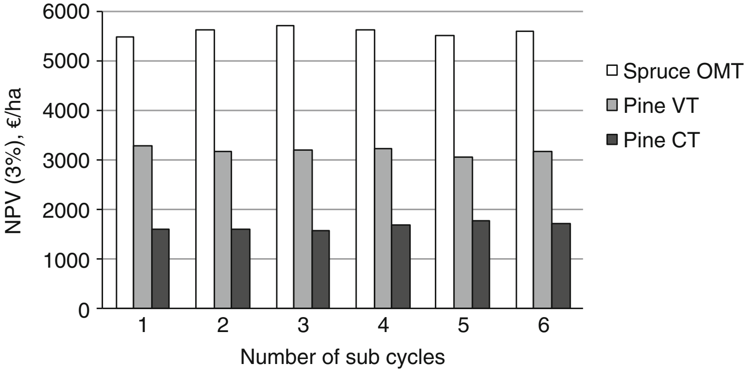


Fig. 3.13 Net present value in the optimal solution for three growing sites when a sequence of 1–6 post-thinning diameter distributions is optimized. Calculation of the NPV assumes that the cycle of 1–6 diameter distributions is repeated to infinity

(A_T in Eq. 3.1) is equal to the value of the first post-thinning stand. The net incomes from the entire 80-year period need to be calculated and discounted to the first year to obtain the periodical NPV, which is converted to the NPV of an infinite series of 80-year periods.

To analyze the effect of temporal variation in stand structure, we replaced the single post-thinning diameter distribution by a sequence of 2–6 distributions. The analyses were done for South Finland for a herb-rich spruce forest (OMT), sub-xeric pine forest (VT), and xeric pine forest (CT). The cutting cycle was 15 years in OMT, 20 years in VT, and 30 years in CT. These cutting cycles result in high enough cutting removals, 70–110 m^3 ha^{-1}.

Optimizing a sequence of distributions improved the NPV of the spruce stand on the herb-rich site by 2–4% (Fig. 3.13). However, differences between successive post-thinning diameter distributions were small. In VT pine, replacing the single post-thinning diameter distribution by a sequence of 2–6 distributions never improved profitability and all distributions were almost similar. The result means that the optimal management does not require temporal variation in post-thinning stand structure.

The result was different for CT pine, where temporal variation clearly increased the NPV. The increase was the highest, 5–12% when a sequence of 4, 5 or 6 distributions was optimized. All these optimizations involved cutting the stand to a very low basal area of 1.5–3 m^2 ha^{-1} once during a 120–180-year period (Fig. 3.14). All trees larger than 12 cm in dbh were removed in these cuttings and almost all trees 8–12 cm in dbh were also removed. The remaining stand after this cutting was a young uneven-aged seedling-sapling stand. Optimization resulted in a schedule which is between strictly even-aged and strictly uneven-aged management.

In this stand, the NPV was the highest when a sequence of five diameter distributions was optimized. In the optimal schedule, the post-cutting stand of year

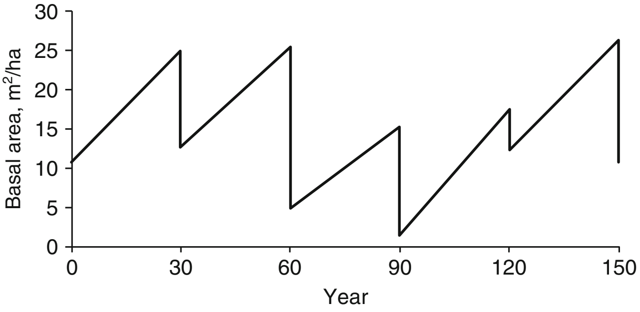


Fig. 3.14 Optimal management of a pine stand on a xeric site in southern Finland when the cutting interval is 30 years. The optimal management involves a sequence of 5 post-thinning diameter distributions, which are repeated with a 150-year cycle. The post-thinning stand in year 150 is similar to the initial stand in year 0

90 was a young stand without any large trees. This stand was uniformly and lightly thinned 30-years later, after which three consecutive high thinnings were done (years 150, 30 and 60 in Fig. 3.15).

The result may be termed as any-aged management according to Haight and Monserud (1990a, b). However, it is still continuous cover management since the site is continuously covered by trees. The optimal solution for CT pine suggests that it is not always enough to optimize a single post-cutting diameter distribution but options allowing temporal variation in stand structure should also be analysed. This is evident when the cuttings of a certain, non-optimal initial stand are optimized (Tahvonen 2009; Tahvonen et al. 2010). However, also the steady state management may involve a long cycle consisting of shorter and unequal sub-cycles.

This kind of CCF management was optimal in only one out of three cases and maximally 12% better than the optimal one-cycle steady-state management. The conclusion may change if species dynamics and mixed stands were analysed. For example, the optimal management of a fertile site may involve a heavy regeneration cut in a spruce stand so as to increase the regeneration of hardwoods. Then, the largest spruces are removed, the residual stand being a young rather even-aged hardwood stand where spruce begins to enter again as under-storey. The next cuttings gradually decrease the share of hardwoods and convert the stand into a spruce stand. Because regeneration often gets more difficult with increasing spruce dominance, another very strong cutting may be required, which initiates a new hardwood-dominated period.

4.6 Variable-Density Thinning as a Means to Increase Non-timber Benefits

The economically optimal management of an uneven-aged Norway spruce stand growing on a fertile site in South Finland involves cutting all trees larger than 20 cm

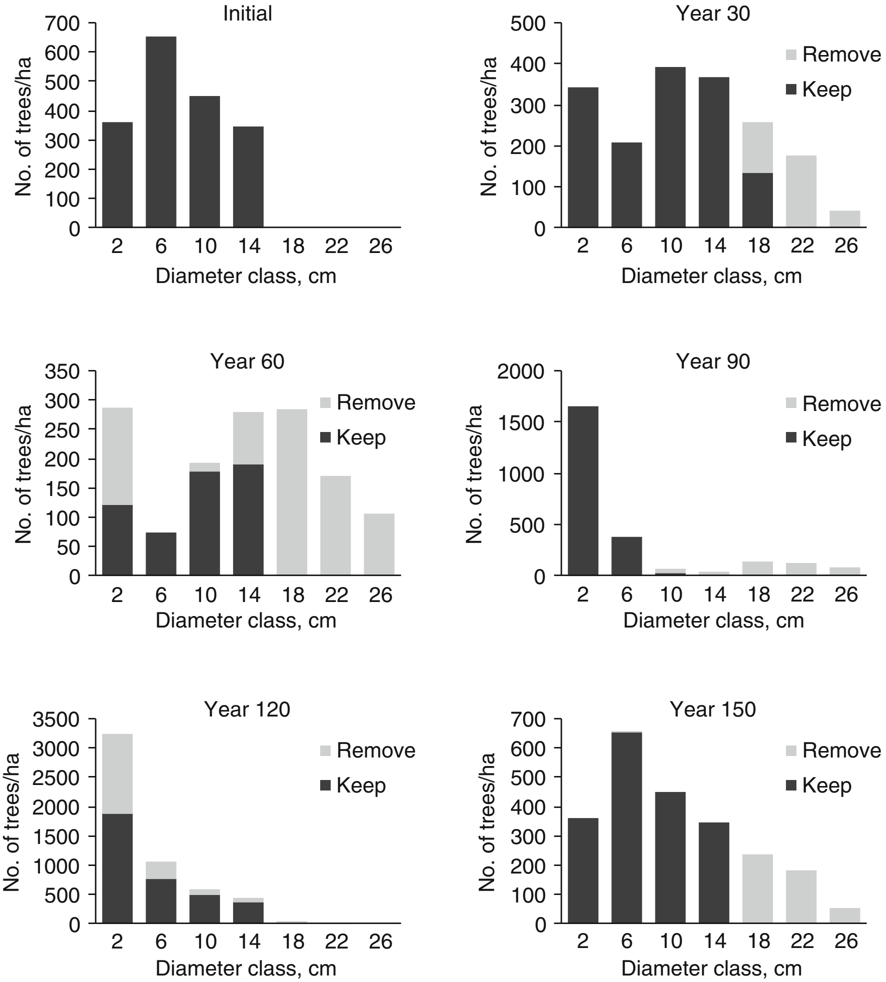


Fig. 3.15 Optimal sequence of diameter distributions in a pine forest growing on a xeric site in South Finland

at 15-year intervals. The newly harvested stand resembles a young sparse forest. The recreationist walking in the forest does not feel like being in a mature forest. The scenic value of the post-cutting forest is not particularly high due to the absence of large trees. The amount of epiphytic lichens, growing on the branches of trees is most probably low due to absence of old trees since it takes time from the lichens to develop (e.g., Johansson 2008). Together with wild berries and mushrooms, lichen constitutes an important food and nest source for several animal species.

One possibility to avoid too sparse post-cutting stands is to divide the uneven-aged forests into segments and harvest only one segment at a time. The densest places with the highest stand volume or greatest number of large trees are harvested first, the second densest places in the next cutting, and so on. A stand having an optimal cutting interval of 20 years may be divided into four segments, each of which is treated at 20-year intervals. Thus, the stand will be visited at 5-year intervals. With two segments, the stand needs to be visited at 10-year intervals and with tree segments at 6–7-year intervals.

This kind of management results in spatially heterogeneous stands. The temporal continuity of large trees is better since a newly cut stand has large trees in the uncut segments. This is an advantage for both scenic amenity and biodiversity maintenance. Many real stands already are spatially heterogeneous with dense and sparse places, and it is logical to restrict the cutting to the densest places. If the post-thinning density of the treated places is lower than the stand density of the non-treated places, it is equally logical to skip the newly thinned places in the next cutting. In this way, the spatial heterogeneity will be maintained in the future.

When using cutting segments the removals are concentrated on smaller areas with a consequence that the visual impact of cutting and the rate of logging injuries may be smaller than when treating the whole stand simultaneously. If the optimal interval between the cuttings of the same place is longer than it would be without segmentation, the removal per hectare and the mean size of harvested trees will be larger, too. This will result in cheaper harvesting. The optimal post-thinning density of the zone may be lower than the optimal density without segmentation, which may have a favourable effect on regeneration. On the other hand, the stand needs to be visited more often, which increases the harvesting cost.

This section explains a simple way of optimizing the management of a segmented stand, in which the steady-state management is optimized using only one post-tinning diameter distribution (per species). The distribution is defined by the same variables as in previous examples (*N*, *bW*, *cW* and *Dmax*). Therefore, the number of optimized variables is the same as in a non-segmented forest, but simulation is slightly more complicated. With three cutting segments and 15-year cutting cycle, the simulation is now done as follows:

1. Using *N*, *bW*, *cW*, and *Dmax*, generate the post-thinning stand for cutting segment 1
2. Simulate the development of segment 1 for 5 years
3. Generate the post-thinning stand for segment 2
4. Simulate the development of segment 1 for 5 years (10 years from beginning)
5. Simulate the development of segment 2 for 5 years (5 years from beginning)
6. Generate the post-thinning stand for segment 3
7. Simulate the development of segment 1 for 5 years (15 years from beginning)
8. Simulate the development of segment 2 for 5 years (10 years from beginning)
9. Simulate the development of segment 3 for 5 years (5 years from beginning)
10. Reduce the frequencies of diameter classes of segment 1 to their initial values; calculate the per hectare removal, roadside value of removed trees, harvesting costs, and net income of cutting: divide the net income by 3 to obtain the per hectare net income for the entire stand
11. Calculate the value of the post-thinning stand using all three segments
12. Calculate the NPV of the schedule

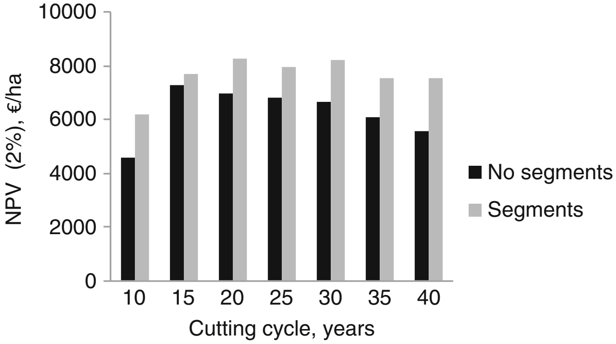


Fig. 3.16 Net present value (NPV) of the optimal management schedule with the requirement of having large trees continuously present in the stand

The post-thinning structure of the entire forest consisting of three segments is found in Step 6 after simulating the development of an initial diameter distribution for 0, 5 and 10 years. Another 5-year time step produces the pre-thinning structure of the entire forest (Step 9). Reducing the frequencies of diameter classes to their initial levels in the densest segment gives the harvest removal and periodical net income, and returns the forest into a post-thinning state (Step 10).

This type of simulation assumes that the stand dynamics of a segment do not depend on adjacent segments; the simulation is "distance-independent" although the management of spatially heterogeneous forest is optimized.

We have optimized the management of segmented spruce stands growing on a fertile herb-rich site in Central Finland with several combinations of segments and cutting cycles (Pukkala et al. 2011d). All optimizations have been done also by using a modified objective function, which includes a penalty if the post-thinning stand includes too few large trees; if the number of trees with dbh between 20 and 30 cm was less than 10 ha^{-1} a penalty of 10,000 € ha^{-1} was paid in the cutting year. Another 10,000 € ha^{-1} was paid if the number of trees with dbh $>$ 30 cm was less than 5 ha^{-1}. The post-thinning number of large trees was counted within the whole stand (not only in the thinned segment).

According to the optimizations, segmentation decreased net present values when the cutting cycle was short. The cost of visiting the stand more often was higher than the benefits obtained from segmentation. However, when lack of large trees was penalized spatially heterogeneous stand was always more profitable (Fig. 3.16). The optimal cutting cycle of a segment became longer when the penalty for lack of large trees was added to the objective function, 20–30-year cutting cycles being optimal instead of the 15-year cycle which was optimal without the penalty.

The requirement for a continuous presence of large trees always reduced profitability. Without segmentation the decrease was 20–30% depending on the cutting cycle. However, segmentation greatly reduced the cost of maintaining large trees; with cutting cycles of 20 years or more, the reduction was less than 5%.

These optimizations suggest that, when economic profitability should be combined with non-timber services, segmentation with spatially heterogeneous management and stand structure should be pursued. Even without the requirement for large trees, segmentation leads to wider post-thinning diameter distributions, continuous presence of larger trees, and smaller temporal variation in the forest. In addition, a spatially heterogeneous stand may be experienced more interesting by people who visit the forest.

Segmentation is not much different from having smaller stands. However, if the segments are small enough or narrow enough, they may be impractical as stands. In the calculations described above, it was assumed that the reason behind segmentation is reduced temporal variation in stand characteristics and better continuity of large standing trees, so as to improve diversity and amenity. When designing the width and size of the zones, the essential thing is how people and animal species experience the cut and un-cut segments. If the segments are so narrow that large trees in the uncut segments are perceived as belonging to the same stand as the cut places, the use of neighbouring uncut segments as providers of large trees is justified. If people or animal species experience the segmented forest as a single mature uneven-aged forest, segmentation is beneficial. For practical reasons, the minimum width of the cutting segment should be equal to the distance between adjacent extraction roads, i.e., 20–30 m. If the narrow cutting segments are not straight, it is unlikely that they are perceived as distinct stands, especially if the thinning intensity within a cutting zone is not uniform. To minimize harvesting injuries, the best option is to harvest most near the extraction road and reduce the thinning intensity gradually with increasing distance from the strip road. Trees that are difficult to fell without causing logging injuries could be left as retention trees.

Another advantage of stand heterogeneity is high flexibility in marketing; the owner can harvest when the price is good but she is not forced to do that. In a uniform stand, especially in even-aged forestry, management is more tightly related to the stage of stand development; timber sales are not possible during some stages whereas cutting is almost obligatory at another stages. This reduces the possibilities to adjust timber sales according to the market situation.

5 Multifunctional Management

This section analyzes the economic performance of alternative management systems when non-timber benefits are added to the landowner's objective function. The non-timber benefits are wild berries and carbon sequestration. Biodiversity maintenance and scenic beauty are also briefly discussed.

5.1 Methods

In the optimization of multifunctional management the objective function may be a formula that calculates the net present value obtained from several products and services (Faustmann or Hartman formula). Another possibility is to use a utility function. A third option is to employ Hartman formula or utility function augmented with a penalty function. It is also possible to have NPV as a component in a utility function.

If the non-timber products and services can be expressed in monetary units it is logical to convert all benefits into net incomes and discount them to the present. Examples of products and services that can be measured in monetary units are berry yields, mushroom yields and carbon sequestration. We have analysed the optimal management and profitability of even- and uneven-aged management of pine and spruce stands when timber, bilberry (*Vaccinium myrtillus* L.) and carbon benefits are all expressed in monetary units and included in the analysis (Pukkala et al. 2011c). Both of the stands were growing in Central Finland, spruce on mesic site and pine on sub-xeric site.

The discounted bilberry and carbon benefits were first calculated for a cutting cycle as follows:

$$NPV_{\mathrm{Cycle}} = \sum_{t=1}^{T} \frac{B_t + C_t}{(1+i)^t}$$

where B_t is the value of bilberry harvest in year t, and C_t is the value of carbon sequestration in year t. If the carbon balance is negative, C_t is also negative (carbon release tax is paid). The NPV of a cycle was then converted into the NPV of an infinite series of cutting cycles:

$$NPV_{\mathrm{Berry+Carbon}} = NPV_{\mathrm{Cycle}} \left[1 - \frac{1}{(1+i)^T} \right]^{-1}$$

The total NPV of uneven-aged management was computed from

$$NPV = NPV_{\mathrm{Timber}} + NPV_{\mathrm{Berry+Carbon}}$$

with

$$NPV_{\mathrm{Timber}} = \frac{N_T}{(1+i)^T - 1} - A_T,$$

where N_T is the net income obtained regularly at T-year intervals (after T, $2T$, $3T, \ldots$ years), T is the cutting cycle, i is discount rate and A_T is the stumpage value of the post-thinning stand.

The empirical bilberry yield models of Miina et al. (2009) modified for uneven-aged stands (Pukkala et al. 2011c) were used to predict the bilberry production along with stand development. The models first predict the coverage of bilberry and then the annual berry yield as a function of stand characteristics and bilberry coverage. The bilberry yield predictions were multiplied by 0.75 assuming that 75% of the total yield of the season is actually harvested (Raatikainen and Niemelä 1983). Bilberry price was taken as 3 € kg^{-1}. In 2008 the average market price paid to pickers in eastern Finland was 1.9 € kg^{-1} for non-cleaned and 4.6 € kg^{-1} for cleaned bilberries (Miina et al. 2010).

The annual carbon balance was calculated based on Romero et al. (1998) and Díaz-Balteiro and Romero (2003):

$$C = a\,(\Delta B + H + R + D) - HR - PR + S - a\,(\Delta P + \Delta R + \Delta D)$$

where C is the annual carbon balance, a is carbon content of biomass (proportion of carbon of dry mass), ΔB is the annual change in biomass, H is the biomass of annual harvest, R is the biomass of new cutting residues, D is the biomass of annual mortality, HR are harvesting releases, PR are processing releases, S are substitution effects, ΔP is the decomposition (decrease of dry mass) of earlier products, ΔR is the decomposition of earlier residues, and ΔD is the decomposition of earlier deadwood. All biomasses are in tons of dry mass per hectare and C, HR, PR and S are in tons of carbon per hectare. Stumps and roots were included in the biomass of living trees, cutting residues and dead trees.

Decomposition of different components of dead trees, cutting residues, and harvested trees was simulated using

$$B_t = B_0 \mathrm{e}^{-kt}$$

where B_t is the remaining dry mass after t years, B_0 is the initial dry mass and k is the annual decomposition rate. Parameter k values for branches and needles were based on literature (e.g., Hyvönen et al. 2000). For stems and stumps (including roots), the annual decomposition rate depended on tree species and breast height diameter of the tree (Pukkala 2006).

The sawlog part of each cut tree was divided into four end product classes: long-term product (sawn wood and plywood), mechanical mass, chemical mass and biofuel (e.g., Karjalainen et al. 1994; Liski et al. 2001). The pulpwood part was divided into three classes: mechanical mass, chemical mass and biofuel. The product classes, their proportions and decomposition rates were adapted from Karjalainen et al. (1994) and Liski et al. (2001). The decomposition rates of products were based on the estimated time-spans during which half of the mass is decomposed (Liski et al. 2001). This time was taken as 50 years for sawn wood and plywood, 3 years for mechanical and chemical mass, and 1 year for biofuel.

Harvesting emissions were calculated from

$$HR\% = 3.0 - 0.05Dq$$

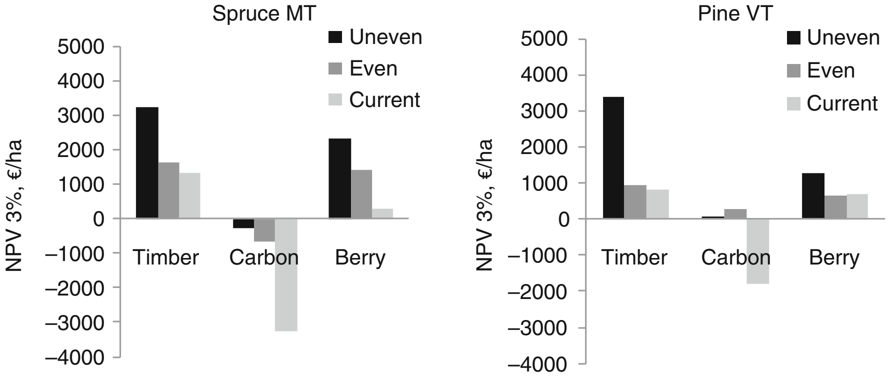


Fig. 3.17 Net present value of different benefits in the optimal uneven-aged, optimal even-aged, and current even-aged management. Discount rate is 3%, bilberry price is 3 € kg^{-1}, and carbon dioxide price is 15 € ton^{-1}. In the pine stand, the NPV of carbon benefits is close to zero in the optimal uneven-aged management

where *HR%* is harvesting release (released carbon in percent of carbon contained in harvested wood) and *Dq* is the quadratic mean diameter of removed trees (cm). Biofuel collection rate was taken as 0.67 (2/3 of the biomass of branches is collected and 1/3 remains in the forest) which corresponds to the current efficiency. Stumps and roots were not collected for biofuel. The price of carbon dioxide (tax or subsidy) was assumed to be 15 € ton^{-1} (http://www.carbonpoint.com), which corresponds to a carbon price of 55 € ton^{-1}.

The decomposition simulator was initialized with near optimal steady-state amounts of decomposing materials in the beginning of a rotation or cutting cycle. The near optimal values were obtained from preliminary optimizations. To guarantee that the carbon dynamics represented a steady state, three rotations were always simulated in even-aged management and four cutting cycles were simulated in uneven-aged management. The annual carbon balances were calculated from the last rotation or cycle.

5.2 Results

The optimal uneven-aged management of the spruce stand yielded the highest NPV of timber benefits, the highest NPV of carbon benefits, and the highest NPV of bilberry benefits (Fig. 3.17). In the pine stand, the optimal even-aged management was better in terms on carbon benefits but uneven-aged management was clearly better in terms of timber and bilberry benefits. The currently recommended even-aged management was much worse than the optimized systems. The main differences between the current and optimal even-aged management systems are that the optimal management postpones the first commercial thinning, uses longer rotation lengths, and applies high thinning instead of the currently favoured low thinning. Summing the net present values of timber production, bilberry collection, and carbon sequestration reveals that the optimal uneven-aged management was by far the best in both species, and the current even-aged management system was by far the worst.

The low net present value of even-aged management, as compared to uneven-aged management, is partly explained by the unfavourable temporal distribution of costs and incomes. Even-aged management has early stand establishment and tending costs but the first incomes are obtained only after 45–50 years since planting. The first incomes of uneven-aged management are obtained after one cutting cycle, i.e., after 20 years. The optimal rotation lengths of the multifunctional forestry were 15–25 years longer than currently recommended or obtained when only timber production is optimized. Therefore, considering carbon sequestration and bilberry production in forestry decision-making leads to the use of longer rotations.

The timber yields were of the same magnitude in all three management systems (Fig. 3.18). The current even-aged management system yielded most timber in the spruce stand, and the optimal uneven-aged management was the most productive in pine. In the spruce stand, the saw log yields were nearly the same in all three management systems but pulpwood and biofuel harvests were higher in the currently recommended even-aged management schedule. Optimal uneven-aged management was the best in terms of carbon sequestration and bilberry yield, in both spruce and pine (Fig. 3.18). With the used substitution rates (0.2 for sawn wood and 0.4 for biofuel), spruce forestry was always a carbon source whereas pine forestry was a carbon sink.

In spruce, the bilberry harvest was very sensitive to the management system so that the optimal uneven-aged management yielded five times more bilberries than the current even-aged management system. The density of the uneven-aged spruce stand was almost constantly near the optimal value for bilberry. Differences in bilberry harvest between the three management systems were small in pine stand.

The carbon balance of the first years of an even-aged rotation is very negative because there are plenty of decomposing cutting residues, stumps and roots, as well as products with short life spans (biofuel and pulp) from the previous clear-felling. The annual balance turns positive after about 30 years (the lines representing accumulated carbon in Fig. 3.19 start to ascend) since most of the fast-decomposing materials have already decomposed and carbon sequestration into new biomass is fast. However, due to discounting, the negative carbon balances of the first years have a much stronger influence on the NPV than the positive balances of later years.

The NPV of the carbon benefits is much better for the optimal even-aged management schedule than for the currently recommended schedule. This is because the optimal even-aged management employs heavy high thinnings during the latter half of the rotation, which decreases the clear-felling removal. As a consequence, much decomposition and carbon releases occur at the end of the rotation when the cutting residues and short-term products of the thinnings decompose, leaving less decomposing material to the first years of the next rotation.

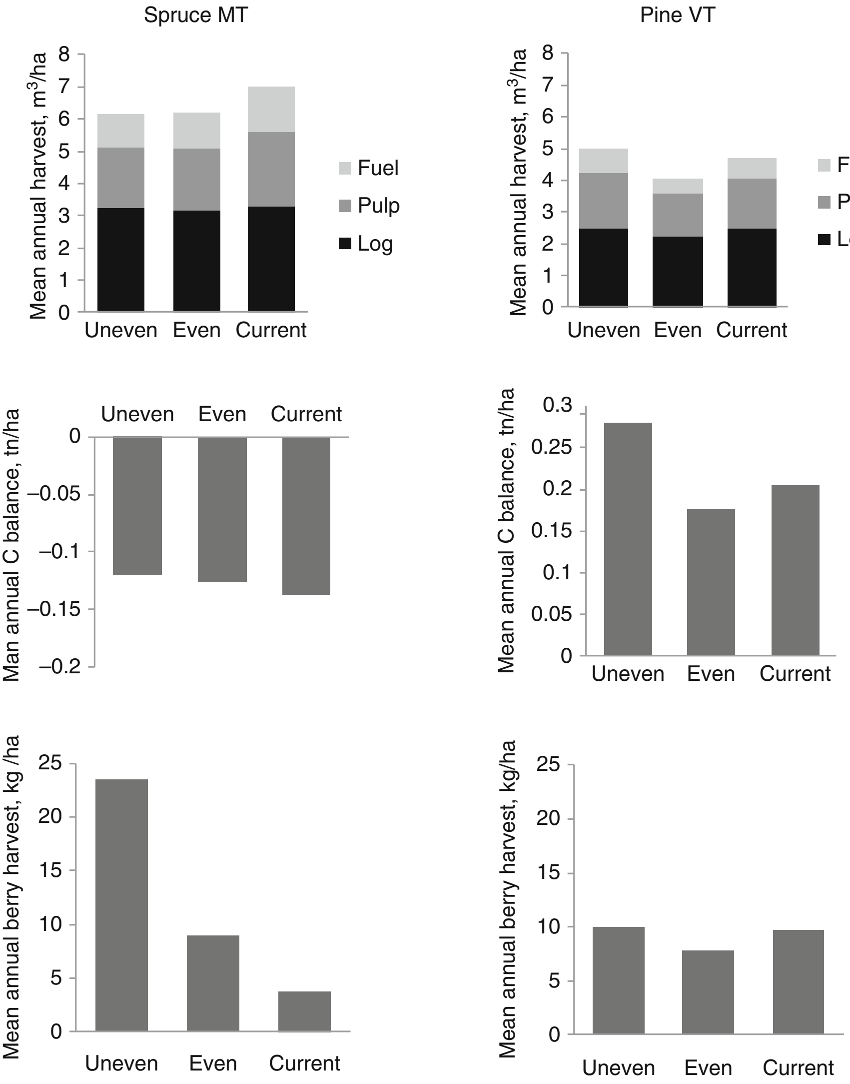


Fig. 3.18 Mean annual wood harvest, carbon balance and bilberry harvest in the optimal uneven-aged, optimal even-aged, and current even-aged management. Discount rate is 3%, bilberry price is 3 € kg^{-1}, and carbon dioxide price is 15 € ton^{-1}. The dry mass of branches collected for biofuel has been converted into cubic metre equivalents and included in the "Fuel" assortment. "Fuel" includes the tops and 2/3 of branches of cut trees

Other important services of Finnish forests are biodiversity maintenance and scenic values. The most recent models for the landscape preferences of Finns are those of Silvennoinen et al. (2001), which predict the scenic quality of a forest stand as a function of mean tree height, skewness of the diameter distribution, number

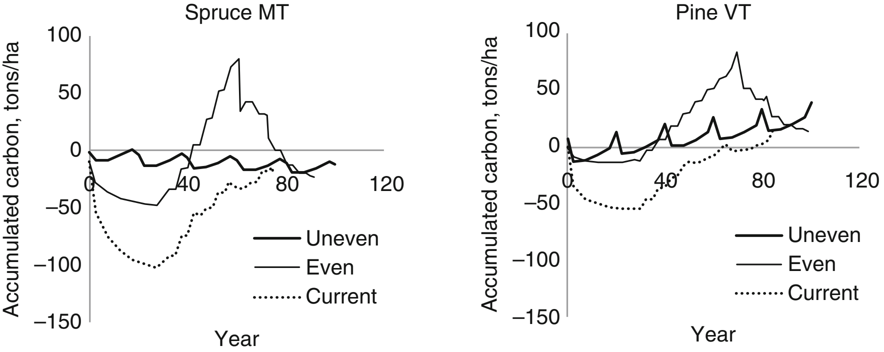


Fig. 3.19 Temporal development of accumulated carbon balance in the optimal uneven-aged, optimal even-aged, and current even-aged management. Discounting rate is 3%, bilberry price is 3 € kg^{-1}, and carbon dioxide price is 15 € ton^{-1}

of trees per hectare, volume of pine, and volume of birch. Figure 3.20 shows the scenic beauty index of Silvennoinen et al. (2001) for the stands and management systems analyzed in this section. It can be seen that uneven-aged and mature even-aged stands are close to each other in terms of scenic beauty, but a young even-aged forest is clearly inferior. Therefore, the long-term mean scenic value is better for uneven-aged management.

Several indices have been proposed for describing the diversity of a forest stand in Finland (e.g., Lähde et al. 1999b; Pukkala 2006; Koskela et al. 2007). Their idea is to measure the number of different structural elements present in the stand such as tree species, tree sizes, and types of deadwood. The indices are based on the assumption that the number of species that can live in the stand increases with an increasing number of structural elements. The LLNS index of Lähde et al. (1999b) was calculated for the three management systems compared in this section (Fig. 3.20). The results suggest that an uneven-aged stand is equal to a mature even-aged stand in terms of structural diversity, but clearly better than young even-aged stands. Taking into account that small temporal variation in the amounts of structural elements is an advantage, the results in Fig. 3.20 suggest that uneven-aged management is better than even-aged forestry also in terms of structural diversity.

The analysis suggests that, in Finnish forests, CCF is better than even-aged management for a simultaneous production of multiple benefits. It is evident that the higher is the number of forest functions included in the analysis the clearer is the superiority of CCF management. However, at the landscape level, RFM and clear-felling may be used to improve the visibility of distant vistas, and to increase the number of different habitat types. In addition, there are some forest products such as cowberry (*Vaccinium vitis-ideae* L.) and some mushrooms, which benefit from clear-felling. Therefore, forest management which is optimal at the landscape level would most probably use both even- and uneven-aged management. This is in accordance with the natural disturbance dynamics of Fennoscandian forests,

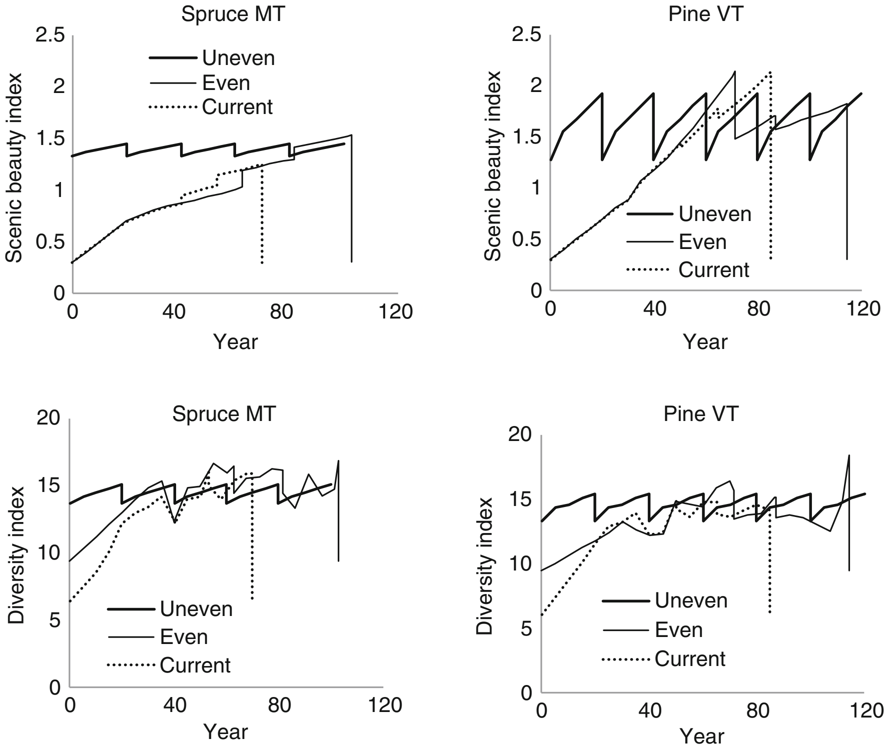


Fig. 3.20 Development of a scenic beauty index (Silvennoinen et al. 2001) and a diversity index (Lähde et al. 1999b) in the optimal uneven-aged, optimal even-aged and current even-aged management system when the total NPV from timber, bilberry and carbon benefits is maximized with 3% discount rate

which includes mainly small disturbances (trees and tree groups destroyed) with large disturbances (clear felling) occurring every now and then (Keto-Tokoi and Kuuluvainen 2010).

6 Discussion

Most analyses conducted so far in Finland and described in this chapter suggest that CCF is more profitable to the forest landowner than the currently recommended RFM (Tahvonen et al. 2010; Pukkala et al. 2010, 2011c). The relative superiority of CCF management improves with increasing discount rate and management costs, and with decreasing site productivity and timber price (Tahvonen 2009). The low profitability of RFM is mainly because of too high stand establishment cost; the low timber yields of most Finnish forests do not warrant high management costs. More extensive management with less human intervention remains the only viable option on poor growing sites, peatland sites and North Finland.

The Finnish results may appear different from some Swedish and Norwegian studies. In Sweden, Wikström (2000) obtained higher NPVs for RFM than for CCF. However, Wikström himself (2001) reminds that his results cannot be used to compare the profitability of RFM and CCF because the growth of uneven-aged stands was underestimated and ingrowth was an arbitrarily set constant. In addition, calculations were done for financially mature stands for which immediate regeneration would be optimal. However, this treatment was prevented by the constraints of the optimization problems. As a result, the uneven-aged stands were constantly too old and dense.

Andreassen and Øyen (2001) found clear felling and planting to be more profitable than selective cutting in financially mature spruce stands in Norway. Similarly to Wikström (2000), the optimal CCF-alternative, i.e. immediate natural regeneration followed by normal uneven-aged management, was not included in the analysis. In addition, the growth predictions were multiplied by 0.85 in CCF. Therefore, these Swedish and Norwegian studies only show that financially mature stands should not be treated with repeated light thinnings; they should be rejuvenated by using heavy regenerative cuttings. The studies do not show that optimal RFM is more profitable than optimal CCF.

RFM can be made profitable to forest landowner by state subsidies, and this is what is currently done in Finland. Non-profitable silviculture is recommended, and it is made profitable again with the help of subsidies. However, the society would gain more by withdrawing these subsidies and promoting economically more viable management. To promote socially optimal forest management the beneficial externalities of timber production could be subsidized instead. As explained in this chapter, the most profitable management would be continuous cover forestry.

When evaluated from the multiple-use and multi-functional forestry perspectives the superiority of CCF increases. This is mainly because of the poor performance of RFM during the first decades after clear-felling. Taking into account the changing uses of Finnish forests, with decreasing importance of timber production and increasing importance of non-timber benefits, it is evident that alternatives to the current even-aged silviculture should be studied and promoted.

From the wood production point of view, the demand for pulpwood is slowly decreasing and the demand for biofuel is increasing. Therefore, the two principal future timber assortments may be saw log and biofuel, instead of saw log and pulp wood. This change improves the relative value of hardwoods as compared especially to spruce pulpwood. At the moment, small-sized spruce timber is clearly more valuable than small-sized hardwood but the order may soon change since the energy content of hardwood is higher than that of conifer wood. As a consequence, the prevailing practice of planting spruce on fertile sites becomes questionable since most cutting areas would regenerate naturally with different hardwood species. This regeneration could be used as biofuel when the trees are large enough for a cheap harvesting operation.

Clear-felled fertile sites regenerate naturally for hardwood and spruce regenerates easily under the hardwood cover. One management option is to produce hardwood biomass for biofuel at first, and then convert the stand to a mixed uneven-sized stand, which is managed with repeated high thinnings (Photo: Olavi Laiho).

Since spruce will enter the hardwood stand as an under-storey, spruce timber will be obtained later on without any cost. The two-layered mixed stand could be managed with repeated high thinnings, as long as regeneration and ingrowth continue. On some sites, such as most peat lands and fertile moist sites, regeneration is always sufficient for CCF. In the other cases, heavier cuttings are required every now and then to enhance regeneration and re-initiate the natural species cycle. This type of CCF management follows the natural species dynamics, is cheaper, and produces assortments that are required in the future. The hardwood stage also prevents spruce butt rot from spreading to the new spruce generation, since the rot will disappear from the root systems of rotten spruce stumps during the hardwood phase.

On poorer sites, where only pine grows well, CCF requires constant regeneration and ingrowth of pine. Group selection may be used and small canopy gaps created if regeneration is otherwise insufficient. It is also possible to use variable-density thinning as well as partial shelter wood and seed tree cuttings to promote regeneration.

Extraction roads are good places for regeneration (Photo: Timo Pukkala)

Improved management of Finnish forests requires experimentation with different variants of CCF. The new experiments mentioned in Sect. 2 that mimic the natural disturbance regimes occurring on different scales will provide information for a continuum of forest management options. However, additional experiments, in which strip roads are an integral part of the disturbance dynamics, are required. Extraction roads may be enlarged into small gaps, or combined with spatially uneven thinning intensity where places adjacent to strip roads are thinned more heavily. Alternatives involving partial seed tree and shelter wood cuttings should also be studied in field experiments. More growing sites with different soil properties and moisture conditions should be included in the experiments to learn about the effect of site characteristics on the regeneration dynamics.

In the management problems analyzed in this chapter, a simple steady-state management problem of a pure one-species stand was solved first. This involved finding the optimal combination of only four decision variables (*bW*, *cW*, *N*, *Dmax*) and simulating the stand development for one cutting cycle. The problem was gradually extended to involve the penalty for lack of large trees, steady-state management of mixed stands, transformation cuttings, and steady-state management in which a long management cycle consisted of a sequence of shorter cutting cycles. The joint production of several benefits was analyzed and optimized in the last example.

Although many types of optimization have already been done in Finland, and can be made with the current models, most analyses conducted so far should be taken as the first tries and preliminary calculations. For example, the post-thinning diameter distribution was described with a Weibull function in almost all optimizations described in this chapter. This may be sufficient for the steady state management of a pure one-species stand, but it may not be a justified simplification in mixed stands or when a sequence of distributions is optimized. The intervals between cuttings may be optimized simultaneously with the diameter distributions. When conversion cuttings are optimized, a better option would be to not fix the length of the transformation period or the structure of the ending forest (Tahvonen et al. 2010). For the near future, optimizations with higher biofuel and saw log price, and lower pulpwood price should also be conducted. Involving the species dynamics in the optimization of CCF management also needs much attention in future studies.

Group selection in a pine stand. Pine regenerates in canopy gaps and in heavily thinned places (Photo: Olavi Laiho)

Some of the optimization problems described in this chapter are spatial. In fact, the problem is always spatial when extraction roads are an integral part of the management system. This is the case especially in CCF where extraction roads and their vicinity are the best places for regeneration. The most convenient way to analyze spatial problems is to use distance-dependent growth models and spatial optimization. At the moment, there are already distance-dependent regeneration models for uneven-aged spruce stands, which predict the amount, survival and growth of regeneration in 4-m^2 cells as a function of over-storey density within and around the cell (Eerikäinen et al. 2007). This model, combined with distance-dependent simulation of the dynamics of larger trees, offers interesting possibilities to analyze spatially non-homogenous management systems.

Improved optimization techniques and economic analyses need improved modeling. The weakest parts in the current models for uneven-aged forests are the regeneration and ingrowth models. It is known that spruce regenerates more easily under pines and birches than under spruces, but this type of species interaction is not included in the models. Therefore, a necessary modeling task before doing more ambitious optimizations for mixed stands, are regeneration or ingrowth modeling for describing the influence that different tree species may have on the regeneration of other species. Also the inter-specific competition effects of larger trees should be modeled better than done so far. The most interesting, most challenging and most useful future research topic would be to include the species dynamics in the analysis of CCF management. This would require new field experimentation, new modeling efforts and new optimization tools.

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Chapter 4
Regulation of Timber Yield Sustainability for Tropical and Subtropical Moist Forests: Ecosilvicultural Paradigms and Economic Constraints

Armin H.W. Seydack

1 Tropical Forest Management Systems in the Context of Sustainable Development

1.1 Timber Yield Regulation as Primary Challenge

Two fundamentally different types of forest management may be distinguished. Following Gadow (2000) these are called rotation forest management (RFM) and continuous cover forestry (CCF). In rotation forest management (RFM), which is usually practiced in industrial timber plantations, the cycle begins with the establishment of young stands by planting. It continues with a series of standard thinning operations and ends with the final harvest at a specified rotation age. Thus, stands are not continuously covered by trees. Rotation is a term used in cash flow analysis and calculations of the normal growing stock. The rotation equivalent in continuous cover forestry is "cutting cycle". CCF is characterized by a broad continuum of silvicultural practices which all have in common that forest cover is never entirely removed. In tropical-type forests monocyclic management implies that all harvestable trees are either harvested during a single operation (clearfelling) or in two or three cuts relatively closely spaced in time (shelterwood systems). Removal of all mature trees takes place at long cutting cycles equal to the intended life spans of the stands and new stands develop from regeneration in existence at felling, representing largely even-aged forests (Whitmore 1991). This resembles rotation forest management (RFM). Under polycyclic management timber harvesting takes place at shorter cutting cycles and only that proportion of mature trees above a specified harvest maturity threshold is removed at any

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_4,

particular harvesting event (Whitmore 1991). Adult trees are continuously present at the canopy level, resulting in the maintenance of largely uneven-aged forests (continuous cover forestry).

Timber yield regulation is the primary challenge in tropical forest management where sustainable timber harvesting is pursued. In its most basic form this requires the determination of the sustainably harvestable timber yield, but may also involve supporting silvicultural measures and stipulations controlling logging practices. Sustainable timber yield regulation is thus here defined to include the scientific and technical components of harvest quota regulation, silvicultural measures and logging approaches. A yield regulation system must provide an unambiguous definition of the trees to be harvested at a defined area at any given harvesting event. Such a harvest must be congruent with successful stand regeneration and yield optimization, i.e. result in maximum sustainable timber yields.

In the development of silvicultural and yield regulation systems a number of interrelated components require consideration and stipulation: the optimum residual growing stock, harvest maturity thresholds (e.g. minimum harvestable diameters), the felling cycle linked to the stand regeneration strategy, the annual allowable cut and harvest tree selection criteria. Stipulation of these parameters will result in a particular yield regulation system which will occupy a position on the rotation forest management (RFM) to continuous cover forestry (CCF) continuum (from realization of net growth accumulation by periodic clearfelling to mortality pre-emption harvesting of selected trees under relatively short felling cycles). Yield regulation approaches encountered in tropical and subtropical forests can thus be similarly grouped as either monocyclic (RFM) or polycyclic (CCF) systems.

Overall, there is a trend from monocyclic to polycyclic yield regulation systems in the tropics, partly for different reasons compared with the RFM to CCF trend in temperate zone forests. The relative merits of the two approaches will be discussed in the context of emerging understanding of tropical forest dynamics and divergent ecophysiological trade-off constraints of trees responding to their edaphoclimatic environment. However, the polycyclic selection cutting systems generally implemented often tend to be so intensive that they result in transformation of the forests into secondary forests. The long-term sustainability of such forests in terms of particular commercial species and timber dimensions is uncertain. Based on an investigation of the performance of various yield regulation approaches sustainability risks related to forest matrix destruction effects, disruption of the functional interspecies dynamics and of regeneration processes are identified. A general lack of consensus among forest scientists is identified as how to ensure successful stand regeneration and how to approach yield optimization. Furthermore, lack of consensus also exists with respect to economic viability of sustainable timber harvesting.

A forest matrix invasion model is presented as more adequately describing the dynamics of tropical forests, contrasting the successional disturbance model generally invoked for temperate forests. Furthermore, the understanding of ecophysiological trade-off constraints of trees in relation to their edaphoclimatic environment was found to permit greater understanding of forest dynamics under divergent environmental conditions. It is concluded that, contrasting the progress made in the application of forest growth modelling and simulation for yield optimization, the ecosilvicultural basis of yield regulation requires intensified attention. Some of the inadequacies and incomplete integration of ecosilvicultural facets into yield optimization approaches are highlighted. These include issues such as an incomplete understanding of crowding effects on tree growth, the role of the forest matrix for the regeneration dynamics of supra-canopy occupying species, the implications of intra- versus interspecific interactions and the absence of considerations regarding genetics in yield regulation for tropical and subtropical forests. Addressing these issues is considered imperative if truly sustainable yield regulation systems are to be implemented in tropical and subtropical forests.

1.2 Sustainability as Guiding Principle

Sustainability is considered the central guiding principle for timber harvesting. Following the definition by Goodland et al. (1990), sustainable use of moist subtropical and tropical forests is here taken to mean that use of natural forest which indefinitely maintains the forest substantially unimpaired both in the environmental services which it provides, as well as in its biological quality. Thus any harvest must not exceed the regeneration rate of the resource, nor impair the potential for similar harvests in the future.

Sustainable yield regulation systems and associated silvicultural measures have seldom been implemented on a widespread basis (Poore et al. 1989; Buschbacher 1990; Panayotou and Ashton 1992; Hartshorn 1995; ITTO 2006). The principal obstacles to sustainable forest management in the tropics and subtropics relate to economic, institutional and sociopolitical constraints (e.g. Barbier 1998; Dawkins and Philip 1998; Blundell and Gullison 2003; Hall et al. 2003; Van Gardingen et al. 2003; Pearce et al. 2003); including marginal financial performance, contrary economic incentives, insufficient professional staff, limited regulatory capacity and government control and low socio-political feasibility.

The successes of implementation of sustained yield regulation is therefore most often determined within the broader socio-economic and political context (Palmer 1975; Weidelt 1989; Bruenig and Poker 1991; Schmidt 1991). The technical challenge of sustainability, i.e. optimally matching harvest rates with biological productivity rates, has to be pursued within the framework of three factors (*vide* Goodland et al. 1990; Bruenig 1996): human population pressure; national resource policies and sustainability of effective management institutions. As phrased by Goodland et al. (1990): "Sustainability has two fundamentals: stabilising populations to live within the carrying capacity of their resources, and stabilising economic systems so that they do not increase strains on the finite environment". Following from these, success in the achievement of sustainability depends on a resource policy which integrates forestry equitably with agricultural and economic development (Bruenig 1996). Under conditions of high population growth and rural poverty the management of tropical and subtropical forests for economic production is considered a key element in their conservation (Schmidt 1991); although Dickinson et al. (1996) caution against the general validity of the “use it or lose it” principle. The successful management for economic production requires pre-conditions such as long-term security, operational control, a suitable financial environment and adequate information (Poore et al. 1989).

Yield regulations systems for the sustainable harvesting of timber in the tropics have to be developed and implemented under conditions of ecological complexity and multiple conflicting interests of socio-economic needs and biodiversity requirements (Seydack 2000, 2002). Most often there will be a trade-off between biodiversity and other non-timber values, on one hand, and maximum financial return, on the other (Pearce et al. 2003). These complexities preclude a reductionist approach to management if overexploitation of resources is to be avoided (Ludwig et al. 1993). The concept of sustainable development integrates biology, economics and politics and allows for goals and approaches to change over time (Salwasser 1993). The role of science within the associated context of adaptive management is to determine what is biologically feasible and what are the trade-offs between alternative choices (Salwasser 1993). Achievement of this role requires the more effective utilization of existing knowledge (Bruenig 1996), especially the greater integration of ecological knowledge into the development of yield regulation systems (Seydack 2000, 2002; Sist et al. 2003a) if effective technological assistance is to be provided by the international community to assist the transition to sustainable forest management (Barbier 1998).

2 Yield Regulation Strategies and Parameters

2.1 Components of Timber Yield Regulation Systems

A sustainable timber yield regulation system is here taken to include the scientific and technical components of harvest quota regulation, silvicultural measures and logging approaches. To be a complete system, all four components, i.e. yield optimization, stand regeneration, harvesting interval and harvest tree selection criteria, require to be addressed. Any particular yield regulation system is derived by quantifying and stipulating some or all of the following yield regulation components (Seydack 2002):

- Optimum residual growing stock (optimum RGS)
- Harvest maturity thresholds (HMT)
- Harvest interval (felling cycle and rotation)
- Annual allowable cut (AAC)
- Harvest tree selection criteria

Definition and stipulation of these parameters will result in a particular yield regulation system which will occupy a position on the rotation forestry management (RFM) to continuous cover forestry (CCF) continuum (from realization of net growth accumulation by periodic clearfelling to mortality pre-emption harvesting of selected trees under relatively short felling cycles). Any resultant system is based on our current understanding of regeneration dynamics, growth patterns, treatment responses and functional dynamics manifesting itself in forest structure, species diversity and dominance relationships.

The residual growing stock is either stipulated in a silvicultural context (minimum seed trees; degree of permissible canopy opening) or in relation to maximum sustainable yields taken to be associated with a particular optimal RGS. Crowding effects are then usually implied. This parameter furthermore relates to the question of whether to perform silvicultural measures that release trees from crowding and competition. The harvest maturity threshold (HMT) is the selected stage of maturity above which particular trees are to be harvested and can be defined either by diameter (minimum harvestable diameters: MHD) or maturity condition criteria (MCC: Seydack 1995; Seydack et al. 1995). The HMT is determined in relation to (1) the type of product pursued (log dimensions); (2) the diameter, volume or value increment culmination patterns and (3) the pattern of wood quality deterioration with age. Selection of the RGS and HMT parameters leads to the choice from a yield optimization continuum having the following extremes (Seydack 2000, 2002):

- Yield realization from accumulated net growth: Achieved by reducing the growing stock and age structure to an appropriately early developmental stage characterised by relatively free-growing trees and low levels of mortality.
- Yield realisation through selective mortality pre-emption: Growing stock levels are relatively high resembling fully stocked stands with predominantly mature-phase trees. More trees are growing at moderate rates and the relatively high mortality levels are converted into yield through mortality pre-emption by selectively harvesting mature trees of declining vigour.

The harvesting interval should be the outcome of an optimized compromise and is affected by the following factors:

- Desired degree of canopy opening (stand regeneration)
- Position on the yield optimization continuum
- Minimum economic cut levels
- Anticipated logging damage levels
- Operational economics

Depending on whether the main commercial gap opportunist species are perceived to be long-lived pioneers or forest matrix invaders (Seydack 2000; Sect. 5.1), the stand regeneration strategy would involve relatively high to low degrees of canopy opening respectively.

The annual allowable cut (AAC) can be maximized in the long-term through the optimization of the above yield regulation parameters in alignment with the inherent productivity of the particular forest. Finally, the optimized yield parameters must be translated into reality through harvest tree selection. Most yield regulation systems applied to tropical and subtropical forests define trees to be harvested by minimum harvestable diameters (MHD); i.e. all trees above a specified diameter are harvested.

Harvest tree definition may also be according to maturity condition criteria (MCC) based on indications of advanced maturity and short remaining life expectancy of trees (Seydack et al. 1995). Extending the yield optimization continuum we obtain the spatiotemporal harvest concentration continuum (which is largely equivalent to the RFM-CCF continuum):

- Towards the high concentration end of this continuum, yield realisation is based on inducing a state of net growth accumulation through maintaining stands at relatively reduced growing stock levels and trees harvested at relatively low harvest maturity thresholds. High diameter increments are maintained during sub-mature phases through the early release of future crop trees. Stand regeneration is pursued through substantial canopy opening based on the assumption that commercially desirable emergent canopy species are light-demanding long-lived pioneers with large gap requirements.
- At the other end of the continuum, yield optimisation is pursued through mortality pre-emption by harvesting trees timeously but generally at relatively high harvest maturity thresholds and from fully stocked stands. This approach capitalises on high volume increments through extended growth of trees at large diameters achieved during relatively mature growth phases. Stand regeneration relies on small-scale natural renewal processes following from natural gap dynamics deemed adequate also for the regeneration of matrix invading emergent canopy species.

Both stand regeneration as well as yield optimisation strategies revolve around the optimal balance between the amount and type of the growing stock to remain (RGS) as opposed to that portion of the growing stock to be removed by harvesting and/or release thinning operations. In practice, the harvestable quantities perceived to represent minimum economic cut levels and the associated logging damage levels often result in a distortion of this balance in favour of too heavy removals and thereby resulting in unsustainable harvesting.

2.2 Stand Regeneration Strategies

Successful regeneration depends on the proximity of seed sources (seed trees) to spaces suitable for seedling establishment (establishment space) and suitable recruitment conditions for established seedlings to survive and reach adult canopy positions (recruitment space). The regeneration space (combined establishment and recruitment spaces) refers to the variable and changing shelter and light requirements of recruits along the time axis in accordance with the metabolic type of the species concerned (traditionally narrowly seen as a shade tolerance and intolerance gradient). Stand regeneration strategies pertain to decisions aimed at the maintenance or creation of regeneration spaces in the effective vicinity of seed trees, thereby sustaining conditions expected to result in successful regeneration (Seydack 2000).

Viewed from the regeneration perspective, yield regulation systems should also been seen as stand regeneration strategies and may be grouped into two main categories: (1) monocyclic systems (uniform and tropical shelterwood systems) and (2) various polycyclic selection cutting systems: with or without regeneration and productivity enhancement measures. Which one of these stand regeneration approaches is favoured depends on the regeneration status (actual representation of regeneration: presence or absence of advanced regeneration) and the regeneration requirements of commercial species in terms of seed source distribution and regeneration space. Under uniform systems (e.g. Malayan Uniform System) all canopy trees are removed in a single operation and existing advance regeneration is expected to form the next tree crop. The aim of tropical shelterwood systems is to produce a more or less even-aged forest by establishing sufficient regeneration of commercial species through canopy and subcanopy thinning within a limited regeneration period starting before full canopy logging. With polycyclic selection cutting systems regeneration is generally assumed to continue naturally; but attention to adequate seed tree retention, regeneration space manipulation or enrichment planting is often required.

2.3 Yield Optimisation Strategies

Selecting a yield optimisation strategy requires decisions regarding (1) the stocking of the residual stand (RGS: residual growing stock) and (2) at what stage of maturity trees are to be harvested (HMT: harvest maturity threshold). Individual tree diameter growth may either be maximised, most often by concentrating growth onto pre-selected crop trees (low residual stand density: fewer faster-growing trees) or maximum productivity is pursued per unit of growing space (high residual stand density: more, slower-growing trees). Furthermore, trees may either be harvested at a relatively premature stage (low target diameter) or harvestable trees may be allowed to continue growing whilst already having attained comparatively large sizes (Seydack 2000).

Depending on the yield optimisation strategy, growing stock reductions in pursuit of productivity enhancement may be implemented through one or more selected interventionist treatments (*vide* Baur 1964; Hutchinson 1988; Uebelhör et al. 1989; Parren and De Graaf 1995; Bruenig 1996).

- Conversion-geared canopy opening. Removal of tolerant understorey trees in order to release regeneration of other species or removal of non-utilizable overstorey trees may be undertaken.
- Climber cutting. This operation results in some opening of the canopy, but its main purpose is to reduce felling damage.
- Exploitation. Harvesting of trees and the associated damage to residual trees normally results in the most substantial canopy opening. Depending on the clumping of harvested trees, the effect is often spatially irregular.

- Post-logging thinning treatments. Refining involves the removal of undesirable elements all over the stand (usually non-commercial species down to a prescribed diameter). Liberation thinnings are carried out to remove competing trees around selected potential final crop trees.

The residual growing stock is either stipulated in a silvicultural context (minimum number of seed trees; degree of permissible canopy opening) or in relation to maximum sustainable yields taken to be associated with a particular optimal RGS. Crowding effects are then usually implied. This parameter furthermore relates to the question of whether to perform silvicultural measures of tree release from crowding and competition or not.

The harvest maturity threshold (HMT) is the selected stage of maturity above which particular trees are to be harvest and can be defined either by diameter or maturity conditions criteria (Seydack 1995; Seydack et al. 1995). The HMT is determined in relation to (1) the type of product pursued (log dimensions), (2) the diameter, volume or value increment culmination patterns, (3) mortality trends over time, (4) characteristics of reproductive maturity and (5) patterns of wood quality deterioration with age.

2.4 Operational Parameters

Any yield regulation system, in order to result in sustainable timber harvesting, must include adequate definition of the following facets:

- Effectively exploitable area
- Harvesting interval (cutting cycle)
- Harvest tree selection criteria

The sustainable yield ultimately depends on the area under forest and its productivity with respect to commercially suitable species. The effectively exploitable area (EEA) is determined through subtraction of topographically inaccessible areas, portions under unsuitable forest types and areas to be excluded for conservation purposes from the total applicable forest area. The maximum area to be harvested annually must be limited to that proportion of the effectively exploitable area which allows for an approximately equal annual harvest in perpetuity (= EEA ÷ harvesting interval in years).

The cutting cycle should be the outcome of optimized compromise as affected by (1) the desired degree of canopy opening (stand regeneration); (2) tree growth patterns (yield optimisation); (3) operational economics and minimum economic cut levels and (4) anticipated logging damage levels. As the intention is to harvest the growth accrued between harvesting events, longer harvesting intervals are associated with higher harvest intensities (favourable operational economics) and thus with higher degrees of canopy opening. Severe canopy opening favours light demanding species, whereas intermediate canopy openings are suitable for forest matrix invading canopy species. Most of the commercially desirable species belong to the latter group (Sect. 5.1).

Finally, a yield regulation system must define which trees are to be harvested (selection criteria). Trees to be harvested may be defined (1) on an area basis; i.e. all trees on a particular area are harvested; (2) through minimum harvestable diameters (MHD); or (3) based on features of tree condition (MCC: maturity condition criteria). Most yield regulation systems applied to tropical and subtropical forests define trees to be harvested by MHDs; i.e. all trees above a specified diameter are harvested. Harvest tree definition may also be according to maturity condition criteria (MCC) based on indications of advanced maturity and short remaining life expectancy of trees (Seydack et al. 1995; Seydack 1995, 2000, 2002).

3 Yield Regulation Systems

3.1 Classification of Systems

Monocyclic systems involve the harvesting of all harvestable trees during a single operation and rely on seedling regeneration to develop into the next, largely even-aged crop over the rotation period. Under polycyclic systems only a portion of the growing stock above a suitably selected harvest maturity threshold (generally minimum harvest diameter limits) is harvested at shorter intervals, the felling cycle. Advanced residuals which continue to grow and exceed the harvest maturity threshold are intended to form successive harvests. The resultant forest stands are largely uneven-aged (Whitmore 1991).

Viewed from the regeneration perspective, yield regulation systems can be taken as stand regeneration strategies and grouped into two monocyclic and two polycyclic system groups (Seydack 2000, 2002):

- Uniform systems (monocyclic)
- Tropical shelterwood systems (monocyclic)
- Manipulistic selection cutting systems (polycyclic)
- Naturalistic selection cutting systems (polycyclic)

Which of these four stand regeneration approaches is favoured depends on regeneration status (actual representation of regeneration) and perceived regeneration requirements in terms of seed source distribution and regeneration space. The stand regeneration systems above are listed in decreasing order of impact on the canopy, from virtually clearfelling the stand (high spatiotemporal harvest concentration) to naturalistic selection cutting systems (low spatiotemporal harvest concentration). Any selection cutting system where the intensity and kind of harvesting is in a manner permitting successful regeneration to take place naturally without any additional regeneration enhancing measures, were termed naturalistic

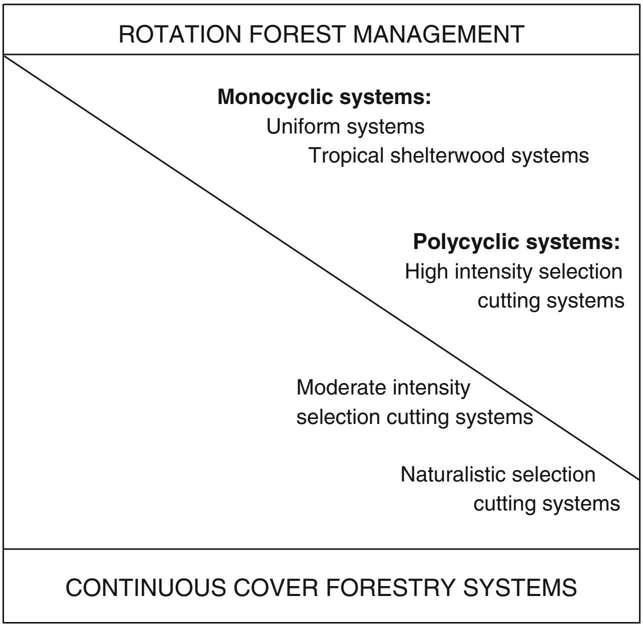

Fig. 4.1 Types of yield regulation systems regarding tropical and subtropical forests on the rotation forest management (*RFM*) to continuous cover forestry (*CCF*) continuum (Seydack 2002)

selection cutting systems. The latter also implies that the spatial concentration and extent of felling must remain in accordance with the grain of disturbance naturally experienced. These types of systems and their position on the monocyclic to polycyclic harvest intensity continuum are depicted in Fig. 4.1.

A typical example of a monocyclic uniform system is the Malayan Uniform System (MUS). The objective of management is a more or less even-aged forest with a high proportion of commercially desirable species. The system was generally operated on a 70 year cutting cycle and required that advance seedling regeneration was present on the ground at the time of felling, which survives undamaged and develops to crop dominance without assistance. The system was successfully applied in the lowland dipterocarp forests of Malaysia (Appanah and Weinland 1992), but was not considered suitable for hill dipterocarp forests, mainly due to the paucity of advance regeneration in virgin stands (Thang 1987). The natural regeneration of commercially valuable species (mainly Meliaceae) in West African tropical rainforests was often considered inadequate. As a result tropical shelterwood systems (TSS) were chosen in an attempt to induce the regenerations of such species. As an example, the Nigerian TSS comprises a series of operations designed to open the canopy in order to induce regeneration and enhance the growth and development of existing regeneration. When the regeneration is rated successful, the temporary shelterwood, functioning as shelter and source of seeds, is removed, leaving only young trees in the stand (Parren and De Graaf 1995).

Selection cutting systems amount to determining minimum harvestable diameter (MHD) felling limits (e.g. 45–60 cm DBH) for harvesting and retaining a stipulated number of advanced residuals for the next harvest at relative short felling cycles (25–40 years). Within this category those selection cutting systems are grouped where the harvesting impact is deemed to require regeneration enhancing measures. Harvesting intensities of commercial species are such that low MHDs result in relative premature harvesting; requiring specific stipulations to retain residual seed trees. This category also includes selection cutting systems involving measures geared to the enhancement of regeneration: enrichment planting, tending and canopy opening through the culling of non-harvest trees in order to favour the regeneration of commercially desirable species. Examples of such selection cutting systems are the Selection Management System (SMS) operated in Peninsular Malaysia (Thang 1987) and the selection system practised in Ghana (Parren and De Graaf 1995).

Most selection cutting systems with high spatiotemporal harvest concentrations with low MHDs, large-scale felling damage and low RGS levels can be termed pseudo-CCF systems. Contrasting these, naturalistic selection cutting systems require that the spatial concentration and extent of felling is in accordance with the grain of disturbance naturally experienced by the forests involved. This implies that the degree of canopy opening associated with harvesting must result in a range of canopy gaps which is congruent with the regeneration space requirements of the species required to regenerate sustainably. Harvesting and natural regeneration are fully integrated and relatively high harvest maturity thresholds (HMTs) apply (*vide* Bruenig 1996). The nature and levels of the RGS are not permitted to deviate substantially from near-primary forest conditions. An example of a naturalistic yield regulation system practised in a South African evergreen moist forest, the Knysna forest, involves mortality pre-emption through the selective harvesting of mature and overmature trees proportional to species-specific turnover. Maturity condition-based harvest tree selection criteria permit the differentiated harvesting of canopy trees of declining productivity and the retention of productive canopy trees. A felling cycle of 10 years applies (Seydack 1995, 2000; Seydack et al. 1995).

3.2 Assessment of Systems Performance

Monocyclic yield regulation systems involve the removal of all overstorey timber at one cut and largely rely on seedlings to form the next crop (Whitmore 1991). Depending on the presence or absence of advanced regeneration, uniform one-cut or shelterwood systems are applied. Under the one-cut uniform system the entire overstorey is removed in a single harvest. A typical example is the Malayan Uniform System (MUS: Thang 1987; Appanah and Weinland 1992). If the presence of advanced regeneration is inadequate, it is necessary to retain some of the overstorey for several years. The aim of Tropical Shelterwoods Systems (TSS) is to produce a more or less even-aged forest by establishing sufficient regeneration of commercial species within a limited regeneration period starting before and extending beyond

Table 4.1 Limitations and inadequacies: monocyclic systems (uniform one-cut and shelterwood systems)

1. Stand regeneration
    - One-cut systems: inapplicable/impractical where levels of advanced regeneration of commercial species at time of exploitation are low or unevenly distributed (Cheah 1991; Lamprecht 1993; Bruenig 1996; Appanah 1998; Dawkins and Philip 1998)
    - Shelterwood systems: Induction of regeneration difficult and success variable (Baur 1965; Parren and De Graaf 1995; Dawkins and Philip 1998)
2. Silvicultural intervention requirements
    - Repeated silvicultural treatments are required to ensure successful regeneration (Lamprecht 1993; Parren and De Graaf 1995; Kleine 1997; Ashton et al. 2001)
    - Complexities of diverse silvicultural requirements and responses varying over sites and species (Dawkins and Philip 1998)
    - Destruction of subcanopy forest matrix results in uniform overcrowded stands requiring timeous thinning treatments (Bruenig 1996; Okuda et al. 2003)
3. Economics of silvicultural interventions
    - Silvicultural treatments are costly and often not economically viable (Panayotou and Ashton 1992; Ashton and Peters 1999; Ashton et al. 2001; Hall et al. 2003)

full canopy logging. This is pursued through the formation of a shelterwood of mother (seed) trees which are subsequently removed when regeneration has become established. Both types of monocyclic systems were considered silviculturally successful, mainly with reference to the successful regeneration of light demanding timber species (Ashton and Peters 1999; Dawkins and Phillip 1998; Ashton et al. 2001; Lamprecht 1993).

Although largely replaced by polycyclic systems, introduction of monocyclic systems for certain forest areas in order to ensure or promote the regeneration of desirable commercial species is advocated (Dawkins and Philip 1998; Ashton and Peters 1999). Some of the main limitations, complexities and inadequacies of monocyclic systems as applied in the tropics, which were responsible for their discontinuation, are listed in Table 4.1.

Polycyclic yield regulation systems are here grouped into three categories: selective logging systems (SLS), selection management systems (SMS) and naturalistic selection cutting systems (NSCS). Selective logging systems in their most basic form involve no silvicultural interventions and yield regulation is based on the undifferentiated application of MHD criteria. This approach amounts to being the standard management practice for tropical forests today. Logging is the only silvicultural event and no additional silvicultural measures are considered necessary or economically justifiable. Minimization of exploitation damage to advanced regeneration through reduced impact logging (RIL) is increasingly undertaken (Putz et al. 2008). Recommended improvements to selective logging systems involve more differentiated harvest definition above MHDs through limitations on harvest intensities and species-specific tree retention rules (Sist et al. 2003a). Optimization of yield regulation parameters in this context is effectively facilitated by stand dynamics modelling and harvesting scenario simulations (Huth and Ditzer 2001; Van Gardingen et al. 2003, 2006; Gourlet-Fleury et al. 2004, Kariuki et al. 2006).

Table 4.2 Limitations and inadequacies: polycyclic systems

1. Inadequate regeneration
   - Logging/harvesting disturbance alleged to be insufficient for the regeneration of light demanding species (Snook 1996; Fredericksen 1998; Brown 1998; Dawkins and Philip 1998; Ashton and Peters 1999)
   - Excessive exploitation impacts jeopardizing regeneration (Van Gardingen et al. 1998; Bruenig 1996; Dawkins and Philip 1998)
2. Unsustainable harvest levels
   - Harvesting all trees above relatively low MHD results in unsustainable harvest intensities; especially in the case of bell-shaped population structures (Lamprecht 1993; Snook 1996; Dawkins and Philip 1998; Sist 2000; Sist et al. 2003a; Van Gardingen et al. 2003; Grogan et al. 2008)
   - Species and quality selective overharvesting (Appanah et al. 1990; Dawkins and Philip 1998; Brown 1998; De Graaf 2000)
   - Associated high levels of logging damage (Dawkins and Philip 1998)
3. Inadequacy of residual growing stock
   - Decimation of seed sources (Weidelt 1989; Plumptre 1995; Snook 1996; Fredericksen 1998)
   - Low MHD/short felling cycles: premature removal of productive and value-producing trees (Rietbergen 1989; Bruenig 1996)
   - Repeated removal of fastest growers: leaving a residual growing stock of slower-growing trees or having dysgenic effects (Weidelt 1986; Appanah and Weinland 1992; Lamprecht 1993)
   - Residual growing stock consisting of trees of low growth performance (Thang 1987; Lamprecht 1993; Seydack et al. 1995; Bruenig 1996)
4. Silvicultural interventions
   - Absence of silvicultural content (Lamprecht 1993; Kleine 1997)
   - Non-implementation of conceptionally required interventions (Cheah 1991; Appanah and Weinland 1992; Bruenig 1996; Dawkins and Philip 1998)
   - Economic viability questioned (Van Gardingen et al. 2003)

In addition to elements of yield regulation as for SLSs, selection management systems (SMS) involve silvicultural interventions aimed at the enhancement of regeneration or forest productivity, resulting in the intentional modification of forest structure and composition. Productivity and sustainability of selection management systems are dependent on the actual implementation of stipulated silvicultural interventions. Some examples of systems in this category are the Selection Management System (Malaysia), the Philippine Selective Logging System (Lamprecht 1993), the Ghana Selection System (Parren and De Graaf 1995) and the CELOS Management System (Suriname, De Graaf 1986, 2000).

Both selective logging and selection management systems, in practice, represent high impact systems at the stand level or in respect of targeted commercial species. As such they are potentially subject to limitations and inadequacies and have the following problems (Table 4.2):

- Inadequate regeneration
- Unsustainable harvest levels
- Inadequacy of residual growing stock
- Silvicultural interventions economically controversial or neglected

In particular, their sustainability over many felling cycles is compromised due to sustainability risks due to forest matrix destruction, conspecific crowding, productivity performance deficiencies, decimation of non-commercial species and disruption of reproductive and genetic systems (Sect. 4).

As primary precondition, a selection cutting system where the intensity and kind of harvesting is in a manner permitting successful regeneration to take place naturally without any additional regeneration enhancing measures may be considered a naturalistic selection cutting system (NSCS). Harvesting and natural regeneration are fully integrated (*vide* Bruenig 1996). This requires that the spatial concentration and extent of felling is in accordance with the grain of disturbance naturally experienced by the forests involved. This implies that the degree of canopy opening associated with harvesting must result in a range of canopy gap sizes which is congruent with the regeneration space requirements of relevant species (Seydack 2000, 2002). An example of such a NSCS, as practised in Afromontane forests in South Africa, is described by Seydack (1995) and Seydack et al. (1995).

Accordingly, the two defining key principles of naturalistic selection cutting systems can be stated as follows:

- Stand regeneration: retention of adequate spatiotemporal representation of seed trees (moderate to high residual growing stock levels) and maintenance of natural range of canopy gap disturbance regime.
- Yield optimization: relatively long individual tree rotations (high harvest maturity thresholds: HMT), i.e. relative high target diameters for harvesting or maturity condition criteria; permitting productivity realization associated with the growth of relatively large trees.

Naturalistic selection cutting systems are thus particularly suited for the production of speciality timber of high quality and value rather than large quantities of commodity timber (Bruenig 1996). Such systems are often considered economically unattractive (e.g. Parren and De Graaf 1995). Especially when in degraded heavily logged over secondary forests the implementation of NSCSs would require long waiting periods which are not considered workable under the generally prevailing socioeconomic conditions (Kleine 1997). However, for large areas of relatively intact primary forests, NSCSs may represent superior options with respect to both conservation and economics as an alternative to once-off timber mining followed by secondary forest situations.

A substantial lack of scientific consensus is evident with respect to key issues regarding yield regulation. Major controversies centre around facets of regeneration, yield optimization and economic constraints. Subject to the successional disturbance model, regeneration problems are attributed to an insufficient magnitude of disturbance. Alternatively, the inadequate retention of seed trees is blamed for cases of regeneration failure. In terms of yield optimization, most seem to favour maximized early-phase productivity enhancement, whereas other stress the opportunities of extended late-phase productivity realization (Sect. 2.1). Furthermore, widely disparate perceptions exist as regarding economically viable minimum harvest intensity levels and the economic viability of silvicultural interventions is controversial. Finally, all yield regulation systems are considered silviculturally successful by some and are challenged by others and all potentially sustainable systems are judged of economically marginal viability (Sect. 6.3). Given the extent of disagreement, the exploration of new paradigms and perspectives is clearly indicated.

4 Sustainability Risks

4.1 Forest Matrix Destruction Effects

Tropical-type forests are characterized by a persistent multi-aged, heterospecific forest matrix (Newbery et al. 1999; Seydack et al. 2011a). The ecological and silvicultural importance of this forest matrix, in particular of the subcanopy and midstorey strata, has been recognized (Hutchinson 1988; Bruenig 1996; Newbery et al. 1999). Important functions of the forest matrix include protection of the soil environment, including nutrient cycles and mycorrhizal populations (Bruenig 1996), amelioration of microclimatic conditions after sudden canopy opening (Hall et al. 2003), as a nurse crop for commercially valuable overstorey species (Hutchinson 1988), as a spacing filter with respect to conspecific cohorts (Bruenig 1996, Sect. 4.2) and providing habitat elements for generalist pollinators (Sist et al. 2003a). Depending on circumstances, diverse complications and sustainability risks may be associated with high impact timber harvesting systems especially when they are implemented repeatedly:

- Proliferation of pioneer species. Early successional pioneer growth may impede prompt regeneration of commercial tree species (Kleine 1997). More importantly, under such conditions the finally recovered canopy layer tends to be dominated by pioneer species, with the commercially desirable overstorey species poorly represented (Hawthorne 1996; Sist and Nguyen-The 2002).
- High post-logging mortality and poor juvenile growth under conditions of high irradiance. Regeneration and recruitment of most commercially desirable overstorey species have disturbance requirements resulting in small to intermediate canopy gap sizes (Negreros-Castillo and Hall 1996; Kuusipalo et al. 1996; Tuomela et al. 1996; Van Gardingen et al. 1998; Hall et al. 2003; Sist et al. 2003a, b).
- Growth stagnation due to intra-cohort crowding. Bruenig (1996) cautions against the removal of the subcanopy matrix which is deemed essential to diversify the light climate and growing space in order to prevent congested and overcrowded tree stands requiring costly tending to release future crop trees.
- Short merchantable bole lengths. Immature residuals growing through a low forest matrix may have reduced clear bole heights and associated volumes of wood production (Assman 1970; Torquebiau 1986; Hutchinson 1988; Poker 1993).

- Early growth peak and senility. Under conditions of high irradiance levels early in the development of many overstorey species they would be cued into an early metabolic performance mode. This may result in early growth peaks, early onset of mortality, reduced final dimensions and possibly lower wood quality (Assman 1970; Seydack 2000, 2002).

As indicated, the absence of an effective subcanopy forest matrix resulting from high impact timber exploitation can have detrimental effects specifically regarding the performance of overstorey species (forest matrix invader species: Sect. 5.1) which typically recruit through a heterospecific, uneven-aged forest matrix. Monospecific aggregations which can result from forest matrix destruction are thus expected to ultimately cause recruitment constraints due to intra-specific crowding effects in the proximity of conspecific adults (Sect. 4.2).

4.2 Disruption of Functional Inter-species Dynamics

Tropical-type forests are comparatively species-rich and characterized by predominantly heterospecific tree neighbourhood configurations. Anthropogenic interventions resulting in the destruction of the forest matrix or culling of non-commercial species in the context of potential crop tree (PCT) release is likely to cause greater conspecific aggregation in successively recovering forests. The fine-grained species mixture generally encountered in tropical forests is presumably brought about by some mechanism counteracting conspecific neighbourhood of trees (Uriarte et al. 2004). Negative conspecific effects at the very local scale (usually <15 m around subject trees) have been documented for many species; particularly for overstorey species in tropical forests (Wright 1999). For example, in a neotropical forest (Barro Colorado Island), canopy tree survival was negatively affected by conspecific density and this effect was expressed more strongly in rare than common species (Hubbell et al. 2001). In another study, negative effects on both growth and survival of saplings in the proximity of conspecific adults were detected, especially among canopy and midstorey species (Hubbell and Foster 1990). Decreased performance of juvenile trees around conspecific adults has often been attributed to disproportionately high mortality of such trees caused by host-specific seed predators, herbivores or pathogens according to the Janzen-Connell model (Clark and Clark 1984; Peters 2003). Alternatively, in a study involving trees of Dipterocarpaceae in Bornean rainforests, resource competition was implicated by Stoll and Newbery (2005) in explaining the suppressed growth of small trees when they had adults of the same species as neighbours. However, the mechanism underpinning conspecific neighbourhood effects has yet to be resolved. Depressed recruitment performance of trees in the proximity of conspecific adults in terms of survival and growth thus appears to be applicable in many species in tropical forests (Okuda et al. 1997; Penfold and Lamb 1999; Hubbell et al. 2001; Peters 2003; Seydack et al. 2011b). Following from these findings it is suggested that artificially induced monospecific aggregations, either arising through silvicultural operations or as a result of high impact logging, may be problematic if survival and growth performance of recruits are to be sustained over successive tree generations (Seydack 2000, 2002).

4.3 Disruption of Regeneration Processes

High intensity timber harvesting involving comparatively high proportions of the growing stock (usually of trees above relatively low MHDs) significantly reduces the spatiotemporal representation of reproductive adults. Such reductions are expected to be particularly disruptive to regeneration in tree species (1) when occurring at low densities with individual trees widely spaced, (2) where fecundity is strongly size-dependent, (3) characterized by population size structures where most individuals are of relatively large size and submature individuals poorly represented, (4) with short dispersal distances and (5) in species which are susceptible to disruption of their genetic and sexual systems when subjected to severe population reductions or distortions of inter-tree spacing configurations.

The extent to which an area is supplied with seed is determined by seed productivity and effectiveness of dispersal. Both can be negatively affected by the selective removal of conspecific neighbours (Guariguata and Pinard 1998). Tree harvesting based on relatively low MHDs results in the decimation or local extinction of adult trees of rare species and those with bell-shaped diameter class distributions (Snook 1996; Brown 1998; Sist et al. 2003a, b; Schulze et al. 2008). Declining regeneration capacities of logged-over forests have been recorded (Tuomela et al. 1996; Snook 1996; Plumptre 1995). Selective logging operations remove trees above fixed diameter limits and therefore result in the loss of the largest trees which are likely to contribute disproportionately to successful regeneration of the forest following harvesting (Plumptre 1995; Grogan et al. 2003; Snook et al. 2005; Nabe-Nielsen et al. 2009); thereby negatively affecting seed availability during spatiotemporally scarce disturbance events favourable for establishment. Larger trees are not only more productive, but their larger stature also favours effective seed dispersal (Snook et al. 2005). In species with relatively low dispersal distances the need for adequate residual population densities or the more evenly spaced retention of relatively large-sized mother trees is accordingly indicated (Sheil and Van Heist 2000; Guariguata and Pinard 1998).

Seed production may be limited by low cross-pollination among widely spaced trees. Self-incompatible forest trees and trees pollinated by small insects are expected to be particularly sensitive to pollinator mediated density dependent effects (Ghazoul et al. 1998). Large trees should therefore be maintained close enough to allow effective pollen exchanges (Caron et al. 2004). Low densities of big trees can decrease the number of flowering trees and increase mating among relatives. Some concerns have been raised that this is likely to lead to in-breeding depression and reduced reproductive output (House 1992; Murawski et al. 1994; Konuna et al. 2000).

Jennings et al. (2001) suggest that the genetic diversity of shade tolerant trees species might be little affected by moderately intense timber harvesting as advanced regeneration should serve as a genetic reservoir for the species being harvested. However, logging at too low minimum diameter limits will drastically reduce the adult population and leave few young individuals in the case of many canopy species with bell-shaped diameter class distributions and poor representation of juveniles (paucity of advanced regeneration). For such species and conditions the adequate retention of seed trees remains crucially important in order to avoid the sustainability risks associated with reductions in the effective population size as a consequence of logging (Andre et al. 2008; Ng et al. 2009). As silvicultural interventions in order to secure successful regeneration are seldom logistically practical or economically viable, adequate seed tree retention is a precondition in order to ensure that trees can effectively utilize spatiotemporally scarce conditions suitable for regeneration.

5 Contrasting Ecosilvicultural Paradigms

5.1 Divergent Models of Forest Dynamics

It is suggested here that the dynamics of tropical-type forests occurs according to the forest matrix invasion model. This model involves the dynamics of transient large gap colonizing species, persistent forest matrix occupants as well as dominance fluctuations/nomadic aggregations of forest matrix invaders. The two main guilds are the forest matrix occupants and the forest matrix invaders. The forest matrix occupants are represented by tree species of somewhat lower stature which form the bulk of the forest matrix vegetation, usually occupying positions of subcanopy and middle-storey status in the presence of forest matrix invaders, but may also constitute the canopy layer in their absence. Forest matrix invaders are species recruiting through the forest matrix in small to moderately sized overhead gaps towards supra-canopy positions of canopy, upper canopy and emergent strata. Forest matrix invaders are accordingly defined by the occupancy of upper canopy strata and not by the degree of light resource responsiveness. These forest matrix invaders, the guild of rain forest emergents and upper canopy occupants, often erroneously termed long lived pioneers, to which most commercially desirable species belong (*Dipterocarpaceae*, *Meliaceae*), differ from forest matrix species. They appear to have greater intraspecific metabolic plasticity, exhibiting pronounce ontogenetic shifts from juvenile to mature metabolic modes. Such capacities allow these forest matrix invaders (Seydack 2000, 2002) to effectively invade the forest matrix due to their juvenile shade tolerance (Ashton 1988; Weidelt 1989; Fickinger 1992), recruitment into the canopy through relatively small overhead gaps (Clark and Clark 1987; Tuomela et al. 1996; Seydack 2000; Brown et al. 2003) and extended growth sustained to relatively large diameters in order to reach the stature required for upper canopy or supra-canopy positions. Trees of this guild thus essentially invade the persistent forest matrix, forming nomadic spatiotemporally shifting aggregations in overstorey strata, resulting in spatiotemporal fluctuations in their dominance of those strata.

Due to the persistence of the forest matrix, recruitment of forest matrix invaders tends to be spatiotemporally dispersed, resulting in a fine-grained disturbance dynamics where trees of divergent life stages are intermixed at an individual tree level (Seydack et al. 2011a). This is in contrast to successional disturbance dynamics often invoked for temperate forests where light response guilds (pioneers, non-pioneer light demanders, long-lived pioneers, gap opportunists, shade bearers) attain successional dominance. Dynamics under this successional disturbance model results in cohorts of trees in the same life stage at the stand level (coarse-grained disturbance dynamics). As indicated here, tropical-type forests are generally subject to the dynamics according to the forest matrix invasion model and thus usually not subject to stand-level synchrony of successional stages (building, mature, old-growth, break-down stages). Rather, individual forest matrix invasion trees recruit through the largely persistent forest matrix towards some canopy positions. This has implications for timber yield regulation for tropical-type forests if such approaches are to remain in alignment with tropical forest dynamics and thereby ensuring long-term sustainability of timber harvesting. The regeneration of individual tree species is often inadequate following timber harvesting as generally implemented in tropical-type forests (Table 4.2). In line with the successional disturbance paradigm the problem is then often linked to disturbance characteristics, i.e. either too much or too little light availability for successful regeneration (Table 4.2). Considerations according to the forest matrix invasion model however place the emphasis on ensuring the adequate retention of seed trees for any given species (relatively high HMT and RGS levels). Only then can scarce regeneration or recruitment opportunities be realized by all tree species concerned.

Yield optimization may be pursued through either net growth or extended large-tree productivity realization (Sect. 2.1). Yield realization from accumulated net growth is achieved by reducing the growing stock and age structure to an early stand-level successional stage. The associated high impact harvesting is more in alignment with coarse-grained successional dynamics which is rather atypical for most tropical-type forests. Large tree productivity realization is achieved through mortality pre-emption by selectively harvesting mature trees of declining vigour (e.g. Seydack 1995; Seydack et al. 1995). The growing stock and individual tree dynamics is largely maintained with this tactic. Such an approach is in line with tropical forest dynamics (spatially fine-grained disturbance dynamics), thereby avoiding the problems associated with the decimation of the forest matrix usually associated with high impact tree harvesting. Thus, both with respect to successful regeneration and timber yield optimization for long-term sustainability the maintenance of comparatively high residual growing stock (RGS) levels is imperative given the nature of tropical forest dynamics according to the forest matrix invasion model.

5.2 Metabolic Performance Modes

The Lowland Dipterocarp forests of Malaysia were characterised by having adequate regeneration on the ground in most cases, allowing the implementation of the monocyclic Malayan Uniform System. In the Hill Dipterocarp forests the adequacy of regeneration was no longer the rule and this resulted in adopting selection cuttings under the Selection Management System (Thang 1987; Cheah 1991). Contrasting this, in West African forests, the Tropical Shelterwood System, a monocyclic system, was adopted when the existing regeneration and stocking of potential crop trees was insufficient (mainly in the moist semi-deciduous forests of Ghana and Nigeria). With sufficient regeneration in the moist forests of Ghana a selection system was implemented (Parren and De Graaf 1995). Application of Tropical Shelterwood Systems was presumably based on the contention that the forest with low levels of regeneration of gap opportunists species (mainly Meliaceae) were successional and that the removal of the canopy would induce regeneration of the light demanding species. It is not clear from the literature whether shelterwood systems were actually successful in inducing regeneration significantly more than would have occurred otherwise. According to Wyatt-Smith (1987) they largely failed in this respect. In contrast, Dawkins suggested that the Nigerian TSS was successful for light-demanding species in the absence of obstructing climbers and weeds (Dawkins and Philip 1998).

Perceptions regarding regeneration dynamics seem to be central to the controversy under discussion. In fully aseasonal forests, with respect to temperature and moisture (tropical lowlands), the presence and performance of regeneration under canopy and particularly on canopy opening appears to be more prevalent and predictable. Under such conditions monocyclic systems were generally perceived to be successful and favoured by many authors (see above). With increasing seasonality (associated with more fluctuating and lower levels of temperature and moisture), such as the Malaysian Hill Dipterocarp forests and West-African semi-deciduous forests, regeneration of commercially desirable emergent species often appeared inadequate. Inducing regeneration through canopy opening (e.g. TSS) was difficult and rarely successful (ex Dawkins and Philip 1998). It appears that the traditional, but simplistic grouping of tree species into light demanding and shade tolerant is inadequate in tropical forests for the understanding of regeneration dynamics (Clark and Clark 1992; Seydack 2000). Also, differences in regeneration dynamics and responses to disturbance between lowland forests and more seasonal hill forests require elucidation before the monocyclic vs. polycyclic debate can be resolved. An attempt in this regard is made below.

In order to explain divergent patterns in forest dynamics, notably regeneration, under variable edaphoclimatic conditions (tropical lowlands versus uplands, aseasonal versus more seasonal forests), the prototype of the metabolic performance trade-off model had been outlined (Seydack 2002). This model of tree metabolic responses has since been further refined and supported (Seydack et al. 2011a, b). According to the metabolic performance model there exists an intraspecific adjustment gradient in metabolic response modes of trees either predominantly confronted with relative constancy of resource availability levels (e.g. aseasonality of water availability and temperature) or those subject to variably fluctuating resource levels (Seydack et al. 2011a). The two associated metabolic performance modes accommodate the fundamental trade-off between capacity for peak or sustained metabolic performance and stress resistance (persistence capacity) by either adopting a resource level responsive (RR) or a resource level buffered (RB) metabolic mode.

Although the propensity to particular metabolic performance modes is generally determined by inherent species-specific characteristics, variable degrees of intraspecific plasticity facilitate the matching thereof to prevailing resource conditions differing in space or over time. The RR metabolic performance mode achieves the combination of capacity for peak performance rates with stress resistance (S capacity: survival and persistence through stress resistance), but at the cost of stress tolerance (sustained functional capacity for production and growth under adverse resource conditions). This involves a close dependency of performance rates with resource availability levels (narrow metabolic amplitude). At the cost of being constrained to a narrow metabolic amplitude, the RR metabolic performance mode combines resource-level sensitive, but highly responsive metabolic performance rates and stress resistance (GS capacity). This performance mode is particularly adaptive under constancy of temperature and moisture (TW) availability, as predominantly found under conditions of aseasonality in the tropics. The associated stress resistance capacity of this metabolic performance mode favours subcanopy stem density persistence (relatively high subcanopy forest matrix content, persistence of regeneration). Also, since growth performance is closely linked to TW resource levels (growth performance tracking resource surpluses), tree size is expected to be sensitive to resource availability levels in addition to tree age (Seydack et al. 2011a).

In contrast to the RR metabolic mode, the partial resource-level independence of the RB metabolic performance mode is based on a higher physiological capacity of adjustability (wide metabolic amplitude and stress tolerance). This involves achieving sustainability of performance under fluctuating and suboptimal resource levels (resource-level buffered and sustained metabolic performance: GT capacity). The enhanced stress tolerance capacity in terms of growth in the face of resource deficiencies (GT capacity) is at the cost of higher levels of maintenance respiration required to sustain adaptability to adverse resource conditions (Taylor 1989). The RB metabolic performance mode thus involves an accentuated trade-off between growth performance and stress resistance (variable juvenile survival capacities). This metabolic performance mode is expected to be typically prevalent under conditions of greater climatic seasonality with respect to water and temperature levels (Seydack et al. 2011a, b).

Responsive high performance rates at low S capacity costs (RR metabolic mode) can effectively support sustained vertically orientated growth (vertical growth mode; extended juvenile metabolism), whereas sustained resource use efficient growth performance (RB metabolic mode) is expected to be conducive to lateral growth (lateral growth mode; optimal expression during mature-stage growth). Vertical growth emphasis appears to be associated with the RR and lateral growth with the

RB metabolic response mode respectively, resulting in RR/VG and RB/LG functionality syndromes (Seydack et al. 2011a, b). In summary, trees in the RR metabolic mode setting function under relatively high GS capacity (persistence of subcanopy and submature trees; growth closely tracking resource surpluses) and vertical shoot growth emphasis; whereas RB metabolic performance is associated with GT capacity (sustained growth under resource stress and fluctuations) and lateral growth priority. Tree functionality under these divergent metabolic performance modes has implications for regeneration dynamics and yield optimization (Sect. 6.1).

Under tropical lowland conditions (e.g. Lowland Mixed Dipterocarp forests), matrix invaders have a propensity towards more RR metabolic performance favouring early metabolic performance and persistence of submature trees (GS capacity). This is associated with the spatiotemporally predictable occurrence of seedlings and recruits and a pioneerlike response to early full release as experienced under monocylcic systems. Vertical shoot growth is fast and recruit survival rates are high due to good stress avoidance capacity. Under conditions with higher degrees of resource fluctuations regarding temperature and water (e.g. Hill Dipterocarp forests, especially on ridges; semi-deciduous West African forests), forest matrix invaders tend towards RB metabolic settings and mature-phase metabolic performance. Regeneration is spatiotemporally sporadic and survival unpredictable on full canopy opening. Such released recruits appear to switch to mature-phase metabolism early in order to gain water stress tolerance and lateral crown formation is initiated at relatively low heights, resulting in relatively short merchantable boles (Poker 1993).

Early full release from overtopping canopies favours early metabolic performance which is more compatible with RR metabolic performance settings postulated to prevail in forest matrix invaders of the tropical lowlands. Due to the reasons explained above, vigorous monospecific plantation-like stands of commercially desirable forest matrix invader species may thus result under such conditions. In accordance with these scenarios, yield regulation systems where regeneration is fully released at an early stage are therefore considered successful and superior particularly in tropical lowland forests by those favouring monocyclic systems. However, silvicultural interventions are required in the plantation-like stands and sustainability of recruitment success and productivity of forest matrix invaders over multiple tree generations remains uncertain under such circumstances.

6 Perspectives on Yield Optimisation

6.1 Timber Yield Optimisation

Yield optimization is usually performed in relation to three groups of yield regulation parameters:

- Residual growing stock levels
- Harvest intensity (cutting cycle)
- Harvest maturity thresholds (e.g. MHDs)

The optimized RGS is such a growing stock which ensures the maximum long-term yield and to which the stand is reduced periodically through harvesting or silvicultural measures. It can be defined on a stand-basis (e.g. Bick et al. 1998; Ong and Kleine 1995) or involve the release of individual potential crop trees. It can also take the form of stipulating some stand normality involving RGS levels differentiated according to size class and species groups (Gadow 2000). Harvest intensity refers to the amount of timber harvested per unit of area at a specific harvesting occasion. Harvest intensity is optimized by determining a cutting cycle which is associated with the maximum sustained yield in interaction with the minimum economic cut and bearable felling damage. For example, Huth et al. (1998), performing simulation of logging scenarios, obtained the highest sustainable logging volumes with a felling cycle of 40 years when reduced impact logging was undertaken. With conventional logging the optimum felling cycle was 60 years. However, simulation of logging scenarios for monocyclic management revealed that felling cycles of 80–100 years are required in order to have a tolerable impact in respect of species composition and soils (Huth et al. 1998).

Most of the volume and value increment of individual trees takes place once the trees have already achieved substantial dimensions. The optimum yield realization strategy is therefore to stipulate relatively high harvest maturity thresholds, usually minimum harvestable diameters (MHD), but before substantial wood quality deterioration sets in. The optimum MHD does, however, not only depend on diameter growth patterns, but also on mortality rates (e.g. Alder 1992). As trees progressively increase in size, so do the number of trees lost due to mortality in the time required to reach higher diameters. The diameter at which volume growth of individual trees is maximized is therefore usually considerably higher than the cutting limit which takes mortality into account (Vanclay 1989). A dilemma now exists, since relatively low stipulated minimum harvest diameters (MHD) result in premature removal of the fastest-growing trees, whereas high MHDs are associated with losses due to mortality of disproportionately many trees before they reach harvestable dimensions (Seydack 2000, 2002). This dilemma is accentuated in tropical forests with predominance of the RR metabolic performance mode. MHD as an effective criterion for harvest definition relies on the assumption that a reasonably close correlation exists between tree diameter and age; and of declining vitality towards the higher diameters. However, congruent with the RR metabolic performance mode prevalent in tropical forests (Sect. 5.2), growth performance and tree size is substantially controlled by resource availability relative to age of trees. This implies that smaller trees are, on average, slower growing and larger trees are often the fastest growers (Seydack et al. 1995; Clark and Clark 1999; Brienen et al. 2006). MHD based harvesting therefore results in the disproportionate removal of the fastest growing trees. Furthermore, relatively old trees occur over most diameter classes. Mortality is linked to age and competition and therefore typically occurs over all diameter classes. The dilemma of trees harvested prematurely as a result of MHD optimization can be partly or largely overcome if maturity condition based harvest tree selection criteria are used which allow effective mortality pre-emption (Seydack 2000, 2002). In addition to an accentuated MHD dilemma, residual trees after each harvesting event grow relatively slower, causing timber yields at subsequent harvesting events to be below expectation (Bruenig 1996).

Substantial progress has been made in yield optimisation based on growth and forest dynamics modelling and harvesting scenario simulations. Both empirical growth models (e.g. Vanclay 1994; Ong and Kleine 1995; Gourlet-Fleury and Houllier 2000) and process-based forest modelling have been undertaken (e.g. Bossel and Krieger 1991, 1994; Ditzer 1999; Huth and Ditzer 2000; Ditzer et al. 2000). Some applications in yield parameter optimization are presented by Vanclay (1989), Ong and Kleine (1995), Kleine (1997), Alder and Silva (2000), Heuveldop et al. (2000), Van Gardingen et al. (2000), Kammesheidt et al. (2001) and Van Gardingen et al. 2006), among others.

The debate concerning the relative merits of poly- versus monocyclic yield regulation systems is eclipsed by the increasing predominance of secondary forest management situations and the advent of yield optimization through forest dynamics simulation modelling. Forest simulation modelling appears to be particularly valuable for degraded, secondary forests or forests under monocyclic management with strong and simplified contrasts. Most models apparently provide realistic results at low to medium levels of resolution (*vide* Huth et al. 1998), predicting good recovery of stand volume between suitably selected harvest events. Such a recovery does not apply to species composition, which usually differs from that of the primary forest state (Van Gardingen et al. 2006). Regeneration of all species is assumed in modelling, but this would require that the simulated forest stand be surrounded by forest with a normal species composition (Huth et al. 1998; Huth and Ditzer 2000). Simulated harvesting scenarios are thus not necessarily sustainable. This is but one example which illustrates that any predicted yield sustainability can only be expected to hold over multiple tree generations if and when a valid ecosilvicultural basis is incorporated in the modelling approach. The validity of conclusions depends on the validity of assumptions (Vanclay and Skovsgaard 1997). Despite considerable progress in the mathematical component of forest dynamics modelling, the ecosilvicultural dimension requires further attention if truly sustainable yield regulation directives are to be provided. Some ecosilvicultural facets relevant in this context will be discussed in the next section.

6.2 Complexities of Forest Growth

Most yield optimisation approaches, either taking the form of silvicultural release measures or pursued through modelling harvesting scenarios, imply crowding effects on tree growth either directly or indirectly. However, crowding effects in tropical and subtropical forests turn out to be complex and elusive. Firstly, the determination of crowding effects by pooling data of subcanopy and canopy cohorts may lead to results which are difficult to interpret. This is because crowding effects in these two cohorts are of a fundamentally different nature (Seydack 2000, 2002). Whereas canopy occupants respond more to site factors, the growth of subcanopy

Table 4.3 Multiple regression analysis of 10-year diameter increment against independent variables for Ironwood (*Olea capensis* ssp. *macrocarpa*) trees in Southern Cape forests Gouna, South Africa (Seydack 2002)

Independent variables	t-Values	P	Model
Canopy trees (DBH > 30 cm)			
Constant	0.2779	0.7812	$R^2 = 0.1962$
Site	5.5021	0.0000	F = 26.87
Tree condition impairment	−6.4108	0.0000	n = 319
Canopy tree density	−0.2662	0.7903	P = 0.0000
Subcanopy trees (DBH 10–20 cm)			
Constant	4.1129	0.0000	$R^2 = 0.0635$
Site	1.8098	0.0712	F = 8.44
Tree condition impairment	−3.7721	0.0002	n = 330
Canopy tree density	−2.3050	0.0218	P = 0.0000
Subcanopy trees (DBH 10–20 cm)			
Constant	3.7961	0.0002	$R^2 = 0.1058$
Site	2.9335	0.0036	F = 13.97
Tree condition impairment	−4.1276	0.0000	n = 330
Conspecific canopy tree density	−4.5792	0.0000	P = 0.0000

Site: based on crown grain ratings related to moisture availability due to topographic position and soil depth; Tree condition impairment: an impairment rating based on the presence of structural damage and degree of crown dieback; Canopy tree density: Number of trees > 30 cm DBH/plot; Conspecific canopy tree density: Number of conspecific trees > 30 cm DBH/plot

trees may well be impeded by overhead crowding (Table 4.3; Seydack et al. 2011b). This concurs with the findings that larger-sized canopy occupants usually remain unaffected by spacing operations (Dawkins 1959; Palmer 1975; Weidelt 1986, 1989), whereas good reactions to liberation thinnings of small to medium-sized trees are commonly reported (Maître 1987, 1992; De Graaf 1986; Woell 1989; Korsgaard 1992; Kammescheidt et al. 2003; Peňa-Claros et al. 2008). This has implications for yield optimization and the importance of viewing natural uneven-aged mixed species forests as two-layered systems (faster growing canopy occupants over partially waiting subcanopy trees) must be stressed in this context (Seydack 2000, 2002). The results in Table 4.3 furthermore demonstrate the negative impact of tree condition impairment on growth and the greater sensitivity of subcanopy trees to conspecific rather than all-species crowding.

As trees grow larger over time, the number of trees in the cohort declines according to the self-thinning rule (Westoby 1984). This progression over time to fewer larger trees may be curbed through competition between trees, especially in monospecific, even-aged stands. Silvicultural intervention in the form of thinnings then applies in order to achieve larger trees within a given time period. In uneven-aged, multi-species tropical and subtropical forests, the size structure (many smaller trees as against fewer larger trees) remains largely stable over time and is determined by site resource factors. This implies that many smaller stems are largely a consequence of poor site conditions and fewer larger stems or high basal

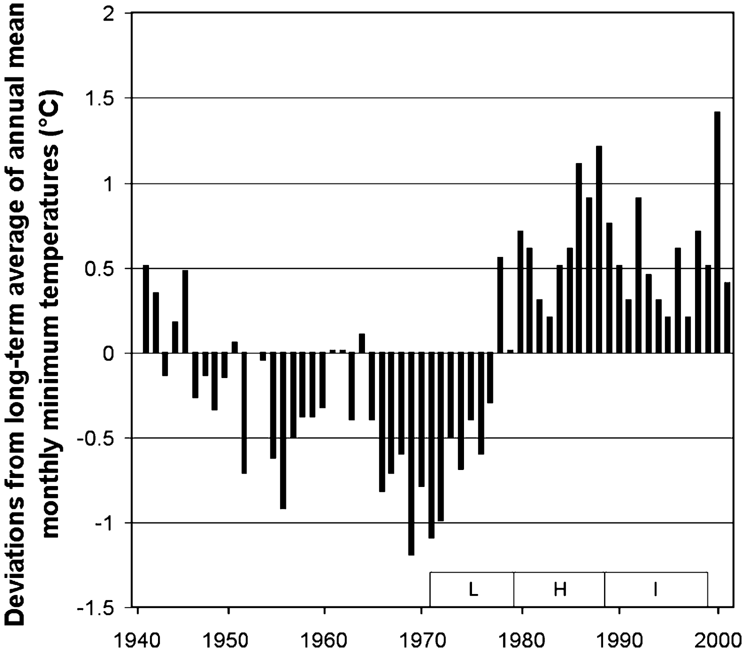

Fig. 4.2 Deviations from long-term average of annual mean monthly minimum temperatures (°C) at George, South Africa. Increment measurement periods L: 1972–1978 (low temperature status); H: 1979–1987 (high temperature status) and I: 1988–1997 (intermediate temperature status) (Seydack 2002)

area levels the consequence of better sites. Thus site and crowding effects are confounded. Competition indices often used, such as stem density within a specified radius around subject trees and sum of diameters or basal areas of neighbours (Moravie et al. 1999) are negatively and positively correlated with site quality respectively. Although this may not be a problem in growth prediction, it makes the differentiation between site and crowding effects rather problematic. In a study of tree growth in the Knysna forests (South Africa), it was found that certain trees within the same species had higher increments in the 1972–1978 increment period, whilst others had periodic increment peaks during 1979–1987. Peak growth performance of the former temporally coincided with relatively low minimum ambient temperatures and those of the latter increment period with relatively high minimum temperatures (Fig. 4.2: increment periods L and H respectively). This pattern was spatiotemporally congruent for most species and intraspecific growth response types (Seydack 2000) have been postulated: low versus high temperature growth response types (TGR types). The spatiotemporal congruence of these growth patterns over species suggests that site factors are involved and that species could be pooled for further analyses (Seydack in prep.).

Further analyses revealed that interannual stand increment fluctuations were indeed affected by site as defined by slope and aspect and that these interactions

Table 4.4 Multiple regression analysis of annual increment for 236 forest stands (all tree species pooled) against independent variables SLOPE and ASPECT (Diepwalle FVC Research Area, Southern Cape forests, South Africa: Seydack 2002)

Increment period	Interannual minimum temperature status	Trees 10–20 cm DBH		Trees > 30 cm DBH	
		t-Value	P	t-Value	P
Independent variable: SLOPE					
1972–1978	Low	−1.1942	0.2336	−1.2402	0.2161
1979–1987	High	3.7602	0.0002	3.0459	0.0026
1988–1997	Intermediate	0.4986	0.6185	0.6861	0.4934
Independent variable: ASPECT					
1972–1978	Low	−3.0423	0.0026	−2.7060	0.0073
1979–1987	High	2.2702	0.0241	−2.2875	0.0231
1988–1997	Intermediate	0.7467	0.4560	−2.6514	0.0086

Slope: in degrees; Aspect: relative light availability ratings: S, SE, SSE, SSW: 2; SW: 3; W, SWW, NWW: 5; NE: 6; NW: 7; N, NNE, NNW: 8. (Interannual minimum temperature status for periods: refer to Fig. 4.3)

were different for subcanopy and canopy cohorts in the case of aspect (Table 4.4). High TGR types were disproportionately concentrated on steeper slopes (Seydack in prep.), resulting in the positive correlation of stand growth rates on steeper slopes during the warmer increment period (Table 4.4; Fig. 4.2). Higher light availability levels appeared to be associated with higher temperature optima for growth (Assmann 1970). Thus, on the assumption that forest canopies on slopes are more broken and allow more light to reach tree recruits (less effective light extinction), such trees are more likely to be cued to high TGR types on steeper slopes. Canopy trees, irrespective of temperature growth response mode, had higher growth rates on southerly aspects (Table 4.4), presumably as a result of higher moisture availabilities associated with such sites within the study area. Subcanopy tree growth is limited by light availability levels, which are higher on northerly as opposed to southerly aspects. As a result, high TGR types predominate on northerly aspects and subcanopy tree growth is higher during the warmer period H (Fig. 4.2) on such sites. The inverse applies for the cooler increment period L (Fig. 4.2; Table 4.4). These results have been confirmed by analyses of stand-level tree growth for selected South African forests (Seydack et al. 2011b).

The greater sensitivity for light availability levels of high TGR types is shown in Fig. 4.3a, b. Good crown form (*re* Dawkins 1958) is important for growth performance of high TGR type yellowwood (*Podocarpus latifolius*) recruits, whereas the performance of low TGR type recruits is independent of crown form, but higher during the cooler increment period. A similar pattern was found for *Olea capensis* ssp. *macrocarpa* recruits with respect to crown position. Gourlet-Fleury (1992 op.cit. Moravie et al. 1999) showed that the original Dawkins code (with five modalities) was the most efficient index to predict diameter increment in light sensitive, but not for non-light sensitive tree species in the Guianan rain forest. The example presented here (Fig. 4.3) shows that divergent sensitivities to light as resource can also apply intraspecifically.

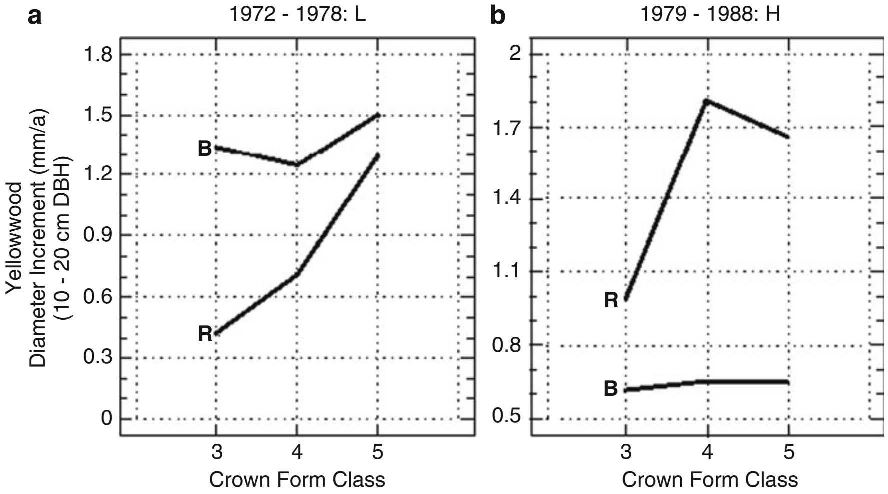

Fig. 4.3 Diameter increments (mm/annum) of yellowwood recruits (*Podocarpus latifolius*) in relation to crown form class for high (R) and low (B) temperature growth response types (TGR types) during the increment measurement periods of low (**a**) and high temperature status (**b**) respectively (Diepwalle, Knysna forests, South Africa). Temperature status of increment measurement periods relate to Fig. 4.2 (L and H), (Seydack 2002)

It has been established that many tropical tree species produce clear annual rings in areas with some degree of seasonality in rainfall (Worbes 2002). Thus, when feasible, tree ring analyses provide reliable data on life-time growth rates. Brienen and Zuidema (2006) demonstrate the advantages of tree ring analyses over growth rates obtained from permanent sample plot data in the context of projecting timber yields. With tree ring analyses the considerable variation in growth rates of tropical trees can be quantified, as well as patterns of autocorrelation revealed (Brienen et al. 2006). Such tree ring analyses reveal that growth rates are frequently temporally correlated within trees and that persistent growth differences exist between trees, i.e., among tree autocorrelated growth attributed to site-specific differences or differences among trees in crown area (Brienen et al. 2006). One result of such growth patterns is that trees of similar ages are often differentiated into a wide range of tree sizes; with larger-sized trees being the fastest growing trees.

From these interactive patterns it becomes apparent that the interpretation of tree growth, particularly in relation to crowding effects, is further complicated by interaction effects of site and interannual increment period, as well as divergent inter- and intraspecific response modes to resource factors. Numerous complexities characterize tree growth in tropical-type forests:

- Divergent site/resource factors affect subcanopy and canopy-level tree growth
- Species-specific divergence of growth responses to edaphoclimatic site and crowding factors

- Growth inhibition through proximity of conspecific neighbouring trees (Sect. 4.2)
- Unpredictable negative impacts of tree condition impairment on tree growth
- Site-crowding confoundment
- Acceleration of growth due to crowding release usually transient (Toledo et al. 2010) and unsustainable
- Adult tree size and age poorly correlated: larger-sized trees generally faster-growing (Sect. 6.1)
- Complex interactions between site and climate (Seydack et al. 2011b).

The complexities, as discussed, currently jeopardize efforts to adequately quantify the effect of crowding on tree growth. It is concluded that the role of crowding and competition as factors influencing tree growth in tropical and subtropical forests is complex and the effective use of these variables in yield optimization is subject to substantial uncertainties. However, the realization that canopy-level trees are generally fast growing and that by far most of the timber volume and value production takes place in canopy-level trees facilitates yield optimization allowing for relatively high RGS levels and thus timber harvesting in alignment with tropical forest dynamics (Sect. 5.1). This is in contrast with the beliefs of earlier forest scientists who considered primary mature tropical forests as unproductive, requiring domestication and transformation into earlier stages of succession.

6.3 Constraints of Economics in Tropical Timber Production

Pearce et al. (2003) noted that the profitability of uncontrolled and unsustainable high intensity logging constitutes a significant obstacle to sustainable timber harvesting. The financial performance of sustainable timber harvesting is usually considered marginal (Pearce et al. 2003), even for the highly productive south-east Asian forests (Van Gardingen et al. 2003). Reducing harvest intensities towards sustainable levels is generally considered to threaten economic viability (Bawa and Seidler 1998; Pearce et al. 2003). The generally high intensity timber harvesting systems implemented require silvicultural interventions in order to promote continued sustainability of timber yields. Such silvicultural interventions are however often considered economically unaffordable and seldom undertaken on a large scale (Whitmore 1990; Panayotou and Ashton 1992; Bruenig 1996).

Compatibility between economically viable natural forest management and conservation of biodiversity has been difficult to achieve (Bawa and Seidler 1998). However, although timber yields under sustainable systems are comparatively lower, non-timber values may more than offset the lower but sustainable yields (Pearce et al. 2003). Much of the tropical timber demand is for particle board, plywood and paper (Bawa and Seidler 1998) and should be harvested with land-use intensive systems (smaller areas, secondary forests). Following these constraints a clear differentiation between intensive and extensive land-use options emerges.

Intensive management (plantation-like secondary forest management) involves the economically viable production of commodity timbers. Extensive, but sustainable primary forest management should be geared towards the economically viable production of specialty timber and biodiversity conservation (Bruenig 1996).

7 Concluding Perspectives

Multiple and increasing resource demands on tropical and subtropical forests require that the full range of forest management situations on the RFM to CCF continuum need to be considered. Tree plantations, monocyclically managed secondary forests and forests managed under naturatistic polycyclic yield regulation systems all have a role to play in satisfying existing demands (Seydack 2002). Three main silvicultural scenarios (timber yield output/silvicultural input options) are identified: high output/high input (high intensity secondary forest management), high output/low input (conventional logging systems as generally practised) and moderate output/low input requirements (sustainable primary forest management with stipulations of timber yield regulation in alignment with tropical forest dynamics).

Conventional logging systems are characterized by comparatively high impact timber harvesting. As such they are subject to sustainability risks and usually require silvicultural interventions in order to remain sustainable. The required silvicultural operations are generally considered unaffordable and consequently neglected. Such high output/low input systems are thus ecologically unsustainable. High intensity secondary forest management for products in high demand requires substantial silvicultural input, but given favourable circumstances may be economically attractive (Emrich et al. 2000). Low intensity primary forest management involves selection cutting systems (notably naturalistic selection cutting systems) where stand regeneration and timber yield optimization are achieved through yield regulation directives in close alignment with tropical forest dynamics. Sustainability risks are avoided and silvicultural interventions are not required.

The moderate output/low input scenario represents continuous cover primary forest management for the production of high value timber in terms of species, dimensions and wood quality. In order to achieve this sustainably, timber yield regulation directives must relate to and be in alignment with the nature of tropical forest dynamics; thereby avoiding sustainability risks associated with high impact systems. This implies that yield regulation must provide for recruitment through an intact forest matrix, realization of extended large-tree productivity (relatively high harvest maturity thresholds which are as far as possible species-specific and site adjusted), relatively high residual growing stock levels and reduced impact logging practices. If this can be achieved, silvicultural interventions will not be required and the compatibility of timber harvesting with biodiversity conservation and the utilization of non-timber forest products will be facilitated.

An increased understanding of the facets of tropical and subtropical forest dynamics mentioned above is expected to facilitate the shift towards and effective implementation of continuous cover forestry yield regulation systems. Benefits of greater certainty of sustainability, lower managerial input and greater versatility and compatibility with multiple interests (conservation, forest inhabitants) can be expected to result from such continuous cover forestry. A central challenge for forest scientists now is to combine powerful simulation modelling tools with an improved understanding of tropical forest dynamics and yield science applications, thereby providing an extended ecosilvicultural dimension to yield optimization (Seydack 2002).

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Chapter 5
The Economics of Continuous Cover Forestry

Thomas Knoke

1 Introduction

The debate about the optimal silvicultural system continues. Controversial and partly emotional discussions about whether or not continuous cover forestry (CCF) would be superior to the worldwide practice of rotation forest management (RFM) characterized by repetitive cycles of clearfelling and re-planting (usually with monocultures) are also fed by results from normative research. Positive normative research has a long tradition in forest science and particularly in forest economics (e.g., Endres 1919). It is still a legitimate scientific desire to find rules for how forest owners and societies may achieve their objectives in forestry at best. Since Hartig (1800), who was an advocate of pure even-aged forests, much progress has been achieved in forest science. Numerous advances in forest economics and forest management were contributed in particular by the Anglo-American and Scandinavian scientific communities (see textbooks by Bettinger et al. (2009) or Amacher et al. (2009) for examples), but still the question of how to best manage a forest remains interesting.

RFM, endowed with numerous textbooks, formulas and models, with the ideas of a fully regulated (so called "normal") forests and the Faustmann formula (including many extensions) on its side, gained a lot of advocates in the past (e.g., Hundeshagen 1826; Judeich 1871; Wagner 1928; Baader 1945; Richter 1963). The tremendous practical success of this silvicultural system certainly benefited from its sound theoretical foundation and its relatively simple application. However, in contrast to Hartig and other promoters of even-aged monocultures, Cotta assumed already in 1828 that mixed forests might possibly be better performers than pure forests.

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_5,

While Cotta (1828) still believed that a mixed forest strategy would not be possible immediately because of practical reasons, Gayer (1886) developed the idea of mixed forests further to establish a new silvicultural paradigm. However, the mixed forest concept was not put into practice for a long time.

Eventually Alfred Möller (1922) added with his concept of CCF (in German "Dauerwald") another aspect to the discussion about silvicultural practices. According to him, forest managers should never clear-cut a forest stand because of ecological reasons. CCF is not equal to the classical selection forest ("Plenterwald"). The concept is much wider and includes any kind of silviculture that aims at permanent tree cover (if possible with trees of merchantable size). Thus, CCF includes the very specific "Plenterwald" system, which is traditionally practiced in France and Switzerland on a scientific basis (Biolley 1922). But it is not synonymous with "Plenterwald".

CCF (or, more generally, variants of uneven-aged silviculture) has not, however, been a real success story so far, if one considers the rather small worldwide area in which this silvicultural system is practiced. It has not only been seen inferior to RFM by many forest practitioners; it also was often hindered by forest administrations and forest laws at least until the 1980th. For example, the current forest policy and practice in Finland and Sweden still oppose CCF systems. In Finland, forest authorities had started a fight against selective cutting methods because of the concern that private forest owners were devastating their forests by applying this method (Siiskonen 2007). This development peaked in 1948, when a group of influential Finnish researchers published a declaration against the destructive impacts of selective logging on Finland's forests. Similarly, also in Germany there has been considerable resistance against CCF after World War II. The colleagues of Möller, for example Eilhard Wiedemann, a famous German professor in forest yield science, were successful in discrediting Möller's idea of CCF (Pommerening and Murphy 2004). It appears that concerns about low productivity, devastation of forests and poor economic performance by means of selective cutting methods still prevail in forest practice and science.

Interestingly, it was in Finland where economic studies (e.g., Pukkala et al. 2010; Tahvonen et al. 2010) questioned the current recommendations, which favor RFM and thinning from below, and have stimulated new discussion about CCF. Economic consideration may thus enrich the discussion about silvicultural practices with new and sometimes even surprising arguments. In this line, my aim is to provide an up-to-date overview about different approaches to analyze the economics of CCF (Sect. 2). I will then discuss the economic pros and cons of CCF (Sect. 3) to speculate about reasons for the skepticism against CCF in forestry practice (Sect. 4). Finally, Sect. 5 will mention some existing gaps in the economic analysis methods of CCF, which need to be filled to complete our still insufficient understanding about the economics of CCF. The hypothesis that guides my contribution is: "A deeper theoretical understanding of the economics of continuous cover forestry will rather reveal arguments in favor than against this silvicultural system."

2 State of the Art

The following discussion is limited to marketable products obtained from forest management, which is mainly timber. As a consequence, I refer to financial issues when I talk about economics. The reader should, however, keep in mind that a true economic analysis would comprise more than mere financial consequences. Pearce et al. (2003) make it clear that an economic analysis obtains the perspective of forest owning nations or even a worldwide point of view. All environmental and social consequences that have an impact on the well-being of humans should be considered in a deep economic analysis. Also, gains and losses outside the country where the forest is located may be considered. An attempt to include also non-marketable goods and services into a cost-benefit-analysis of CCF is presented by Price and Price (2008). Certainly a broader view of the provision of ecosystem services (including marketable and non-marketable goods and services) will become increasingly important for future studies. However, financial consequences are the most important ones for forest owners. As crucial ‘drivers’ of land-use decisions (Pearce et al. 2003) financial consequences will directly influence silviculture. It thus seems justified to concentrate on financial issues. However, although focusing on financial issues, I still use terms such as “economics”, “economic” and “economical” as they are intuitively understood.

A literature review reveals several approaches in analyzing the economics of CCF. While international studies reported in English language mainly focus on the stand-level, estate-level studies have been carried out in Europe frequently. However, the majority of studies at hand ignore uncertainty. Stand-level studies can be further distinguished methodologically (Table 5.1). We find:

(a) studies which compare a limited number of predefined treatments (CCF and RFM);
(b) studies which compare found steady states and seek investment efficiency; and
(c) studies which compute optimal forest management solutions dynamically.

The type-b-studies do not consider the transition phase from the initial state to the steady state or from a given steady state into another steady state. They usually consider an appropriately capitalized sustainable return produced over one cutting cycle and subtract the initial investment from the capitalized return. The initial investment is usually represented by the financial value of the standing timber

Table 5.1 Schematic of approaches applied in the economic analysis of CCF

	Stand-level	Estate-level
Uncertainty ignored	Predefined treatments (comparative); investment efficient optima; dynamic optima	Comparative
Uncertainty considered	Predefined treatments (comparative); stand structure and treatment follow from optimization	Comparative

volume (at the beginning of the considered period) which is necessary to produce the return. In contrast, type-c-studies explicitly consider the time dynamics of the transition phase. They do not use ad hoc specifications of forest management. When forest management is dynamically optimized the resulting prescription is conditional to the initial state and the specified objective function to be maximized. One could say that under this approach, treatments and stand structures follow predefined economic objectives of forest management. Type-c-studies may be seen as the most sophisticated among the known approaches to analyze the economics of CCF. The following overview will start with stand-level analyses and concludes with estate-level studies.

2.1 Stand-Level Analyses

International authors analyze the economics of CCF often at the level of forest stands. The three aforementioned economic approaches are approximately equally represented among the analyzed studies (Table 5.2). The results of the literature analysis will be structured according to the approach applied for economic valuation.

2.1.1 Comparison of Predefined Treatments

Tarp et al. (2000) compare predefined management strategies for European beech in Denmark. They find RFM being superior to CCF only when the considered stand is already close to the optimal rotation length. However, RFM with natural regeneration was also very profitable. In this case CCF was superior only when it achieved a price premium and/or bigger logs than RFM. Tarp et al. (2000) show the advantages of an early start of transformation from the even-aged into the uneven-aged CCF structure. In line with Tarp et al. (2000), Knoke and Plusczyk (2001) could prove that an early beginning of a transformation strategy in a Bavarian even-aged spruce stand is superior to RFM for interest rates of 2.6% and above. Knoke and Plusczyk's (2001) study considered an establishment of White fir (*Abies alba*) in small gaps starting with a stand age of about 40 years and continuing the regeneration process until an uneven-aged stand with four age cohorts was formed during a transformation period of about 80 years. Gap creation in combination with thinnings deliver regular and frequent net revenues and enhance the growth of remaining trees so that favorable economic results can be achieved. Knoke and Plusczyk (2001) also confirmed that the regeneration process with White fir is important to secure long-term continuity of net revenues. Hanewinkel (2001a) analyzed a transformation strategy for Norway spruce (*Picea abies*) in Baden-Württemberg and used European beech for stand regeneration. He showed that this strategy leads to increased costs when compared to RFM. However, these costs decrease with increasing interest rate. Ralston et al. (2004) were able to identify

Table 5.2 Selection of international stand-level studies on potential economic effects of CCF, where some use rotation forest management (RFM) as a reference; see also Roessiger et al. (published online) for a similar table

Authors	Method according to Table 5.1	Contents/results
Duerr and Bond (1952)	Investment efficient optimum	Authors propose a methodology of how to find the optimum stocking in a selection forest.
Adams and Ek (1974)	General dynamic setting, simplifications	Methodologically oriented study that seeks to optimize the transition process from an irregular into a target structure.
Chang (1981)	Investment efficient optimum	At higher interest rates CCF is superior to RFM: CCF produces small but frequent earnings, RFM leads to high but infrequent earnings.
Haight (1985)	Dynamic optimization	Optimal transition does not converge to investment efficient state.
Haight (1987)	Dynamic optimization	Clear-cutting and planting are suboptimal in economic terms.
Haight and Monserud (1990)	Dynamic optimization	Convergence to CCF by maximizing net present values (4% interest). Productivity of CCF is not less than the productivity of RFM.
Buongiorno et al. (1995)	Dynamic optimization	Investment efficient steady states are approached by means of goal programming.
Schulte and Buongiorno (1998)	Predefined treatments (comparative)	Authors report that CCF with Loblolly pine is economically superior to RFM at high interest rates and low stand densities.
Kant (1999)	Investment efficient optimum	With rising interest rate CCF converges into RFM. Caused by the ingrowth model RFM (continuous profits every 5 years).
Tarp et al. (2000)	Predefined treatments (comparative)	Transformation into CCF in beech stands as early as possible. RFM superior only if stand age is close to optimal rotation length.
Wikström (2000)	Tabu search and greedy heuristics	CCF is disadvantageous compared with RFM in all cases. The initial stand age (63 years) is beyond the financial maturity (50 years).
Knoke and Plusczyk (2001)	Predefined treatments (comparative)	CCF is advantageous to RFM under many basic conditions. RFM is superior only at very low interest rates.
Knoke et al. (2001)	Predefined treatments (comparative)	In CCF frequent operations (cuts) lead to a compensation of fluctuating timber prices. CCF therefore shows a lower financial risk.

(continued)

Table 5.2 (continued)

Authors	Method according to Table 5.1	Contents/results
Hanewinkel (2001a)	Predefined treatments (comparative)	Costs for transforming RFM into CCF decrease with increasing interest rates.
Andreassen and Øyen (2002)	Predefined treatments (comparative)	See Wikström (2000): RFM always advantageous (the analyzed stands are already financially mature).
Ralston et al. (2004)	Predefined treatments (comparative)	Some Douglas fir CCF strategies exist that show a small financial advantage compared with RFM.
Sánchez Orois et al. (2004)	Investment efficient optimum	CCF superior on poorer soils, RMF superior on richer soils.
Price and Price (2006)	Predefined treatments (comparative)	Under certain assumptions transformation from RFM to CCF is favorable.
Pukkala et al. (2010)	Investment efficient optimum	Increasing discount rates and decreasing site quality improve the superiority of CCF.
Tahvonen et al. (2010)	Dynamic optimization	For volume maximization: RFM appears to be superior to CCF. If regeneration and harvesting costs are considered, CCF turns out to be superior at least with increasing interest rates.
Chang and Gadow (2010)	Investment efficient optimum	Generalized Faustmann model allowing for varying residual growing stocks and number of years from one cutting cycle to the other.

economically favorable CCF strategies for Douglas fir (*Pseudotsuga menziesii*). Price and Price (2006) prove target diameter cutting (diameter-limit cutting) as a profitable strategy to carry out transformation strategies. We thus see that most of the analyzed comparative studies that were based on predefined treatments could identify situations in which CCF strategies would be economically superior to RFM.

2.1.2 Seeking Investment Efficient Optima

The method to seek investment efficient optima has a long tradition. Already Duerr and Bond (1952) used this method to determine the optimum stocking for a selection forest. Here, the stumpage value of a forest is considered an investment, which produces future net revenues. Those occur periodically and are appropriately capitalized. The optimal growing stock volume (or stumpage value) is then found where the difference between capitalized net revenues (mostly considered constant and occurring at the end of the cutting cycle) and stumpage value is at the maximum. This approach is comparable to the famous “Gordon-model” used to optimize harvest intensities in fisheries, where the growth (reproduction) rate of fish populations is density dependent (Gordon 1953). Also Chang (1981) found that CCF has the tendency to become advantageous from an economic point of view when interest rates increase. However, both the length of the cutting cycle and the level of the residual growing stock are sensitive to changes in the stumpage prices of trees selected for harvest, in the stumpage prices of the residual growing stock, and in the interest rate (Chang and Gadow 2010). Small but frequent profits delivered by CCF are able to outcompete the large but rather infrequent profits produced by RFM. Even more clearly in favor of CCF, Chang’s (1981) results are reproduced by Pukkala et al. (2010) in an up-to-date study for Finland.

2.1.3 Seeking Dynamic Optima

Studies carried out in a dynamic framework have shown that optimal transition strategies would not necessarily converge into static investment efficient optima (Haight 1985). Despite this finding, the fundamental result that CCF is often economically superior to RFM is also, and even more convincingly, underlined by dynamic optimization approaches. When considering the management path from one state to another, it becomes evident that the optimal management would often converge to CCF. While Adams and Ek (1974) still used many simplifications in their approach, Haight (1987) already proved that dynamic optimization would not favor clear-cutting and planting. Later, Haight and Monserud (1990) analyze “any-aged” silvicultural management problems and point out that (at an interest of 4%) the management strategies always converge into CCF. Haight and Monserud (1990) also found that optimized CCF does not result in reduced volume production. Although especially in Finland great skepticism against CCF has been evident among the forest authorities (see Siiskonen 2007), also there the economic advantage of CCF over RFM was confirmed in a recent study (Tahvonen et al. 2010), which gained much public attention in Scandinavia. Using a sophisticated dynamic analysis (nonlinear programming solved with an interior point algorithm combined with a special strategy to avoid local optima) Tahvonen et al. (2010) found superior economic results for CCF to be almost universally valid. Only if the applied interest rate is very low, RFM appeared to be better.

Summarizing the above results we may conclude: If we avoid the application of artificial management strategies, thinning rules, silvicultural production programs or steady states to be achieved, an economically optimized management appears to have the tendency to (almost always) converge into CCF. Later we have to discuss, however, whether practical management would be possible without using simple silvicultural rules. Finally, we also have to note that CCF as resulting from dynamic economical optimization is not necessarily a “Plenterwald”. CCF is, from an economical point of view, every management which avoids large scale clear-cuts and tree planting.

2.1.4 Studies Revealing Disadvantages for CCF

Among the studies of all three methodological approaches of economic analysis I found very few examples which show that CCF is unfavorable. Wikström (2000) as well as Andreassen and Øyen (2002) found that the implementation of CCF is economically inferior to RFM without exception. In the case of Kant's (1999) study the results depend heavily on the interest rate applied. However, here the interest rate plays against CCF, contrary to the results of the other studies mentioned earlier. Studies which are in conflict with the hypotheses of an economically superior CCF are particularly important as they show limitations and also the need for further research. We shall explain and discuss these results later in Sects. 3.4 and 5.

2.2 Estate-Level Analyses

When considering the level of forest estates (see also Knoke 2009), we find comparative analyses from Europe. Mostly, these comparisons prove an economic advantage of CCF systems over RFM. For the case of CCF the analyzed forests mainly consist of Silver fir, Norway spruce and European beech, with a size structure typical to a classical "Plenterwald" (Table 5.3). However, as Hanewinkel (2002) already pointed out, the differences revealed from estate-level analyses are often not only caused by the applied silvicultural system. Leaving the basic concerns with estate-level analyses aside for a while, the analyzed studies may nevertheless help us to derive some basic economic characteristics of CCF.

One of the first comparisons was carried out by Ammon (1951) based on his experience in Swiss forests. He found average net revenues from CCF being 57% higher when compared to RFM. Mitscherlich (1952) derives from experimental areas in the German Black Forest a 14% greater timber price for logs produced in "Plenterwald" plots when compared to logs from even-aged plots. This difference could be traced back mainly to the bigger log dimensions produced under CCF. Mayer (1968) derives as much as 333% greater profits from CCF when compared with RFM (small scale clear-cutting system used as a reference) for the situation in Austria. A model calculation carried out by Siegmund (1973) for a hypothetical forest in Baden-Württemberg reveals 45% greater net revenues for the CCF system. Siegmund (1973) argues that this result can be obtained mainly because the CCF system allows a higher proportion of coniferous tree species when compared to RFM. Results comparable to those obtained in the mentioned studies are presented by Schütz (1985), who used data from experimental plots in Switzerland, Schulz (1993) for the German Black Forest and Knoke (1997) for the German Bavarian Forest. Hanewinkel (1998) obtained similar results for both silvicultural systems, when elements of CCF, such as diameter limit cutting, are also applied to RFM (Table 5.3).

Knoke (1998) presents model-based calculations where hypothetical CCF and "fully regulated, normal forests" for the case of RFM are compared. In contrast

Table 5.3 Selection of estate-level studies comparing CCF with even-aged silviculture (RFM) forests, a case for which only European studies (German, Swiss and Austrian) were found (a similar table can be found in Knoke 2009, however, in German language)

Author	Economic criterion	Ratio economic criterion, CCF to RFM in %	Comment
Ammon (1951)	Net proceeds per ha and year	157	Experience based estimation.
Mitscherlich (1952)	Average timber price per m³	114	Data from sample area, results not related to area.
Mayer (1968)	Profits and losses, uncertainty considered	433	Model based calculation. Small scale clear cut leads to losses due to risks of 25%, in CCF the loss is only 5%.
Siegmund (1973)	Earnings minus costs	145	Model based calculation. Reference = small scale clear cut.
Schütz (1985)	Net timber sales per ha and year	About 120	Data from sample area is being compared to yield table.
Schulz (1993)	Net timber sales per ha and year	132	Calculation of options with real timber assortments gained under equivalent felling volumes (8.1 m³/ha/year).
Knoke (1997)	Net timber sales per ha and year	113	Calculations for both silvicultural systems based on records of timber assortments over 31 years. Analyses of a real enterprise based on the felling value planned for 1972. Annual harvests in CCF around 16% below the felling value of RFM.
Knoke (1998)	Annual value growth	141–167	For calculations based on a growth modeling for different rotation lengths (RFM) and target diameters (CCF). Both silvicultural systems have identical financial value of timber stocks.
Mohr and Schori (1999)	Totals per m³ (costs for administrative personnel and certain input by entrepreneurs not included)	Advantage of CCF: ∼20% per m³ timber	Comparison of “Plenterwald” and “Femelschlag” system based on operational accounting sheets.

(continued)

Table 5.3 (continued)

Author	Economic criterion	Ratio economic criterion, CCF to RFM in %	Comment
Hanewinkel (1998)	Earnings minus costs, uncertainty considered	CCF similar to RFM, if no risk is considered in RFM	Model based calculation with high amount of older timber in both silvicultural systems. RFM based on growth model; concept of diameter limit cutting applied in RFM, very long phase of diameter limit cutting (age 90–140), CCF as empirical model.
Hanewinkel (2001b)	Earnings minus costs	286	Very high amount of pruned timber in CCF.
Hanewinkel (2002)	Literature review. Results: Comparisons carried out so far are inadequate. Either enterprises or forest stands lack comparability or judgmental assumptions within the model calculations were made. The author suggests comparing both silvicultural systems under optimal management.		

Table 5.4 Simulated value increments for comparable hypothetical fully regulated even-aged and uneven-aged forests (Knoke 1998, older currency, DM, converted into Euro by the factor of 1/1.95583)

	Fully regulated even-aged forests after Pretzsch and Kahn (1996)				Uneven-aged forests (Knoke 1998)			
	Rotation period (years)				*Maximum diameter* (cm)			
	80	100	120	130	*51*	*60*	*78*	*82*
Average value increment (€/ha/year)	291	355	376	379	423	502	618	633
Average financial stumpage value (€/ha)	4,752	9,410	13,882	15,996	5,017	8,993	11,413	16,025
Average relative value growth (%)	6.1	3.8	2.7	2.4	8.4	5.6	4.3	3.9

to other studies, this study considers (more or less) identical stumpage values represented by the standing timber volumes which are needed to produce the analyzed value increment for the compared silvicultural systems. In doing so, the costs for the capital tied up in the standing timber stock are kept equal for both systems, an aspect which is often excluded when comparing silvicultural systems at the estate level. Under this setting, the CCF systems produce 41–67% greater value increments when compared to the RFM systems (Table 5.4).

Mohr and Schori (1999) compared two management systems which both may be classified as CCF systems. Both authors compare the “Plenterwald” system – which is basically a single tree selection system based on more or less J-shaped diameter distributions – to the “Femelschlag” system. “Femelschlag” may be roughly characterized as a group selection system, which results in long periods of regeneration, and structures which appear uneven-aged during the regeneration phase. However, after 30–50 years the regeneration periods come to an end and all financially mature crop trees are harvested. This final felling of large-sized trees eventually results in a more or less age-homogenous stand structure. Mohr and Schori (1999) show superior economic characteristics for the “Plenterwald” over the quite similar “Femelschlag” forest. The “Plenterwald” shows lower costs for stand establishment and tending, leading to 25 Swiss “Franken” higher price (around 20% of the gross timber price) per cubic meter harvested in the “Plenterwald” than in a “Femelschlag” forest (Table 5.3).

Hanewinkel (2001b) highlights another aspect which is particular to CCF. He points out that CCF allowed a very high amount of pruned large-size logs in his study, which may be due to greater single-tree resistance of CCF forests against natural hazards. The pruned logs achieve very high timber prices, which, in turn, lead to net revenues of CCF 186% greater than those of even-aged forests used as a reference (Hanewinkel 2001b, see Table 5.3).

We may thus conclude from the presented evidence in estate-level studies that CCF systems may enable greater log dimensions, produce more valuable timber than RFM, and save costs for tending and establishment. Furthermore, wider growing space allocated to dominant trees in CCF systems compared to often densely stocked RFM stands leads to better profitability of single trees, so that greater value increment may be produced with a given capital investment. However, the finding of greater log dimensions and more valuable logs provides greater resistance of CCF systems against natural hazards when compared to RFM systems. Whether this precondition is actually met will be explored later.

2.3 Studies That Consider Uncertainty Are Rare

The above literature analysis reveals that the important aspects of risk and uncertainty are rarely addressed. Among the stand-level analyses only Knoke et al. (2001) consider timber price volatility and showed that a CCF strategy leads to a special diversification effect. Frequent harvesting operations carried out at different points in time lead to a more even distribution of timber harvests and provide an effective protector against price volatility, so that the standard deviation of net present values is lower under CCF as compared to RFM. Under the estate-level studies only Mayer (1968) and Hanewinkel (1998) integrate the possible effects of risk assuming different damage probabilities for the CCF and the RFM systems. Overall, we may however conclude that the impact of risk and uncertainty on the economic attractiveness of CCF and RFM has not been studied extensively so far. Particularly the perspective of a risk-avoiding decision maker has not until now been addressed in the available studies, except for a first attempt by Roessiger et al. (published online), where stand structure and treatment follow from optimization.

3 Economic Pros and Cons of CCF

This chapter concentrates on the role that time may play in the economic analysis of CCF, on the consideration of financial maturity, diversification effects and risks as well as on issues summarized under the “economies of scale”.

3.1 The Role of Time in Analyzing CCF

This section discusses the role of time in economic analysis. The discussion and illustration follows Knoke (2009). Time is mighty in economics. Most of the international studies we have considered in the previous chapters use the practice of discounting to integrate the effect of time on the value of the payment when net revenues occur. Discounting is employed to express the time preferences of decision makers in balancing out the present and future benefits. Summing all discounted net revenues expected now and in future (where the sum is called net present value, NPV) is thus commonly used to quantify the economic performance of forest management. Equation 5.1 formalizes the procedure for an unlimited period of time, in which case the NPV is called "soil expectation value" (SEV) or "willingness to pay for land" according to Martin Faustmann (1849):

$$SEV = \sum_{t=0}^{\infty} n_t \cdot q^{-t} \tag{5.1}$$

where n_t is net revenue obtained in year t and q is equal to $1 + \text{rate}/100$ (rate is interest rate in percent). One may approach an unlimited time horizon by considering very long periods (let's say 1,000 years) until the change in NPV with increasing time increments approaches zero. Martin Faustmann (1849) proposed an alternative in computing the sum of all net future values of forest management activities, where the sum is subsequently capitalized appropriately as a constant periodical rent, occurring at the end of every rotation, T. Future values are computed by means of compounding, while present values follow from discounting. Equation 5.2 represents the basic idea in a highly simplified version, where $V(T)$ as the stumpage value of the crop trees harvested at time T represents the total future value of forest management activities – meaning that establishment, tending, thinning and overhead costs are ignored.

$$SEV = \frac{V(T)}{q^T - 1} \tag{5.2}$$

where $V(T)$ is the financial value of crop trees after T years (stumpage value) and T is the rotation length (years). If we, however, could receive constant annual net revenues, a, we obtain SEV by means of Eq. 5.3.

$$SEV = \frac{a}{r} \tag{5.3}$$

where a is constant annual revenue and r is interest rate.

For a high SEV it is crucial how early and how frequently net revenues occur. Early and frequent but low net revenues may well outweigh late and high net revenues, depending on the interest rate applied. This finding will be explained with an example. Consider a simplified RFM that starts with natural regeneration and leads periodically, always after a rotation of 40 years, to a net revenue of 8,000 €/ha (this shall be the stumpage value of the crop trees, $V(T)$). We may thus compute 200 €/ha/year as the average value increment, $i_v(T)$, from dividing 8,000 €/ha by 40 years. The SEV computed with an interest rate of 0.02, as formed by the periodical rent of 8,000 € every 40 years according to Eq. 5.2, amounts to 6,622 €/ha. Interestingly, constant annual net revenues of only 132 €/ha/year (Table 5.5)

Table 5.5 Financial data for a hypothetical forest stand under two rotation periods

Rotation (T) (years)	Stumpage value crop trees $V(T)$ (€/ha)	Average annual value increment, $i_v(T)$, to build up $V(T)$ (€/ha/year)	Soil expectation value SEV (€/ha)	Constant annual net revenue, a, to form SEV (€/ha/year)
40	8,000	200	6,622	132
70	23,450	335	7,818	156

would already be high enough to form the same SEV (this follows from Eq. 5.3). A forest management that results in immediate and regular net revenues would thus need to produce only 66% of the average value increment of an RFM system to produce an identical SEV. Often, forest managers try to maximize the average value increment during the rotation. However, this intention ignores the important implications inherent in the concept of "time value of money". The time value of money has to be considered, unless we assume that money is an unlimited resource provided without cost.

It is clear that the above example is quite hypothetical, as it is impossible to create net revenues right from the beginning of forest production, when a young forest stand has just been established (we might, however, consider growing some annual agricultural crops in the early years or sell Christmas trees soon). However, the example shows how important early and regular net revenues may be, if we acknowledge the time value of money appropriately. Therefore, if it is possible to obtain earlier and more regular net revenues by means of CCF through regular thinning operations and/or regeneration harvests, the profitability of forest management may benefit substantially when compared to a clear-cutting system without thinning operations. However, to fully utilize the beneficial effect of creating an early starting and continuous series of net revenues it is important that money is made not only by harvesting trees. This was, for example, shown by Schulte and Buongiorno (1998) who confirmed that high grading (removing the best individuals) leads to great long-term disadvantages. In fact, it is crucial that the remaining trees will benefit from every harvesting operation, by increased growth or resistance against hazards, as it happens for trees at the edges of small gaps created to start advance regeneration and also as an effect of thinning operations, which deliver net revenues and promote remaining trees at the same time.

The above demonstrated advantage of immediate and regular net revenues even increases when the rotation of the RFM system extends. In our example, the maximum SEV from RFM is not obtained with a 40-year but with a 70-year rotation (Fig. 5.1). For a 70-year rotation the stumpage value of crop trees amounts to 23,450 €/ha giving a SEV of 7,818 €/ha (Table 5.5). While the average annual value increment is 335 €/ha/year, a regular annual payment of only 156 €/ha/year would give the same SEV. This means that an annual value increment, which could be realized every year, of only 47% would be great enough to produce the same SEV as the RFM system. Consequently, significant losses in value increment may be compensated for by an advantageous distribution of net revenue flows over time.

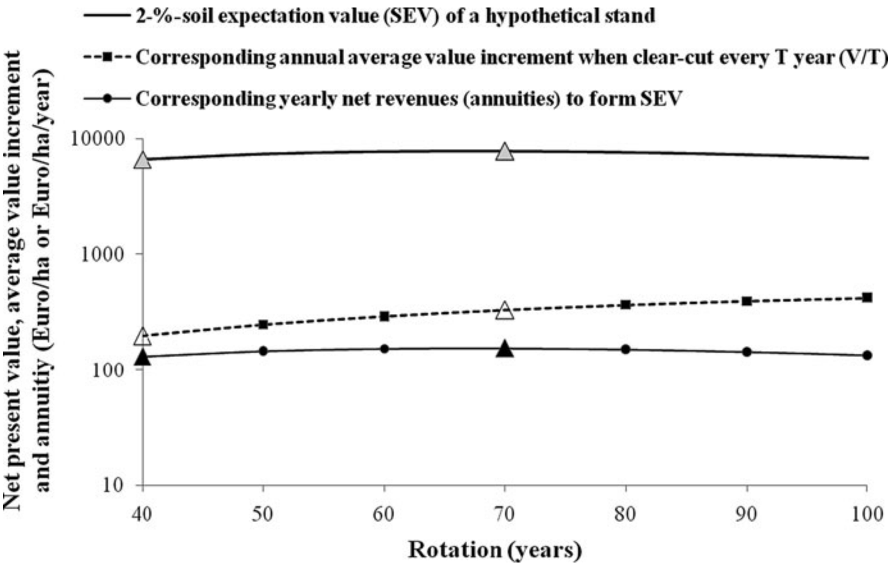

Fig. 5.1 Soil expectation value (SEV), average value increment produced in an even-aged stand and necessary annual net revenues to form the same SEV as the even-aged stand, possibly provided by uneven-aged management (triangles mark values contained in Table 5.5)

We may also learn from the above example that a possible transformation from RFM to uneven-aged CCF systems should start as early as possible. The earlier we may start creating regular and frequent net revenues the more profitable forest management will be. Vice versa we may say that starting the transformation process late, when already a high value of crop trees has been built up, is not optimal. In such a situation the transformation process would leave many valuable trees uncut to provide shelter for young trees in CCF. However, this means an expensive investment which is often not justified by the expected relative value increment of the older trees (see Wikström 2000; Andreassen and Øyen 2002) and which exposes valuable trees to high risks. The timing of the silvicultural transformation process is thus very important.

One may now argue that the above consideration would hold true only for a stand level analysis, because a fully regulated even-aged forest (a so called "normal" forest) would also provide regular annual net revenues in a perfect way. This is of course true. However, first we must acknowledge that we will hardly find a fully regulated forest in the real world (this model is rather a theoretical concept). It is also important that we have to consider the transformation phase; even if we have a fully regulated even-aged forest it may be advantageous to opt for a transformation towards an uneven-aged system. When starting transformation in suitable (possibly young) age classes and increasing the SEV for the respective stands through earlier and more regular net revenues in conjunction with simultaneous promotion of remaining trees, the total forest estate would also gain value.

3.2 Addressing Financial Maturity

Tree profitability varies greatly between individuals in natural forests and also in most of the artificial, man-made forests, where trees have not been genetically altered to become homogeneous in growth (Table 5.6). Profitability depends mainly on the tree's value increment (to a large part driven by the tree's growing vigor) and on the stumpage value it has already achieved (which is closely correlated to its diameter at breast height (dbh)). These characteristics define how profitable it would be to leave a tree in the forest to further produce timber. In a simplified form, which leaves neighbor effects and the opportunity costs of forest land aside, we may use Eq. 5.4 to compute the profitability of a single tree, *i*, in percent.

$$i = \left(\sqrt[t]{\frac{v(T+t)}{v(T)}} - 1 \right) \cdot 100 \tag{5.4}$$

In Eq. 5.4, t is the time we consider as the planning horizon (which in our example is 5 years, but may also be any other time period), $v(T)$ is the actual stumpage value of the tree and $v(T+t)$ is the expected future stumpage value of the tree. From Eq. 5.4 it follows that it is much more economical to harvest trees with low profitability, i, before those trees with higher profitability. Trees with low profitability will show a relatively large actual stumpage value, $v(T)$ in comparison to their expected future stumpage value, $v(T+t)$. Thus the immediate harvest would result in a relatively small future value increment loss. However, when clear-cutting all trees at once, the individual tree profitability is disregarded. Thus, some trees will be cut at the correct time, others will be harvested too early and some too late. It is thus clear that felling all trees at once is likely to result in lost revenue.

This finding shall be illustrated with an example (Table 5.7). Let us consider a possible cutting cycle of 5 years. The sample of 18 trees in Table 5.6 has achieved already financial maturity on average, if 2% is the required rate of interest. Clear-cutting all the 18 trees would realize an income of 2,798 Euro (Table 5.7). The stumpage value of all clear-cut trees would accumulate to 3,089 Euro when invested at 2% over the next 5 years.

However, viewing every tree individually we realize that 10 trees still produce a relative value increment beyond 2%. CCF would apply a selection system in this case and leave those trees performing above threshold uncut, for example during the next 5 years. Only those eight trees with profitability below 2% would be harvested in a single tree selection system. Those would yield a stumpage value of 981 €, which would accumulate to 1,083 € when invested at 2% over 5 years. The remaining trees would grow to a total value of 2,059 € during the 5 year period. Both amounts, the stumpage value of the trees which is realized immediately and invested at 2% (1,083 €) and the value accumulated from the remaining trees by means of natural growth (2,059 €) sum up to 3,142 €, which is 53 € more when compared to the amount obtained by clear-cutting. It follows that clear-cutting would reduce profit by 18% in our example.

Table 5.6 Financial data for 18 hypothetical trees

Species	Crown size	Age (years)	Dbh (cm)	Stumpage value today $v(T)$, €	Stumpage value in 5 years $v(T+5)$, €	Profitability (%) acc. to Eq. 5.4	Cutting ($i < 2\%$?)	Received tree value after 5 years
Spruce	Normal	100	45	87.9	99.3	2.5	No	99.3
Pine	Small	140	85	295.4	306.9	0.8	Yes	0.0
Spruce	Normal	115	50	114.3	127.5	2.2	No	127.5
Spruce	Big	110	60	177.1	203.2	2.8	No	203.2
Pine	Normal	135	45	55.6	59.9	1.5	Yes	0.0
Spruce	Small	160	55	144.0	152.4	1.1	Yes	0.0
Spruce	Big	95	75	296.5	339.8	2.8	No	339.8
Pine	Big	90	70	183.3	205.3	2.3	No	205.3
Pine	Normal	160	60	123.3	130.4	1.1	Yes	0.0
Spruce	Big	100	80	343.0	390.9	2.7	No	390.9
Pine	Big	80	75	217.7	245.4	2.4	No	245.4
Spruce	Normal	140	55	97.8	104.4	1.3	Yes	0.0
Spruce	Big	150	70	253.3	284.3	2.3	No	284.3
Pine	Small	150	35	25.3	25.6	0.3	Yes	0.0
Spruce	Small	150	45	87.9	92.9	1.1	Yes	0.0
Pine	Normal	90	45	55.6	61.9	2.2	No	61.9
Pine	Small	140	65	151.8	157.3	0.7	Yes	0.0
Spruce	Big	140	45	87.9	101.6	2.9	No	101.6
Sum/average				2,798	3,089	2.0	Stumpage value: 981	2,059

Table 5.7 The cost of clearfelling vs. not clearfelling

	Immediate harvest		Future value in 5 years (€)				
Silvicul-tural system	No. of trees	Net revenue (€)	Value of remaining trees after 5 years	Immediate net revenues invested at 2% over 5 years	Total	Total appreciation[a] (€)	Difference clear-cut minus selection system (€)
Clear-cut	18	2,798	–	3,089	3,089	291	−53 (−18%)
Selection	8	981	2,059	1,083	3,142	344	

[a] Difference in the available amount of money (including interest) after 5 years and initial amount of money

The limitations in our simplified example include: We have ignored the reinvestment which is possible after clear-cutting by means of establishing a new forest stand immediately. We thus ignored the suppressed soil rent when leaving trees growing. However, for the long production periods which are common in Europe, the suppressed soil rent may well be ignored. On the other hand, we considered only that clear-cutting would harvest trees too early. However, it may also be that trees are harvested too late. In our example eight trees show profitability below 2%, some close to or below 1%. Those trees would most likely have been harvested much earlier under a consistent CCF regime. Investing the revenue from these earlier harvests at 2% would already have accumulated more money than the value the trees could have accumulated if left to grow. Also, it is very important to consider another limitation that we have ignored: The growth of a given tree may reduce the growth of neighboring trees (Härtl et al. 2010). In more sophisticated and detailed studies such effects should be considered. The tree's profitability would decrease on average when integrating the mentioned effects. Still, considerable variation between individual tree profitability will remain. Therefore the principle of wasting money when cutting all trees simultaneously remains valid, even within a more complex consideration. Under which circumstances the above described beneficial effects of the selection system may be negated by other effects (particularly effects summarized under the term "economies of scale"), which might push RFM into an economically advantageous position, will be discussed later.

3.3 Stand Resistance Against Natural Hazards, Managerial Flexibility and Risk Diversification

An important precondition for selective harvesting under CCF is great single tree resistance against natural hazards. Although we do not have extensive comparative studies about this aspect, we often find such experience from practitioners that long crowns of conifers assure greater vigor enhancing the resistance against bark beetles and storms. Empirically, this effect has been confirmed by Lenk and Kenk (2007). A high single tree resistance allows one to adjust harvests and sales to the market situation. In a study for the Bavarian Forest Knoke (1998) showed that this strategy applied in a "Plenterwald" enhanced yearly net revenues by about 20 €/ha. This is one of the rather few empirical confirmations, achieved by means of real estate-level data, that flexible, market adapted harvest policies may actually be successful.

In addition to the above mentioned empirical observation, many theoretical studies prove financial advantages of market-adjusted timber sale operations (Brazee and Mendelsohn 1988; Knoke and Peter 2002 among others). Regarding Norway spruce and European beech (*Fagus sylvatica*), Knoke and Wurm (2006) confirm that net present values of robust European beech stands could be raised by 30% if harvest operations are carried out depending on timber prices, while for Norway spruce this was possible to a lesser extent only, assuming that spruce is affected by the frequent occurrence of storm damages, as is often observed in RFM. Single tree resistance can therefore be highlighted as a key factor to success in CCF.

In a recent study, Hyytiäinen and Haight (2010) analyzed the effect of fire on the attractiveness of CCF. They confirm the superiority of CCF over RFM with increasing interest rates, which has been reported in several other studies before. However, the advantage of CCF over RFM decreased when damages by fire were modeled. From the perspective of fire hazards great tree size diversity appears to be a disadvantage.

Another aspect of CCF is its potential to diversification of risks. This effect has been documented for even-aged mixed forests in various studies (e.g. Knoke et al. 2005; Hildebrandt et al. 2010; Beinhofer 2010a). Mixing tree species in RFM leads to diversifying risks of natural hazards, as different tree species do not show the same susceptibility to a particular hazard (see von Lüpke and Spellmann 1999). Moreover, there exists a positive effect of broadleaves on the resistance of conifers, which promises great financial improvement (Knoke and Seifert 2008). Finally, timber prices for different tree species and logs are sometimes only weakly correlated so that another effect of diversification may be utilized by means of mixing tree species (Beinhofer 2010b). However, these effects would be obtained also by mixing tree species in RFM. Yet, CCF enables another diversification option. As we mentioned earlier, CCF distributes timber harvests more or less uniformly over the considered time periods, while RFM often realizes most of the produced timber in a final clear-cut. In other words, we can say that CCF diversifies harvesting times greatly so that the risk of low timber prices is diversified. This effect leads to a significant reduction in the volatility of NPV (Knoke et al. 2001; Roessiger et al. published online). However, still too few studies have investigated the effects which risk and uncertainty may have on the choice of the silvicultural system to be applied. In particular, a simultaneous consideration of the effects of mixing tree species and of diversifying harvesting times is lacking.

3.4 Limitations of CCF and Modeling of CCF

3.4.1 Limitations of CCF

Regarding the findings presented before, it may be surprising that CCF is applied only occasionally in temperate and boreal forests. I see factors which we may summarize under the concept of “economies of scale” as important reasons for this situation. If it is possible to substantially reduce harvesting costs, overhead costs and costs for infrastructure, such as forest roads, by means of clearfelling systems, one can expect that these effects would over compensate for the economic advantages of CCF. Unfortunately, we have only little empirical evidence about these scale effects.

Some empirical evidence exists for the case of harvesting costs. In a comprehensive study for Bavaria Pausch (2005) confirms that cost differences between near-natural forest systems and highly mechanized silviculture may be only moderate.

Table 5.8 Increase of logging costs when avoiding clear-cuts (according to Price and Price (2008), with alterations, British Pounds converted by a factor of 1.174 into €)

Treatment	Costs (€/m^3)	Cost increase compared to clear-cut (€/m^3)
Clear-cut	10.73	
Low thinning	15.26	+4.53
Group felling	14.56	+3.83
Shelterwood felling	13.07	+2.34
Single tree selection (creaming)	11.53	+0.80

He estimated logging costs under near-natural forestry to be 2.5 €/m^3 higher, on average. Cost differences greater than those reported by Pausch (2005) were found by Price and Price (2008) in their cost-benefit-analysis of CCF (Table 5.8).

But would these cost increases compensate for the possible advantages of CCF? Let us take the cost increase derived by Pausch (2005) and assume the sustainable harvest at the estate-level to be 8 m^3/ha/year. It follows that we would have 20 €/ha/year higher logging costs under CCF as compared to RFM. However, Knoke (1998) derived by model calculations between 132 and 241 €/ha/year higher value increments when CCF in the form of a "Plenterwald" was compared with RFM at the estate-level. In this specific case, increased harvesting costs were far from putting the economic advantages of CCF into question. This holds true also when considering the study by Price and Price (2008). The cost increase was quite low particularly under the considered single tree selection system, which Price and Price (2008) named "creaming", and would certainly not compensate for the beneficial effects of harvesting single trees according to their individual financial maturity.

However, we mentioned Andreassen and Øyen (2002) as well as Wikström (2000) as examples who found generally inferior results for CCF when compared to RFM. This result was to be expected because of the constrained management options in both studies, which considered initial stands very near or beyond financial maturity. These studies therefore do not prove a general inferiority of CCF. They make, however, clear that it is inappropriate to start transforming an age-class stand into CCF when this stand has achieved already an older age, close to or beyond financial maturity. If financial maturity is already exceeded, it seems disadvantageous to transform the stand into CCF using a series of light high thinnings, at least from an economic point of view. In such situation it may be better to apply RFM for a while and later on transform the newly established even-aged stand into a CCF system. In summary, it is not advisable to start the transformation process from even-aged to uneven-aged stand structure in mature stands, as was also shown by Tarp et al. (2000).

Cost savings by reduced infrastructure or forest personnel may change the overall very positive economic picture of CCF. Under CCF it is more difficult to standardize operations. Spatially explicit decisions are required which may increase the need for experienced staff. These are considerations relevant for rather large forest estates and too few studies are available so far to derive any conclusions. In summary, we may say that at least for small and medium-sized forest estates one may expect economic advantages from applying CCF.

3.4.2 Limitations in Modeling Approaches

Also the modeling approaches applied so far do often still show substantial gaps. An important precondition to model CCF is to have reliable predictions on growth and timber yield. In this line, the requirement for appropriate and sufficient regeneration and ingrowth to the smallest diameter class is of utmost importance. This is evident, for example, in Kant's (1999) study. His conclusion that CCF becomes unattractive when the interest rate would increase (this was found in contrast to many other studies) emerged from a special model assumption. This assumption shows a general limitation in modeling growth and timber yield of CCF (Roessiger et al. published online). Kant (1999) "… presumes an infinite, continuous, cost-free recruitment of young trees into the smallest diameter class, even if only very small trees remain after harvesting. It is unclear, however, whether these young trees are at all able to produce regeneration. The legitimacy of the "recruitment in any case" assumption is criticized by the author himself, as well as authors of other studies: Jübner (2006) points out that within the study by Buongiorno et al. (1995) recruitment of trees into the smallest diameter class is modeled, even though no seed trees exist. Sound modeling of recruitment is, however, essential in order to make a fair evaluation of uneven-aged management. Also in Wikström's (2000) study the modeling of ingrowth for CCF appeared to be problematic. Pukkala et al. (2009), for example, published a study using observations from existing plots to feed their recruitment model for uneven-aged stands. These authors were able to point out that the existence of understory trees, their shade tolerance, and their ability to recover their growth after a long time of suppression is crucial for uneven-aged, near-natural management (Roessiger et al. published online).

The mentioned limitation has important economic implications: If we assume that recruitment is always available even a single age-class forest would generate substantial net revenues continuously, for example every 5 years for an infinite period of time. In doing so we would transfer the crucial advantages of CCF delivering continuous and regular net revenues to a single age-class stand – a projection that appears unrealistic.

One of the most sophisticated and convincing studies is the investigation by Tahvonen et al. (2010) for CCF in Finnish Norway spruce. The authors used a size-structured transition matrix model to predict stand growth. The optimization problem, i.e. the maximization of NPV over an unlimited (or approximately unlimited) time horizon, is solved by nonlinear programming. The usual problem of multiple local optima, from which nonlinear programming often suffers, is circumvented by Knitro optimization software that applies gradient-based methods with interior point solutions (Tahvonen et al. 2010). I mentioned already that this study finds almost generally superior economic results for CCF by showing that practically without limitations the economically optimal forest management would sooner or later converge to CCF, if appropriate regeneration and recruitment is possible. However, even this study still has limitations, which the authors mention: Future studies must include logging damage, a better ingrowth model, non-timber values and variable cutting cycles. Above all, it is evident that the applied simulation and analysis is highly complex. In the practice of CCF, however, it would be very crucial to harvest the right trees (those showing low profitability) and to leave the more profitable ones to continue growing. Of course, profitability is a function over time and has to be evaluated always new. It is thus evident that it must be decided for every tree individually and for every single harvest operation what the best solution would be. Studies like that by Tahvonen et al. (2010) have thus to be transferred into tree marking guides to help implementing CCF efficiently. Simplified silvicultural rules will thus be necessary to apply CCF successfully in practice.

4 Reasons for Skepticism Against CCF

Approximately 25% of the world's timber production comes from fast grown pure plantations managed under the clearfelling system (Siry and Cubbage 2003). Around 50% of the world's timber production is firewood often coming from natural forests. "Other naturally regenerated forests" account for the remainder. Despite the rather small share of plantations, it does not seem likely that CCF will soon become a very popular silvicultural system. Still the skepticism against CCF, which is an efficient form of "forest gardening" (*jardinage*) requiring greater ecological know how than the much simpler and more straight-forward RFM systems, is great. Studies such as the one conducted by Cubbage et al. (2007), who compute internal rate of return (IRR) beyond 20%, support the general impression that an even-aged management with exotic and genetically homogenous trees in short rotations, with high chemical and mechanical input, will always outperform CCF. However, long-term ecological sustainability is one aspect that still needs to be evaluated by short-rotation industrial forestry.

Leaving the concerns about sustainability aside, we must admit that RFM may be more cost-efficient if all trees achieve the financial maturity approximately at the same time (if they are, for example, genetically modified), if risks are not important in forest management, and/or, if greater cost reduction regarding necessary infrastructure, personnel or harvesting may be achieved. When considering the situation in South America, where an industrialized short-rotation RFM is possible and where most of the calculations used in Cubbage et al. (2007) are based upon, we should, however, keep in mind the extremely high local risks (Benitez et al. 2007). For the South American types of high IRR forest investments, very high interest rates must already be applied to account for such country risks. This situation cannot be transferred to European countries and neither to the northern part of the American continent. The common skepticism against CCF seems thus hardly justified, for example, when considering forest management in the boreal and temperate zones of the northern hemisphere: If individual tree productivity and profitability is variable and if timber production comprises more than only one decade, as is typical for many places in the world, CCF offers the opportunity for a superior economic performance.

5 Concluding Remarks

There exists much evidence for a possibly superior economic performance of CCF over RFM, at least in the northern hemisphere. Worldwide, however, skepticism concerning CCF persists, which has possibly to do with concerns about the economies of scale. If one wants to promote CCF, studies on the economies of scale of silvicultural systems would thus be crucial. Moreover, the analyzed studies on the economics of CCF (at least the most convincing ones) are often limited to pure forest stands. Given the CCF tradition initiated by Gayer (1886), multi-species stands are a vital silvicultural option. What is lacking in the most recent considerations of the economics of CCF is the inclusion of the effects of mixing tree species, as studied for example by Knoke et al. (2005), Hildebrandt et al. (2010) and Beinhofer (2010a, b). These must be combined with the now well-known effects of uneven-aged silviculture and CCF. This implies an intensified consideration of risks and uncertainties (Staupendahl 2011) in conjunction with the effects of diversification inherent in CCF. Understanding the effects of uneven-aged silviculture in combination with mixed species forests more comprehensively, in the sense of our initial hypothesis:

> A deeper theoretical understanding of the economics of continuous cover forestry will rather reveal arguments in favor than against this silvicultural system

would be of great practical relevance. Improving the understanding of the functioning of CCF and uneven-aged silviculture would also constitute a highly interesting research area. CCF research has the potential to provide truly sustainable alternatives to short-rotation plantation forestry, which often requires high input of fertilizer and other chemicals, and intensive soil preparation which leads to soil erosion and degradation and provokes general environmental concern through depleting biodiversity.

Acknowledgements I am grateful to Laura Carlson for the language editing of the manuscript and to the German Research Foundation (DFG) for generously funding this research (KN 586/7-1).

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Chapter 6
Optimizing Continuous Cover Forest Management

Kari Hyytiäinen and Robert G. Haight

1 Introduction

The practice of silviculture involves the art and science of controlling the establishment, growth, composition, health, and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis (Helms 1998). Silvicultural practices are often divided into two broadly defined management systems, which in North America are usually referred to as "even-aged" and "uneven-aged" management. In the context of this book, "uneven-aged" management is synonymous with continuous cover forestry (CCF). The choice of appropriate management system is guided by case-specific considerations, including ecological requirements of tree species currently present and desired; effects of timber harvest on forest flora and fauna; risk of damage from wildfire, insects, or pathogens; and financial and other landowner objectives for the managed forest. Skillful use of silvicultural practices can achieve the landowner's objectives with greater assurance of success than will reliance on natural processes alone.

Financial objectives are important for both public and private forest land managers. The goal of this chapter is to lay out the theoretical foundations for evaluating the relative economic efficiency of alternative forest management systems and to present an overview of approaches and applications used for studying the economic questions of uneven-aged management. Our review focuses on stand-level economic studies. A forest stand serves as the smallest operational unit of forestry, and is thus the appropriate level for investigating the efficiency of alternative chains of stand management. Like the majority of studies in this field, we consider and formulate

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_6,

the forest management problem from the point of view of non-industrial private landowners. Such analysis is relevant for societies where land ownership rights are well established and private forests represent an important share of total forest area.

Profitability is one important factor landowners consider when they make decisions regarding their forest. Basically, the same economic principles apply for continuous cover forestry as for any other forest management system: the forest should provide, in the long run, more benefits than costs, including capital costs. The chains of management activities differ significantly between even-aged and uneven-aged management, and different formulations are needed for evaluating economic performance.

The remaining parts of this chapter are organized as follows. First, we present a method for computing net present value, which serves as a measure of economic efficiency. We provide a general formulation and then, as separate special cases, equations for even-aged management and uneven-aged management. Second, we review the literature in the discipline. Third, we present a detailed study that evaluates the relative efficiency of even-aged and uneven-aged management under the risk of wildfires. The final section presents our conclusions and suggestions for further research.

2 Landowner's Management Problem

A landowner for whom profit is the only objective manages his or her forestland such that the economic surplus from management is the highest possible. This is achieved by planning stand management such that the net present value of the future flow of forest products and services, net of costs, is maximized. For a given stand of trees and chain of silvicultural operations, the net present value, *NPV*, is computed simply by discounting all future revenues and costs, occurring in year t after the present, to the present and taking the sum over all years:

$$NPV = \sum_{t=0}^{\infty} e^{-rt} \left[\sum_{i=1}^{m} p_{it} q_{it} - \sum_{j=1}^{n} C_{jt} \right] \tag{6.1}$$

Annual gross revenues are computed by multiplying the amounts of m forest products and services q_{it} by their prices p_{it}. Products and services may be timber, hunting licenses, or amenity services that are purchased from the landowner. Costs are determined for all n management activities, including harvests and silvicultural activities. Silvicultural activities can be considered as investments that are done to improve production capacity of the stand. The rate of interest is denoted by r. It represents the expected return from alternative uses of capital. Discounting is a financial process making commensurable the costs and revenues occurring in different periods of time. The amounts of goods and services obtained from the stand in year t depend on stand structure Z_t and the level of n harvesting and management activities, u_{jt}, $j = 1, \ldots, n$.

$$q_{it} = f(Z_t, u_{1t} \ldots u_{nt}),\ i = 1, \ldots, m \tag{6.2}$$

For timber harvesting, harvesting intensity, u_{jt}, can be described as a proportion of trees removed from the stand. Prices of products and services, p_{it}, and costs of activities, C_{jt}, are also typically functions of stand structure and activities carried out in that year:

$$p_{it} = f(Z_t, u_{1t} \ldots u_{nt}),\ i = 1, \ldots, m \tag{6.3}$$

$$C_{jt} = f(Z_t, u_{1t} \ldots u_{nt}),\ j = 1, \ldots, n \tag{6.4}$$

In order to predict the future cash flow, we need a model that describes the growth and development of stand characteristics over time:

$$Z_t = f\ (Z_{t-1}) \tag{6.5}$$

The growth model estimates the birth, growth, and mortality of trees. Stand structure, Z, is a scalar (or vector) that describes stand structure by only one (or few) aggregate variables, such as stand basal area or timber volume. Alternatively, Z may be a matrix of more detailed stand description that classifies trees by tree size distribution or other characteristics (see Box 6.1 for descriptions of growth and yield models used in economic analysis).

A profit-oriented forest owner plans the future harvests and investments such that the net present value of the stand is maximized, i.e. he maximizes (6.1) with respect to (6.2)–(6.5). However, this problem formulation is not technically solvable, because planning of activities should be extended to infinity. A more tractable framework for calculating the value of forest land under even-aged management was established in the mid-nineteenth century in Germany (Faustmann 1849). A special feature of even-aged management is that all trees in a forest stand are about the same age and are all harvested in a final clearcutting. After clearcutting a new generation of trees is established by planting, sowing or from naturally regenerated seedlings. Even-aged management chains typically include thinnings and silvicultural activities such as tending of seedling stands. The net present value of bare forestland (Soil Expectation Value, SEV) before the start of stand establishment activities is given by

$$SEV = \frac{\sum_{t=0}^{T} e^{-rt} \left(\sum_{i=1}^{m} p_{it} q_{it} - \sum_{j=1}^{n} C_{jt} \right)}{1 - e^{-rT}} \tag{6.6}$$

Box 6.1: Stand Growth Models

Stand growth models used in economic analysis of forest stand management can be classified based on the level of aggregation. Growth prediction can be made for an entire stand, groups of trees (divided in classes based on tree diameter, age etc.), individual trees or parts of a tree (leaves, branches, bole, roots).

The simplest models of forest stand management describe the development of stand volume or value as a function of time. Such models are classified as density-free whole stand models (Davis et al. 2001) or univariate models (Getz and Haight 1989), and they have been used to study optimal rotation periods under even-aged management. However, to study uneven-aged management including thinnings and selection cuttings, more detailed whole stand models have been developed to include stand density as an independent variable. Such models are called variable-density whole stand models (Davis et al. 2001). Density is typically described in terms of stand basal area, i.e. the total cross-sectional area of trees measured at breast height per area unit (see e.g. Chang 1981). Whole stand models are useful tools especially in theoretical work (Garcia 1994).

Transition matrix or stage-structure models describe the stand state with a tree size distribution. Trees are classified in discrete size classes, typically characterized by tree diameter measured at breast height. Each class is represented by average tree volume, tree height, and number of trees. Growth is described as the transition from one class to another at discrete time intervals. Recruitment and survival functions define in-growth and mortality. Transition matrix models are popular and useful especially in the economic analysis of uneven-aged management. The model structure is reasonably simple, but detailed enough to account for the effects of in-growth and selection harvests.

Individual-tree (or single-tree) models describe a forest stand using a list of tree records. Each tree is characterized by a number of state variables reflecting its current dimensions (diameter, height, crown ratio etc.). The tree vectors evolve over time due to in-growth, growth, mortality and harvesting. With distance-independent individual-tree models, growth and mortality are specified as functions of stand density variables. Each tree in the list is assigned an additional state variable representing the number of its kind in the stand. With distance-dependent (or spatial) individual-tree models, in contrast, growth depends explicitly on a tree's location, height, and crown relative to its neighbors.

The most detailed stand growth models used in the economic analysis are process-based models that describe the biomass production and accumulation in different compartments of the tree. Process-based models describe the causal relationships between the resources (light, water, nutrients) and tree growth. Descriptions of the fundamental ecophysiological processes and morphology make it possible to study, in a detailed manner, the effects of varying harvests on the growth and timber quality of the remaining trees.

Here t refers to stand age. It is enough to optimize the stand management until the first clearcutting only ($t = T$). The later rotation periods are optimally similar as the first one provided that the stand's production capacity is not altered between the rotations. The sum of discounted net revenues from the first rotation is divided by the term $(1 - e^{-rT})$ in order to obtain the net present value of an infinite chain of similar rotation periods.

Bare land is a special case because forest land is bare only over a short period of time between clearcutting and establishment of new tree generation. The net present value of forested stands is computed by discounting the net revenues from the remaining part of the ongoing rotation and the bare land value from the end of rotation:

$$NPV = \sum_{t=t_0}^{T} e^{-r(t-t_0)} \left(\sum_{i=1}^{m} p_{it} q_{it} - \sum_{j=1}^{n} C_{jt} \right) + e^{-r(T-t_0)} SEV \tag{6.7}$$

The initial stand age is denoted by t_0. The value of an even-aged stand can be computed by maximizing (6.6) or (6.7) with respect to (6.2)–(6.5) depending whether the stand is initially bare or forested.

Under uneven-aged management, a forest stand will never be clearcut. Instead, selection harvests aim simultaneously at removing large merchantable trees, retaining vigorous trees of different ages, sizes and species and creating favorable conditions for natural regeneration. One common approach to determining the value of an uneven-aged stand is to assume the stand reaches a steady state within a given time T' (Haight and Getz 1987; Wikström 2000). At steady state, birth and growth of existing trees are in balance with harvests and the time between consecutive harvests, c, is fixed. The stand reaches the same state after every harvest cycle, and the net harvesting revenues from later harvests remain at the same balanced level. The management problem then is to determine the sequence of selection harvests that converts the current stand to an optimal steady-state harvesting cycle. The problem can be formulated as:

$$NPV = \sum_{t=1}^{T'} e^{-rt} \left[\sum_{i=1}^{m} p_{it} q_{it} - \sum_{j=1}^{n} C_{jt} \right] + \frac{\sum_{t=T'}^{T'+c} e^{-r(t-T')} \left[\sum_{i=1}^{m} p_{it} q_{it} - \sum_{j=1}^{n} C_{jt} \right]}{r} e^{-rT'} \tag{6.8}$$

$$Z_t = Z_{t-1}, t = T, T + c, T + 2c, \ldots \tag{6.9}$$

Thus, the value of uneven-aged stand can be computed by maximizing (6.8) with respect to (6.2)–(6.5), and (6.9). The first term in (6.8) denotes the net returns from the conversion or transformation period and the second term net returns after the steady state has been achieved.

The literature has many variations on how to solve (6.8) and which parts of the equation are exogenous and which parts are optimized. The traditional studies solving the steady state harvesting policies (see Sect. 3.1) focus solely on the latter part of (6.8). There are also studies that take the steady state as given and optimize harvesting over the transformation period (Torres Rojo and Sánchez Orois 2005; Yousefpour and Hanewinkel 2009). The steady-state structure of the stand can be optimized simultaneously with the transformation cuttings (see Sect. 3.2 for review of dynamic optimization studies). On the other hand, the steady state condition (the latter part of (6.8)) can be neglected if the planning horizon is long enough such that the system reaches some equilibrium or cycle endogenously (Haight et al. 1985; Tahvonen 2009). Extending the length of the transformation period and removing the restriction of reaching a final steady state may be justified on economic grounds (Haight and Getz 1987; Bare and Opalach 1988) or biological grounds in light of the criticism questioning the stability of reverse J-shaped diameter distributions (Linder 1998, Chap. 2).

The economic efficiency of different management systems can be investigated by setting the same initial state, applying coherently selected cost and price parameter values for alternative stand management systems, and maximizing the net present value for the two sets of equations ((6.2)–(6.6) or (6.2)–(6.5) and (6.7) for even-aged management and (6.2)–(6.5), (6.8), (6.9) for continuous-cover forestry). The first part of the Eq. 6.8 may also serve as a general objective function for both even-aged and uneven-aged management systems provided that removal of all trees (clearcutting) is allowed as an option, both natural and artificial regeneration are possible and the planning horizon is long enough to cover several life cycles of trees. The optimized variables are often described in terms of harvesting rates or diameter distributions of the remaining trees and the numbers of trees.

3 Stand-Level Studies on the Economics of Uneven-Aged Management

Forest economists have thought about uneven-aged management since the 1950s (Duerr and Bond 1952). Economic formulations of uneven-aged management include optimization of steady-state harvesting and dynamic harvesting without steady-state yield requirements. The problem formulations evolved in tandem with models for stand growth and yield (see Peng 2000 for review) and eventually were used to evaluate and compare the efficiency of even-aged and uneven-aged management systems. This section reviews the development of these formulations. See also Table 6.1 for an overview of literature in the field. Studies are listed in

Table 6.1 References of stand-level studies on the economics of uneven-aged management

Study	Approach	Model	Species	Country and region
Duerr and Bond (1952)	1	1	1	USA, South
Adams and Ek (1974)	1,2	2	4	USA, Wisconsin
Adams (1976)	1	2	4	USA, Wisconsin
Buongiorno and Mitchie (1980)	1,4	2	4	USA, Wisconsin
Chang (1981)	1	1	1	USA
Hasse and Ek (1981)	4	4	4	USA, North
Haight et al. (1985)	3	2	4	USA, Wisconsin
Haight (1985)	1,3	2	4	USA, Wisconsin
Mitchie (1985)	3	2	4	USA, North-central
Haight (1987)	3	2	1	USA, Arizona
Haight and Getz (1987)	3	2	2	USA, California
Bare and Opalach (1987)	1	3	2	USA, Idaho
Kaya and Buongiorno (1987)	5	1	4	USA, Wisconsin
Bare and Opalach (1988)	1	2	4	USA, Wisconsin
Hotvedt et al. (1989)	1	1	2	USA, Arkansas and Louisiana
Buongiorno and Lu (1990)	1	2	4	USA, Wisconsin
Haight and Monserud (1990a)	3	3	2	USA, Idaho
Haight and Monserud (1990b)	3	3	2	USA, Idaho
Haight (1990)	5	2	1	USA, California
Gove and Fairweather (1992)	1	2	4	USA, North-central
Haight et al. (1992)	3	3	2	USA, Idaho
Anderson and Bare (1994)	3	2	4	USA, Wisconsin
Gove et al. (1994)	1	2	4	USA, Wisconsin
Buongiorno et al. (1994)	1	2	4	USA, Wisconsin
Buongiorno et al. (1995)	1	2	5	France, Jura mountains
Volin and Buongiorno (1996)	4	2	5	Italian Dolomites
Boscolo et al. (1997)	4	2	6	Malaysia
Lin and Buongiorno (1998)	3	2	4	USA, Wisconsin
Schulte and Buongiorno (1998)	4	2	2	Southeast USA
Kant (1999)	3	2	1	Canada, Ontario
Buongiorno et al. (2000)	4	2	4	USA, North
Wikström (2000)	3	3	1	Sweden
Tarp et al. (2000)	4	1	3	Denmark
Mendoza et al. (2000)	1	2	6	Indonesia, Kalimantan
Knoke and Plusczyk (2001)	4	4	5	Germany, Bavaria
Buongiorno (2001)	3	2	4	USA, Wisconsin
Hanewinkel (2001)	4	4	5	Southwest Germany
Andreassen and Øyen (2002)	4	1, 5	5	Norway
Nord-Larsen et al. (2003)	4	5	4	Northern Germany
Wagner et al. (2003)	1,4	3	4	USA, Northeast
Ralston et al. (2004)	4	2	1	USA, Pacific Northwest
Sánchez Orois et al. (2004)	1,4	2	1	Northwest Spain
Trasobares and Pukkala (2004)	1	3	2	Northeast Spain
Conrad et al. (2005)	3	1	6	Not specified

(continued)

Table 6.1 (continued)

Study	Approach	Model	Species	Country and region
Rollin et al. (2005)	5	1	5	France, Jura mountains
Torres Rojo and Sánchez Orois (2005)	2	2	1	Spain, Galicia
Tarp et al. (2005)	4	2	3	Denmark
Hao et al. (2005)	3	2	4	China, Northeast
Zhou and Buongiorno (2006)	5	2	5	USA South
Jacobsen and Helles (2006)	2	2	3	Denmark
Liang et al. (2006)	5	2	2	USA, Pacific Northwest
Lohmander and Limaei (2008)	5	1	4	Iran, Caspian forests
Yang and Kant (2008)	1	2	4	Canada, Ontario
Tahvonen (2009)	3	2	1	Finland
Yousefpour and Hanewinkel (2009)	2	4	5	Germany, Black forest
Hyytiäinen and Haight (2010)	3	3	2	USA, Idaho
Chang and Gadow (2010)	1	1	2	USA, South Central
Pukkala et al. (2010)	1	3	2	Finland
Tahvonen et al. (2010)	3	2	1	Finland
Xabadia and Goetz (2010)	3	2	1	Spain

Approach
1. Optimizing steady state harvests
2. Optimizing conversion harvests
3. Simultaneous optimization of conversion period and steady state
4. Simulation based comparison of conversion strategies
5. Adaptive harvesting under stochastic conditions

Model or data used for projections
1. Whole stand model
2. Transition matrix model
3. Distance-independent individual-tree model
4. Spatial model
5. Experimental data

Species
1. Conifer
2. Conifer mixture
3. Hardwood
4. Hardwood mixture
5. Mixture of hardwoods and conifer species
6. Tropical forest

chronological order and classified according to the primary approach and the type of growth and yield model applied. Also the country, area and tree species employed in each case study are reported to indicate the geographical coverage of this line of research.

3.1 Optimization of Steady-State Harvesting

Uneven-aged management is usually defined as the periodic removal of a portion of the trees from a stand while maintaining a balanced uneven-aged structure with three or more age classes of trees (e.g., Smith 1962, p. 467). The ideal stand structure is a downward-sloping diameter distribution containing sufficient trees in each diameter class to produce an unvarying number of trees that are harvested periodically. Harvests include selection cutting (i.e., removing all trees greater than some maximum diameter) and thinning (i.e., removing a portion of the trees in smaller diameter classes) (Smith 1962, p. 482). We use the term selection harvests to include both thinning and selection cutting in uneven-aged management regimes.

The first economic formulations of uneven-aged management problems focused on calculating the optimal level of growing stock of a balanced uneven-aged stand, where optimal stocking was the timber volume that maximized net benefit through periodic harvest of the volume growth. Duerr and Bond (1952) used marginal analysis to show that an optimal growing stock is one in which the marginal value growth percent is equal to the discount rate. Chang (1981) showed that optimal stocking can be calculated by maximizing the difference between the present value of equilibrium harvests and the liquidation value of the residual growing stock, which represents the opportunity cost of holding the residual growing stock. This formulation appealed to forest economists because if the growing stock is viewed as a capital investment, the problem is equivalent to maximizing land expectation value (LEV) as defined by the Faustmann formula (see Chang 1981; Chang and Gadow 2010).

While the analyses above calculated the optimal aggregate growing stock of a balanced uneven-aged stand, they did not address the problem of determining the optimal steady-state stand structure. Adams and Ek (1974) were the first to formulate and solve an optimization problem for a steady-state diameter distribution of the residual growing stock and harvest. They demonstrated a two-stage technique for determining the optimal stand structure. In the first stage, they used a gradient projection method to determine the diameter distribution that maximized stand value growth for a given stand basal area and cutting cycle. In the second stage, having solved this nonlinear program for several alternative basal area levels, they chose the stand structure that satisfied a marginal value growth percent criterion (Duerr and Bond 1952). Adams (1976) subsequently distinguished between the use of value and basal area measures of growing stock as the appropriate constraints for determining investment-efficient diameter distributions. These analyses used a stage-structure model for the growth of uneven-aged northern hardwood stands in Wisconsin (Ek 1974). The model included density-dependent nonlinear equations for the movement of trees between diameter classes, mortality, and regeneration. Extensions of optimization problems with steady-state diameter distributions included determination of the optimal cutting cycle and species composition (Buongiorno and Michie 1980; Bare and Opalach 1987).

Adams and Ek (1974) also addressed transition harvesting for a given initial stand structure. They formulated and solved the transition strategy problem of choosing harvesting levels of diameter classes that maximized present net value with the constraint of achieving a given diameter distribution after a specified number of periods. They used a gradient projection method, but the number of decision variables and constraints exceeded algorithm capacity for problems with more than three transition harvests.

3.2 Optimization of the Sequence of Selection Harvests

In a departure from steady-state analysis, Haight et al. (1985) addressed the problem of determining the optimal sequence of diameter distributions and selection harvests for an existing stand without the constraints of steady-state harvest or specified equilibrium endpoint. The objective function was formulated to seek diameter-class harvest levels that maximize the present value of net returns, assuming that all remaining trees are harvested at the end of the planning horizon. No restrictions are placed on the form of the terminal diameter distribution. They coupled a nonlinear programming technique called the method of steepest descent with the stage-structured model of northern hardwood stands developed by Ek (1974) and modified by Adams and Ek (1974) to determine stand-specific management regimes for three different stumpage value functions over a 150-year horizon. Their results suggested that investment-efficient residual diameter distributions determined by Adams (1976) were not optimal in the context of this dynamic problem. In a subsequent mathematical analysis, Haight (1985) showed that steady-state management regimes obtained in the context of a dynamic harvesting problem were not the same as steady state regimes developed with static optimization, which maximizes the present value of equilibrium harvests minus the value of residual growing stock (Adams 1976; Buongiorno and Michie 1980; Bare and Opalach 1987).

When the objectives of timber harvesting include the maximization of present value and the achievement of a steady-state harvest policy, the management problem can be formulated as a dynamic harvesting model with fixed or equilibrium endpoint constraints (Haight and Getz 1987). Fixed endpoint problems involve the determination of a target steady state and a transition regime that reaches the target after a finite transition period (e.g. Adams and Ek 1974). Equilibrium endpoint problems involve the determination of transition and steady-state harvest levels with equilibrium endpoint constraints that do not require the achievement of a specific target stand structure (see (6.8)). Michie (1985) formulated and solved an equilibrium endpoint problem using a fixed parameter matrix model for stand growth and linear programming methods. The resulting steady states depended on the initial stand structure and the conversion period length. Computational limitations prevented the author from solving problems with three or more transition harvests.

Haight and Getz (1987) focused on issues associated with fixed and equilibrium endpoint problems. For a given transition period length, the solution to the equilibrium endpoint problem has a higher present value than the solution to any fixed endpoint problem, since the equilibrium endpoint formulation places fewer constraints on the terminal steady state. The equilibrium endpoint policy depends on the initial stand structure and transition period length. As the transition period lengthens, the cost of the terminal steady-state constraint approaches zero.

Wikström (2000) applied a Tabu search solution algorithm for analyzing and solving uneven-aged and even-aged problem formulations with individual-tree models. He studied different problem formulations with and without steady-state constraints. Approximating the infinite time horizon problem with a finite time horizon turned out better both technically and with regard to the reliability of the results. Tahvonen (2009) and Tahvonen et al. (2010) focused on developing any-aged problem formulations that do not include any restrictions on the forest management system or final state. In these studies, the infinite horizon solutions are approximated by increasing the number of periods until further lengthening of the horizon does not change the solution trajectory toward some steady state or stationary cycle. In some cases, the planning horizon needs to be extended up to between 800 and 1,500 periods. Long run simulations were made technically feasible for Norway spruce (*Picea abies*) stands by the use of transition matrix stand growth models.

3.3 Comparing Economic Returns from Even-Aged and Uneven-Aged Management

The problem of evaluating and comparing the efficiency of even-aged and uneven-aged management systems was first addressed by comparing simulated yields from steady-state management regimes (i.e., a repeated sequence of even-aged stands managed with the clearcut system and a repeated sequence of single-tree selection harvests from a sustainable, uneven-aged diameter distribution). For example, using a simulator for Wisconsin northern hardwoods, Hasse and Ek (1981) compared mean annual increments measured in various physical units for even-aged stands with the corresponding yields from steady-state uneven-aged management regimes. Chang (1981) computed a steady-state uneven-aged management regime that maximized LEV and compared the LEV associated with uneven-aged management to LEVs computed for even-aged plantation management regimes.

These comparisons of the economic returns of alternative management systems did not consider the more general problem of managing an existing stand. For even-aged management this problem is split into conversion and plantation components: (1) determining the timing and intensity of silvicultural treatments for the current stand and determining the time when the stand is clearcut and replaced with a plantation, and (2) determining the timing and intensity of silvicultural treatments and clearcut age for the plantation (see (6.7)). For uneven-aged management the problem involves determining the sequence of selection harvests that converts the current stand to steady-state uneven-aged management (see (6.8) and (6.9)). Haight (1987) described models that allow the comparison of the present values of optimal management regimes that fit these definitions of even-aged and uneven-aged management. The optimization problems included a stage-structured model for growth and yield of ponderosa pine (*Pinus ponderosa*) stands in the western United States.

It is important to caution the reader that defining uneven-aged management as the conversion of an existing stand diameter distribution to an inverse-J shaped diameter distribution may not be justified on economic or biological grounds. Inverse-J shaped diameter distributions may not represent virgin forest structures, which often have bimodal or more complicated distributions, especially in multi-species stands (Chap. 2). Further, the assumption that there is a unique and ideal diameter distribution for an uneven-aged stand cannot always be substantiated by empirical evidence (Chap. 2).

To address uneven-aged management problems without steady-state constraints, Haight and Monserud (1990a) developed an any-aged management formulation and applied it to mixed-conifer stands in the Northern Rocky Mountains. The any-aged management problem is to determine the best time sequence of harvests and plantings without constraints on the stand age or size structure. With no constraints, optimal management may prescribe stand structures that vary from even-aged to irregular, and thus regimes are termed any-aged. Because the any-aged management problem includes even-aged and uneven-aged management problems as special cases, optimal any-aged regimes should be superior to regimes that fit these standard definitions of stand management. In a departure from previous models, Haight and Monserud (1990b) developed an optimization procedure for application to a single-tree stand simulator, which was becoming the standard model for growth and yield projection in the United States.

The existing results on the economic superiority of even-aged versus uneven-aged forestry are mixed. According to Haight and Monserud (1990b), optimal any-aged management regimes for conifer cultures in the Rocky Mountains tend to converge to a sequence of selection harvests that maintain a multispecies, uneven-aged stand structure. However, optimal transition regimes depend on initial stand structure, and may include sections of even-aged forestry. Clearcutting and planting are practiced for understocked stands with basal area lower than 5 m^2/ha. However, with natural regeneration and repeated diameter-limit cuts, even the monocultures finally evolve to mixed-species uneven-aged stands.

Wikström (2000) found even-aged management superior for Norway spruce stands in Sweden. Depending on the type of the steady state constraints and initial stand structure, the maximized net present values of uneven-aged management varied between 67% and 95 % from the maximized net present values of even-aged management. However, optimal solutions for uneven-aged management were constrained by either an upper bound for thinning intensity (30%) or a lower bound for residual volume (150 m^3/ha). The level of in-growth was also relatively low (10 trees/ha/year). Relaxing these constraints and making in-growth depend inversely on stand stocking would obviously have improved the relative profitability of uneven-aged management. Tahvonen et al. (2010) relaxed any constraints on harvesting and carried out similar optimizations for Norway spruce in a neighboring country, Finland. They found that uneven-aged management gives somewhat lower timber yields in the long-run than even-aged management, but becomes economically superior for most initial stand states when the costs of regeneration and harvest, interest rate, and the price differential between the saw timber and pulpwood, are accounted for. The properties of the growth, in-growth and harvesting models that drive the optimality of uneven-aged vs. even-aged management have been studied by Chang (1981), Tahvonen (2009), and Khazri and Lasserre (2011).

3.4 Simulation-Based Comparisons of Transformation Strategies

Converting single species even-aged conifer stands into more irregular stand structures involving multiple species has recently become a widespread management objective in Central Europe (Gadow et al. 2002; Pommerening and Murphy 2004). The potential environmental benefits associated with this objective are improved soil properties, water quality and habitat quality and reduced susceptibility to natural hazards. Interest in this objective has increased the demand for research investigating the economic and ecological consequences of transformation strategies. The approach used to investigate these consequences has been to select one or several initial stand states and project stand development associated with various conversion strategies for comparison with a conventional management strategy (typically even-aged management). The analysis is completed by comparing and analyzing the economic and ecological outcomes of the exogenously selected management regimes (Buongiorno 2001). Knoke et al. (2008) review the literature on admixing broadleaved to coniferous trees species.

Knoke and Plusczyk (2001) and Hanewinkel (2001) combined empirical data and growth projections produced with the spatial growth model Silva to investigate the economic effects of transforming stands dominated by Norway spruce in Germany. Transformation regimes included a chain of selection harvests and occasional enrichment plantings in forest gaps generated by snow breakage or thinnings. The transformation strategy was found to lead to smaller net incomes than even-aged management. However, the incomes occurred earlier and were more evenly distributed over time, which made the transformation strategy a financially viable option especially at higher rates of interest. Tarp et al. (2005) carried out a similar comparison of an even-aged, shelterwood natural regeneration regime of beech (*Fagus sylvatica*) to a conversion regime based on target diameter harvesting. They found very little difference in the economic profitability of those regimes. The regime using target diameter harvesting and leading to an uneven-aged stand structure was superior for interest rates of 2% and higher.

Alternative chains of stand management have also been evaluated according to non-timber benefits. For example, Schulte and Buongiorno (1998) projected the development of loblolly pine (*Pinus taeda* L.) using eight different management guidelines, and evaluated the outcomes in terms of economic performance, timber production divided by assortments (sawtimber and pulpwood), and tree-species and tree-size diversity. Some studies compare several transformation strategies and select the best one based on single or multiple criteria.

3.5 Adaptive Optimization of Cuttings Under Stochastic Conditions

Most studies on optimal forest stand management assume that forecasts of economic parameters and stand growth are known with certainty. However, several model components and parameters that are treated as fixed values in optimization models are in fact stochastic. Timber prices, in particular, fluctuate greatly over time. Studies that recognize inherent uncertainties in some of the key parameters employ stochastic optimization to design adaptive timber management strategies and account for unpredictable changes in market condition or stand state.

In adaptive optimization, the optimal decision at each point in time is made conditional on the stand and market state. A Markov decision process, or discrete time stochastic control, provides a framework for formulating such problems, and dynamic programming provides a means to solve them. Kaya and Buongiorno (1987) determined the optimal cutting intensity for different combinations of stand basal area and timber price. The two state variables were defined by three levels of timber price and five levels of stand basal area. Later, Markov decision models were developed to include different tree sizes and species groupings (Lin and Buongiorno 1998; Rollin et al. 2005) and natural hazards (Zhou and Buongiorno 2006). Dynamic programming proved to be a powerful tool to solve stand management problems in which the stand state can be described by a few discrete state variables. For more complex problem formulations involving four or more state variables, the computational burden of dynamic programming becomes overwhelming (Bellman and Dreyfus 1962).

There are other methods for developing adaptive cutting rules with fluctuating timber prices. Haight (1990) employed stochastic simulation to develop feedback thinning rules for uneven-aged stands of white fir (*Abies concolor*). Stumpage price was assumed to evolve over time according to a stationary random process. The decision variable, given as a function of timber price, was thinning intensity. Adapting the harvest removal to timber price increased the expected net revenues significantly compared with a harvest policy that did not account for random price changes. Jacobsen and Helles (2006) developed a vector autoregressive model for timber assortment prices and used stochastic dynamic programming to determine the optimal timing of selection harvests for seven different harvest guidelines for beech. They found that adapting the harvesting decisions to price fluctuations changes the ranking of alternative management strategies compared with the results of deterministic optimization.

Stochastic problems can also be formulated such that the inherent variation in the model parameters is accounted for but without adjusting the harvest decisions to the stand or market state at each point in time. Bootstrap simulation with response surface analysis is one such approach (Liang et al. 2006), and simulation-based optimization is another (Hyytiäinen and Haight 2010). The benefit of an anticipatory approach to uncertainty is that it can be easily applied to complex stand growth models involving several sources of uncertainty and state variables. The disadvantage is the limited potential to adjust the decisions with changes in the stochastic variables.

3.6 Joint Optimization of Timber and Amenity Benefits

Forests provide multiple benefits. Some of these benefits are tangible products, such as timber, the value of which is determined in competitive markets. However, forests provide a number of other products and services that may be of high value for the landowner and society, but which are more difficult to quantify and account for in the economic analysis. Watershed protection and water purification are examples of non-market values of forests. Forests are also subject to non-use values that derive from the pleasure people feel from the knowledge that the forest ecosystem functions well and provides a sustained flow of goods and services for the present and future generations. While a wide range of advanced techniques (e.g. cost-based methods and contingent valuation) have been developed to estimate the monetary value of non-market and non-use benefits of natural resources (Freeman 2003), monetary estimates of non-market and non-use values are subject to considerable measurement uncertainties (see Merlo 2005, p. 17–35 for application of total economic value for forests).

Amenity benefits have been inserted in optimization formulations of uneven-aged stand management as constraints in dynamic (Haight et al. 1992) and static models (Buongiorno et al. 1994, 1995; Lin and Buongiorno 1998). Amenity benefits have also been inserted in the objective function and given a weight reflecting its value to the landowner or society (Yousefpour and Hanewinkel 2009). An alternative approach is to insert the undesirable environmental effects of timber harvesting as penalties or costs (Khazri and Lasserre 2011). There are also studies that optimize stand management with respect to a diversity index (Gove et al. 1994) or with respect to both ecological and economic criteria for comparison of the outcomes (Liang et al. 2006).

Out of a wide array of non-timber benefits, habitat quality has been the most common service accounted for in economic problem formulations. The most popular measure of habitat quality has been the Shannon index, which accounts for diversity in tree sizes and tree species composition (e.g. Lin and Buongiorno 1998; Rollin et al. 2005). The value of carbon sequestration is accounted for in some studies (Boscolo et al. 1997; Yousefpour and Hanewinkel 2009; Pukkala et al. 2011). Berries and mushrooms are important non-timber forest products that have been included in some recent formulations of even-aged (Palahí et al. 2009; Miina et al. 2010) and uneven-aged (Pukkala et al. 2011) management.

4 A Case Study for Mixed Conifer Stands in the Rocky Mountains: Effect of Fire Risk on Efficiency of Even-Aged vs. Uneven-Aged Management

In this section we describe in detail a study that evaluates the economic efficiency of even-aged and uneven-aged management under the risk of wildfires (Hyytiäinen and Haight 2010). First we present the case study and introduce the distance-independent individual-tree model to simulate stand growth and fire effects. Then, we develop an optimization method that accounts for uncertainties both in tree growth and occurrence of wildfire. We describe the relative efficiency of the two management systems with and without fire risk. Finally, we add a diversity index to the model of Hyytiäinen and Haight (2010) and demonstrate the environmental consequences of continuous cover forestry as opposed to traditional even-aged management.

4.1 Case Study

The case study is a mixed conifer stand located in Idaho Panhandle National Forest in the Rocky Mountains. The stand is a mature two-storied stand that can be either converted to even-aged conifer plantation by immediate clearcutting and artificial regeneration or managed with selection harvests. The species present in the stand initially include Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), subalpine fir (*Abies lasiocarpa*), western hemlock (*Tsuga heterophylla*), western larch (*Larix occidentalis*), and western redcedar (*Thuja plicata*). The stand belongs to the western hemlock habitat type with an elevation 880 m above sea level. The initial basal area is 28.6 m^2/ha and the average top height is 30 m. The species considered for even-aged management is Western white pine (*Pinus monticola*).

Wildfires pose a threat of damage and economic loss to landowners especially in regions with hot and dry seasons. Wildfires may cause total destruction of the stand by killing all trees, or may cause only minor damage to some trees, or any degree of damage between these two extremes. This chapter demonstrates how anticipating the fire risk and its negative consequences affect the choice of management system.

4.2 Simulation Model

Simulation models are needed for predicting stand structure development as a consequence of growth of existing trees, regeneration, harvests, silvicultural activities, and natural and catastrophic mortality. In this study, stand growth was predicted by using the Forest Vegetation Simulator (FVS), and its model version validated for mixed conifer stands in northern Idaho (Dixon 1989, 2002). The fire effects on stand structure were simulated using the Fire and Fuels Extension of the FVS (Reinhardt and Crookston 2003).

FVS is a family of forest growth simulation models developed by the USDA Forest Service. The first versions of the simulator date back to the early 1970s. The original version of the simulator was called the Prognosis model (Stage 1973) and it was developed for northern Idaho and western Montana. Later on, data from other geographic areas became available and were used to develop new model variants. In the early 1980s, Prognosis was renamed the *Forest Vegetation Simulator* (FVS) and was adopted by the USDA Forest Service National Forest System as the national standard for forest growth and yield modeling. At present, FVS consists of more than 20 variants calibrated for different geographic areas, and soil and weather conditions in the United States and Canada. Prognosis and FVS have been applied actively in the economic analysis of forest management (Bare and Opalach 1987; Haight and Monserud 1990a, b; Haight et al. 1992).

Each variant includes thousands of state variables describing trees and fuels and predicts tree growth and mortality as functions of natural factors (such as competition, aging, wildfire) and human intervention (harvesting and silvicultural activities). The northern Idaho variant used in Hyytiäinen and Haight (2010) also contains a full establishment model (Dixon 2002). It adds in-growth periodically during the simulation, and pulses of regeneration following significant stand disturbances. It is also possible to add trees by planting. The northern Idaho variant of FVS can be applied for simulating both even-aged and uneven-aged management regimes.

FVS variants are distance-independent, individual-tree models. The structure of the forest stand is described using a list of tree records. Each tree is characterized by a number of state variables reflecting its current dimensions (e.g. diameter, height, crown ratio) and an additional variable representing the number of trees of its kind in an area unit (typically number of trees per acre). The variable values evolve over time due to growth, mortality and harvesting. Tree growth is described as function of stand density variables and shading by neighboring trees.

Figure 6.1 shows the general flow of operations within FVS. A projection of stand development begins by reading the tree and stand level inventory records and the description of management schedule. If measured data on past growth are available, the increment equations are calibrated to better reflect the forest growth capacity of the stand. The tree basal area growth equations are scaled such that the predictions match the actual increment core measurements on the trees in the stand (Stage 1973). After this, the first projection cycle begins. The growth cycle length can be adjusted. The default length is 10 years.

Each projection cycle starts with a check to see if any harvesting or silvicultural activities have been scheduled for the particular growth cycle. This is followed by computing periodic diameter increment, periodic height increment, and periodic mortality rates. These estimates are then adjusted for effects of insects and pathogens. Tree records resulting from regeneration within the cycle are created

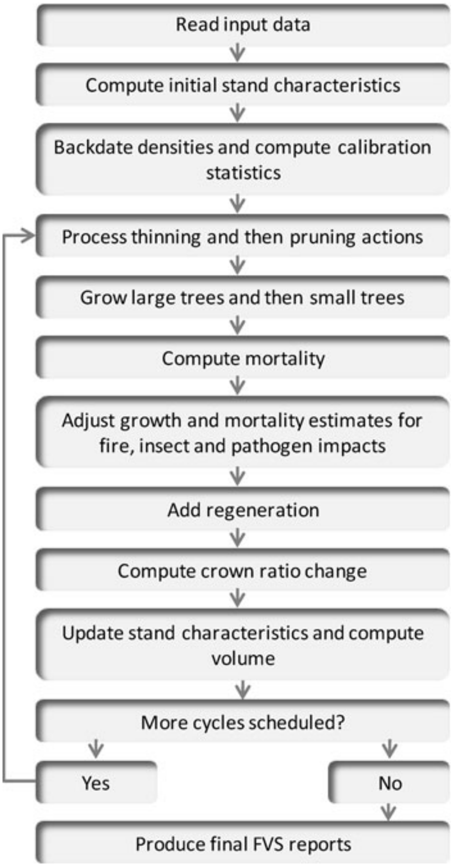


Fig. 6.1 Overview of the general FVS processing sequence (Adapted from Dixon 2002)

next, and changes in crown ratios are computed for each tree record in the projection. Finally, the tree attributes are updated, tree volumes are calculated, and tables that summarize projected stand conditions are compiled.

The stochastic features in tree growth are assigned to the distribution of errors associated with the prediction of tree diameter increment. The random component of change in tree diameter is treated by (1) adding a random deviate to the basal area increment of each tree record, and (2) augmenting (tripling) the number of tree records for consecutive periods for stands with small initial number of tree records. The random variables associated with each tree record are saved until the following cycle in order to account for serial correlation (Stage 1973). Simulations can be replicated with different random number seeds to obtain scenarios of stand development.

Several extensions have been developed for the FVS in order to predict the effects of wildfires and specific insects or pathogens on stand and tree development and mortality. Fire and Fuels Extension to the FVS (Reinhardt and Crookston 2003) integrates FVS with elements from existing modes of fire behavior and fire effects. It simulates the effects of stand development and management actions on fuel dynamics, fire behavior, and fire effects. The model predicts fire intensity and fire type, which can be surface fire, passive crown fire or active crown fire. Fire intensity and tree mortality are predicted as functions of the vertical distribution of fuels, tree characteristics (e.g. crown length, diameter, and species), stand characteristics (e.g. slope) and environmental variables at the moment of fire (fuel moisture, wind speed, temperature). The probability of fire arrival does not depend on stand state or prior management, and must be given separately to the model.

FVS and its extensions are publicly available tools for analyzing stand management. Links to technical documents, software, samples of data, and support and training services can be found on the USDA Forest Service web site: http://www.fs.fed.us/fmsc/fvs/.

4.3 Approach

The approach used in this study is simulation-based optimization. In this approach, the expected value of the objective function is estimated via stochastic simulation until suitable approximations of the optimal values of the decision variables are found. The first step is to estimate the expected value of the objective function for a given set of decision variable values. This sample average is computed using a set of scenarios where each scenario is composed of random samples from the distributions of the random variables. The second step is to search for a solution that provides a suitable approximation to the optimal values of the decision variables.

One advantage of applying simulation optimization to forestry questions is that the model of stand dynamics can be used without modification in the search of the best management regime. A disadvantage is its computational intensity: a large number of replications may be required to obtain a useful estimate of the expected value of the objective function for each management regime. Another disadvantage of this approach is that the optimal stand management regime is decided in advance and is independent of future fire occurrences.

Applying simulation based optimization with the FVS turned out to be computationally intensive. A typical simulation of harvests and wildfire requires 10–30 s on a laptop computer, and hundreds of simulations may be needed to stabilize the expected objective function value. Therefore, a number of simplifications were made in the problem formulation in order to reduce the computational burden. First, a small number of discrete decision variables were developed separately for even-aged and uneven-aged management. Second, the minimum step lengths for decision variable values were increased. Third, harvesting was made to take place on a 20-year interval rather than the 10-year prediction interval allowed by the simulator. Fourth, wildfire occurrences were discretized such that a fire may occur once during each 20-year period.

These simplifications reduced the set of possible harvesting regimes compared to earlier studies of any-aged stand management (e.g. Haight and Monserud 1990a) and made it possible to use FVS and its Fuels and Fire extension in the analysis. In this study, the focus is on the management decision in the first period: whether the stand is retained under uneven-aged management or converted to an even-aged plantation.

The net present values of stands were computed as follows. First it was assumed that the initial stand can be either retained under uneven-aged management by repeated diameter-limit harvests or converted to even-aged management with an immediate clearcut followed by artificial regeneration and an infinite series of rotations. The expected present value of the initial stand is computed separately for each management system. The optimization problem for each management system is to select the values of the decision variables (denoted by a vector x) to maximize the expected net present value of the stand, $E(NPV)$, under risk of wildfires and stochastic tree diameter growth:

$$\max_{x} \quad E\,(NPV) = \sum_{i=1}^{m} \sum_{j=1}^{n} pr_{ij}\,(\lambda)\; NPV_{ij}\,(x, \chi, \lambda) \tag{6.10}$$

The probability distribution for NPV is estimated by repeating the simulations for m fire scenarios and n sequences of tree growth predictions (see Fig. 6.2 for illustration on the structure of simulation optimisation framework). For each set of decision variable values x, $E(NPV)$ is estimated by first computing NPV_{ij} for the ijth scenario of fire and tree growth, then multiplying NPV_{ij} by the probability of occurrence of scenario ij (pr_{ij}), and finally summing the products over the mn simulations. The probability of fire occurrence in a period is denoted by λ ($0 \leq \lambda < 1$). The seed random number used in diameter growth predictions, χ, is specific for each simulation.

It is assumed that fire enters the stand by spreading from outside. Thus, the occurrence of wildfire does not depend on stand characteristics or prior management. The period length w is 20 years and the number of periods q is 6. Let ψ_k be a dummy variable with value of 1 or 0 for whether or not a fire occurs during period k. The probability that a wildfire occurs in period k is $P(\psi_k = 1) = \lambda$ and the probability that the wildfire does not occur is $P(\psi_k = 0) = 1 - \lambda$. The simulations are repeated for all $m = 2^q = 64$ possible realizations of fire occurrences. The probability of each fire scenario i and sequence of diameter growth predictions, j, is:

$$pr_{ij} = \frac{1}{n} \prod_{k=1}^{q} P(\psi_k^i), \quad i = 1, \ldots, m, \; j = 1, \ldots, n \tag{6.11}$$

With uneven-aged management, the decision variable x is the diameter limit of selection cutting (given in cm). A diameter-limit cut removes all trees above a

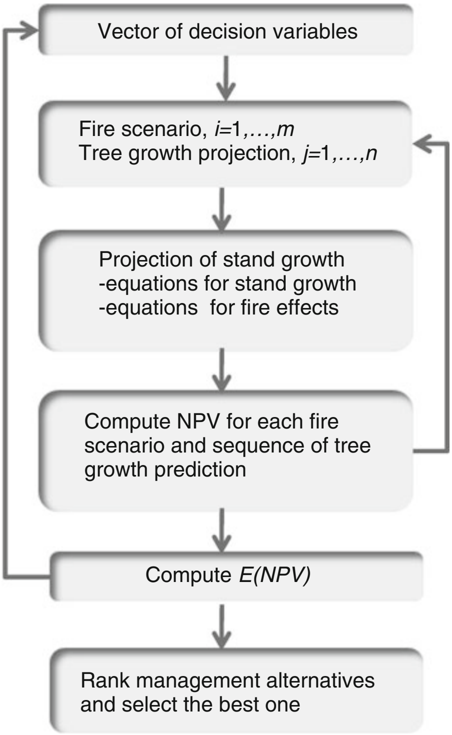


Fig. 6.2 Steps in simulation based optimization framework

minimum diameter. In addition, we assume that 15% of smaller trees not exceeding the diameter limit are removed by thinning. These include inferior quality stems and those stems that are damaged during the logging operation. With even-aged management, x is a vector that consists of three decision variables: (1) planting density (trees/ha), (2) residual basal area in thinnings, (m^2/ha), and (3) rotation period (years). After clearcutting, the stand is replanted immediately. The new tree regeneration is thinned to the same residual basal area after every 20 years and clearcut at rotation age. Thinnings are assumed to remove equal proportions of trees from all tree records until the target residual basal area is reached. For additional details and specific cost and price parameters see Hyytiäinen and Haight (2010).

Finally, a Shannon index (see e.g. Rollin et al. 2005) is linked to the model to demonstrate the development of habitat quality for alternative chains of stand management. The Shannon index, S, reflects habitat quality and describes a stand's richness with respect to tree sizes and tree species (assuming that these two characteristic of trees are independent of each other).

$$S = -w_{sp} \sum_{u=1}^{y} \frac{B_{ku}}{B_k} \ln\left(\frac{B_{ku}}{B_k}\right) - w_{size} \sum_{v=1}^{z} \frac{B_{kv}}{B_k} \ln\left(\frac{B_{kv}}{B_k}\right) \tag{6.12}$$

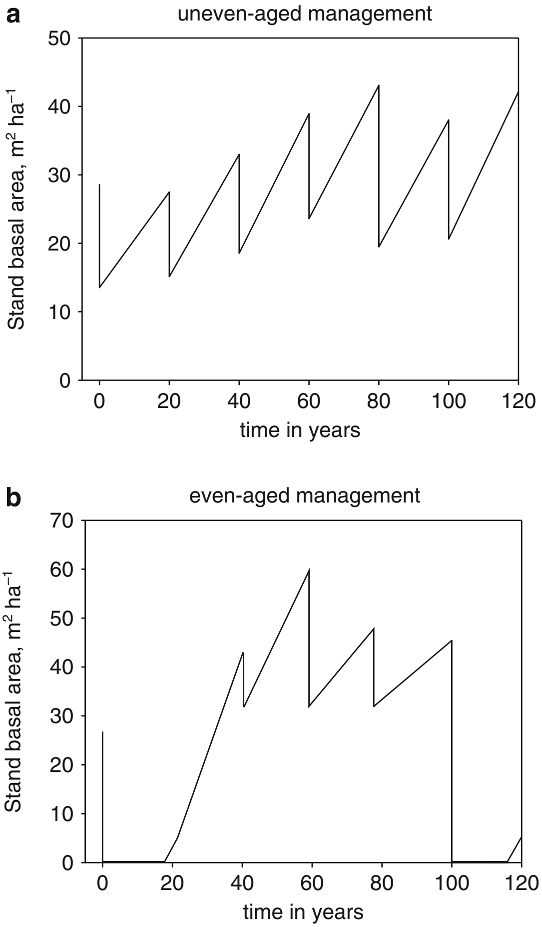


Fig. 6.3 Projections of stand development for optimal management under (**a**) uneven-aged management and (**b**) even-aged management without the risk of wildfires

The weights of species and size diversity are denoted by w_{sp} and w_{size}, respectively. B_k is the total basal area at the period k, B_{ku} is the total basal area of trees of species u, and B_{kv} is the basal area of trees of diameter class v.

4.4 Economic Efficiency of Even- vs. Uneven-Aged Management

Optimal chains of stand management and the efficiency of alternative stand management systems are first evaluated in case of no fire risk. The default rate of interest used in computations is 3%. With uneven-aged management, the optimal management regime is to carry out selection cuttings to the diameter limit of 35 cm. The development of stand basal area for a given initial state and such harvest is shown in Fig. 6.3a for the first 120 years. The selection cuttings are repeated at a 20-year interval. The retained basal areas vary from 14 to 21 m^2/ha.

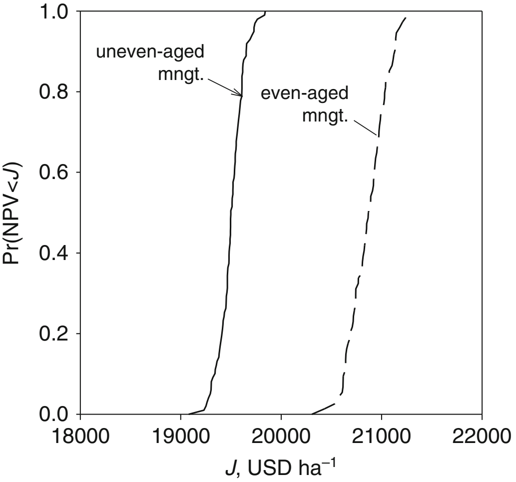


Fig. 6.4 Cumulative probability distributions of net present values for optimal continuous cover forestry and even-aged management with no risk of wildfire

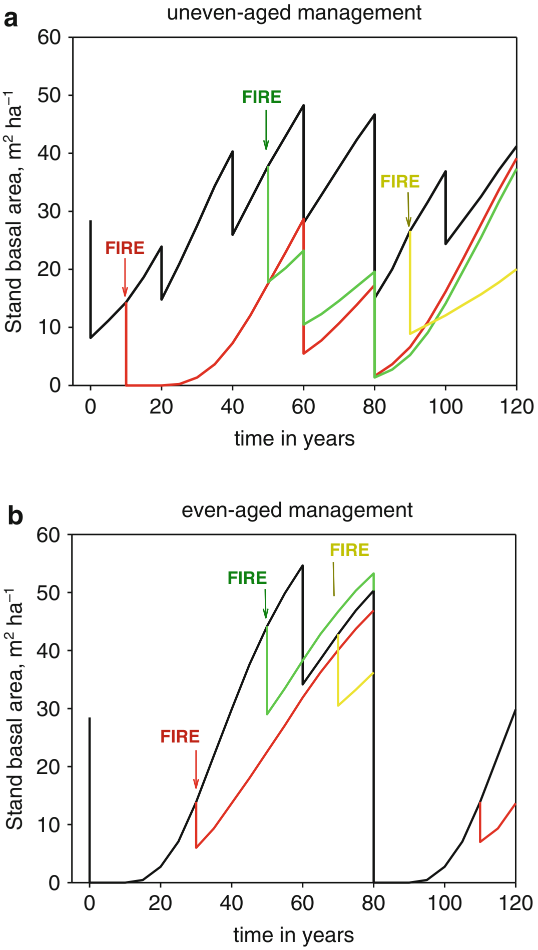
With even-aged management the optimal management regime is to plant 1,500 seedlings per hectare at the time of regeneration, to thin the stand to a residual basal area of 34 m^2/ha at a 20 year interval and to carry out the final clearcutting at stand age of 100 years. Figure 6.3b shows one projection of stand basal area for this management regime.

Figure 6.4 shows the cumulative probability distributions of net present values for the two management regimes shown in Fig. 6.3. Carrying out an immediate clearcutting, planting the stand with white pine and performing even-aged management leads to higher expected net present value than uneven-aged management. The stochastic feature in tree growth, as accounted for in FVS simulator, contributes to only a small variation in the net present values. The distributions of present values of the management systems do not overlap and even-aged management is clearly the favored system for forest owners interested in timber production. The expected net present value of uneven-aged management is about 93% of the expected net present value of even-aged management.

Inclusion of fire risk reduces the profits of forest management and alters optimal stand management. Risk of losing commercial trees to fire damage makes it rational to cut more trees earlier. Under uneven-aged management, inclusion of fire risk of $\lambda = 0.4$ reduced the diameter limit of selection cutting from 35 to 30 cm. Under even-aged management, inclusion of fire risk reduced planting density from 1,500 to 1,000 trees per hectare and shortened the rotation length from 100 to 80 years.

Figure 6.5 demonstrates the effects of alternative realizations of wildfires on stand development for optimal harvesting regimes of both management systems. Figure 6.5a shows one stand development without fires (black line) and three developments with one fire event in each for optimal continuous cover forestry. The red, green and yellow lines denote stand developments for fires occurring at 10, 50 and 90 years after the initial time. The level of mortality and the size of trees killed by a wildfire depend highly on stand properties. Wildfire causes high mortality if the

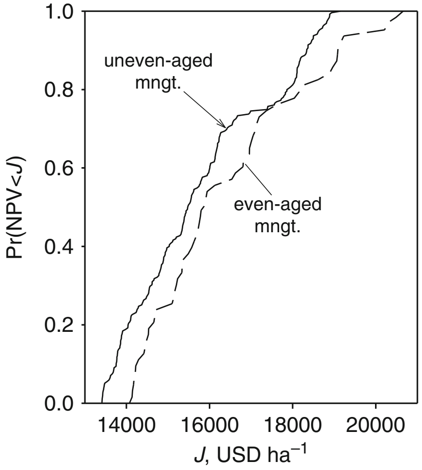
Fig. 6.5 Development of stand basal area for optimal regimes under four different realizations of wildfires. The *black line* denotes stand management without fires and *red*, *green* and *yellow lines* the developments where one wildfire occurs 30, 50 and 70 years after the initial time, respectively



levels of ladder and crown fuels are high enough to trigger an active crown fire. This is the case in stands that are dominated by young trees. In Fig. 6.5a, full mortality occurs if a fire occurs 10 years after the first selection harvest. At this stage, the stand has a large number of small understory trees and ladder fuels following a heavy diameter limit cut 10 years earlier. However, in uneven-aged stands that are dominated by older and taller trees, ladder fuels are limited making those stands less vulnerable to fire damage. This is the obvious reason why later fires (at 50 and 90 years after initial time) lead only to partial damage.

Figure 6.5b shows corresponding stand developments for even-aged management. Under even-aged management, tree mortality is 100% if wildfire occurs at a seedling phase (not shown). However, tree mortality declines with stand age and elevated canopy base. Fires occurring at later ages lead to only partial damage. Also, the level of mortality under even-aged management is lower compared with the mortality under uneven-aged management.

Fig. 6.6 Cumulative probability distributions of net present values under the optimal management for continuous cover forestry and even-aged management with a $\lambda = 0.4$ risk of wildfire



Once a wildfire occurs, it typically reduces the amount of harvestable timber and causes economic damage. In some exceptional cases wildfires may have positive economic effects by reducing the number of less valuable species and reducing the damage of future fires by consuming fuels. The cumulative probability distributions of net present values with wildfire risk of $\lambda = 0.4$ are shown in Fig. 6.6. Fire risk reduces expected net present value and widens the probability distribution (compare Figs. 6.4 and 6.6). Another important consequence is that the probability distributions for the two management systems are now overlapping. The expected net present value is still higher with even-aged management. However, the difference in the expected values is small compared to the variation in net present values. This implies that in regions with high fire risk, the choice of management system may be economically less important than considering ways to reduce the risk of wildfire.

The interest rate is an important factor affecting the relative efficiency of alternative management systems. Figure 6.7 shows the ratios of expected net present values of converting the stand to even- and uneven-aged management under alternative rates of interest. The values exceeding one indicate that even-aged management is superior to uneven-aged management. Two levels of fire risk are considered: no risk ($\lambda = 0$) and high risk ($\lambda = 0.4$). Even-aged management is the superior management system at low rates of interest less than or equal to 3%, while uneven-aged management is superior for higher rates of interest. The most obvious explanation is that small but frequent revenues from uneven-aged management become more favorable than larger and infrequent revenues from even-aged management at high rates of interest (Chang 1981).

According to Fig. 6.7, an increasing fire risk improves the relative efficiency of even-aged management for both low rates of interest ($r \leq 0.02$) and high rates ($r \leq 0.05$). Fire risk improves the relative profitability of even-aged management at high rates of interest because even-aged management allows for immediate harvest

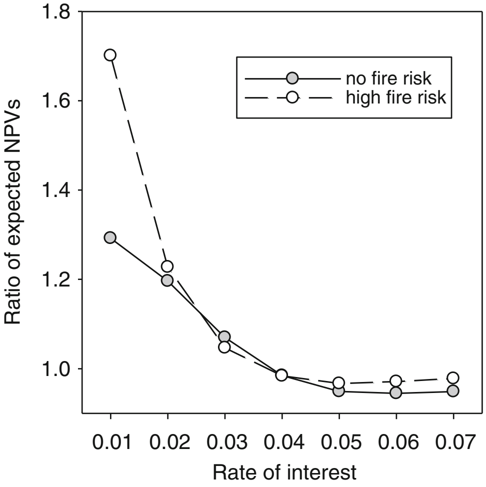


Fig. 6.7 The ratio of expected net present values between even-aged management and continuous cover forestry for two levels (zero and high) of wildfire risk

of the initial stock without the risk of losing any trees. However, this result is likely to change if the forest stand under consideration is younger and not yet mature for clearcutting. At low rates of interest, fire risk improves the relative profitability of even-aged management because stand structures composed of an even-aged cohort of trees are, over the course of the entire rotation, less vulnerable to fire damage than uneven-aged stands containing more than one cohort. The fuel structures in even-aged stands are less likely to promote intense fires with high levels of damage than fuel structures in uneven-aged stands.

In addition to the rate of interest, there are many other factors affecting the relative profitability of management systems. One critical factor is the relative price of planted species under even-aged management in relation to prices of naturally regenerating species and the species mixture in naturally regenerating stand. Here the planted species is white pine, which is a commercially valuable species. The computations were also carried out for two other common plantation species: Douglas-fir (*Pseudotsuga menziesii*), and lodgepole pine (*Pinus contorta*). These two species show similar growth rates as white pine, but the stumpage prices are clearly lower. If either Douglas fir or lodgepole pine was the species considered for planting, uneven-aged management turned out superior to even-aged management at all rates of interest (relative present values from even-aged management varied between 0.7 and 0.95 compared to uneven-aged management). On the other hand, the good economic outcome of uneven-aged management depends on the existence of valuable naturally regenerating trees. For this case study of a mixed conifer stand in Northern Idaho, a mixture of Western red cedar, which is more valuable than any of the planted species, makes uneven-aged management a financially competitive alternative to white pine plantation.

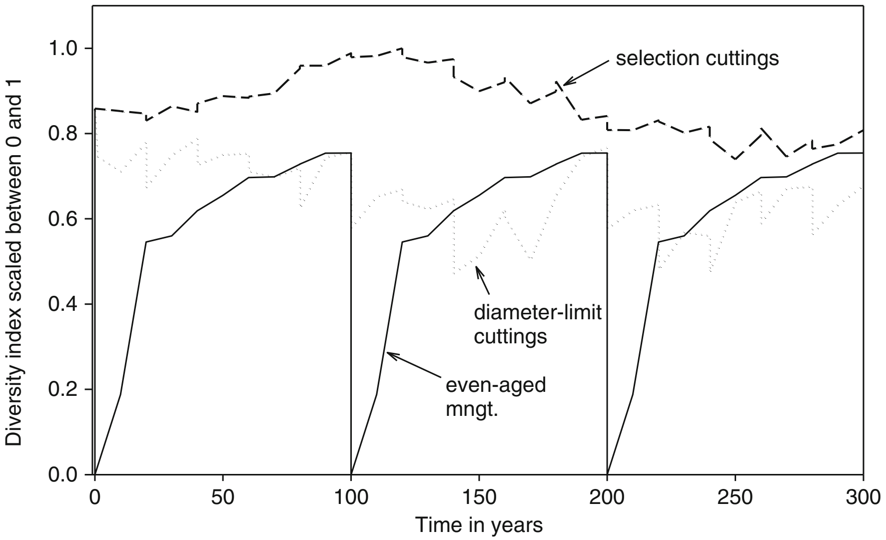


Fig. 6.8 Development of stand diversity index for a mixed conifer stand over a 300-year simulation period for three alternative chains of stand management

4.5 Environmental Considerations

Earlier examples shown in this chapter assume that timber is the primary forest product with value. Here, we investigate the consequences of alternative stand management systems on stand diversity as measured by the Shannon index (Eq. 6.12), which reflects habitat quality and describes a stand's richness with respect to tree sizes and tree species. Figure 6.8 shows the development of stand diversity index for such even-aged and uneven-aged management regimes that are optimal at a 3% rate of interest and zero risk of wildfire. In addition, development of diversity is shown for a sequence of selection harvests that develop and maintain a reverse J-shaped diameter distribution. In this regime, the residual basal area after a selection harvest is 14 m^2/ha and the q-ratio of the residual trees in adjacent diameter classes is 1.4. Trees are arranged in 10 cm diameter classes and equal weights are applied for species and size diversity ($w_s\ p = w_{size} = 0.5$). In Fig. 6.8, the Shannon diversity index values are scaled between 0 and 1 to better demonstrate the relative differences between the compared management regimes.

Based on the NPV with low interest rate, the ranking of the management systems is: (1) even-aged management, (2) uneven-aged management with diameter-limit cuts (7% reduction in expected net present value) and (3) selection harvests that maintain the inverse-J shaped diameter distribution (37% reduction in expected net present value). However, the ranking is reversed if stand diversity is used as the ranking criteria. Uneven-aged management with diameter-limit cuttings and selection harvests that maintain the inverse-J distribution retain high species and tree size diversity throughout the planning horizon. With even-aged management, on the contrary, the diversity fluctuates considerably during a rotation period. Wildfires cause temporary reductions in the diversity index for all management systems. However, these reductions are typically followed by a peak during the subsequent periods (not shown).

According to Fig. 6.8 and empirical results (Fuller et al. 2004), carefully planned selection harvests may provide more valuable flows of non-timber benefits than even-aged management. On the other hand, continuous cover forest management does not guarantee environmentally desirable forest structures. Unconstrained diameter limit cuttings may lead to very low stocking levels and stand structures that deviate considerably from those of natural forests. In the case of private landowners practicing continuous cover forestry, additional compensation for retaining the species composition and stand stocking at appropriate levels (Chap. 3) or legal restrictions banning undesirable harvesting practices may be needed to ensure adequate provision of non-timber benefits.

5 Conclusions

The problem of how to manage a forest stand optimally has been considered over centuries – at least as long as forests have been managed in an organized manner for timber production purposes.

Research on the economics of continuous cover forestry was initiated in the USA in the 1950s and expanded in the 1970s in tandem with developments in the fields of forest growth modeling and mathematical programming. The interest spread to Europe in the mid 1990s in response to the public's desire to convert conifer monocultures to mixed-species stands and increase environmental benefits (Knoke et al. 2008). Recently, studies of the economics of uneven-aged management have been undertaken worldwide (Table 6.1).

Interpretation of optimization results from the point of view of practical forestry requires careful consideration. Individual studies show results from models that are tailored to only one or a few forest stands and specific economic conditions. Replicating the computations with different initial states and soil qualities would facilitate the development of more general management guidelines. Sensitivity analysis with respect to economic variables is needed to derive guidelines for different regions and landowners. Comparison of the results obtained with alternative stand growth models would give a forestry practitioner better assurance of the reliability of model outcomes.

The relative economic performance of alternative stand management systems is driven by several factors. One critical factor for the desirability of uneven-aged management is the initial state: repeated selection harvests have been found to suit best those stands that include trees of different sizes, ages and species at the beginning of the planning horizon (Haight and Monserud 1990a; Wikström 2000). In addition, there are several economic, biological and technical factors that affect the relative profitability of even-aged vs. uneven-aged management. Important economic factors include timber prices and costs of silvicultural activities. The rate of interest also may become decisive: a low rate of interest tends to favor even-aged management and high rate of interest uneven-aged management (Chang 1981; Tarp et al. 2000; Knoke and Plusczyk 2001; Pukkala et al. 2010; Hyytiäinen and Haight 2010). Important biological factors include the amount, species composition and spatial distribution of natural regeneration (Haight and Monserud 1990b; Tarp 2000) as well as tree growth and mortality in different shading conditions (Tahvonen 2009). Technical considerations include costs of different harvesting technologies and damages to the remaining trees (Mendoza et al. 2000).

Central questions related to the financial feasibility of continuous cover forestry include the following: How costly are selection harvests compared to thinning and clearcutting of even-aged forest? How will the market demand of plantation vs. naturally regenerated tree species evolve in the future? How much and what type of logging damages do alternative stand management systems cause to the remaining trees? Does the growth of suppressed trees recover soon after more light becomes available? Is natural regeneration distributed evenly over the forest, or does regeneration occur in patches leaving large empty areas with no seedlings? Does natural regeneration include shade tolerant and economically valuable tree species? Should species mixture be steered through planting, slashing or other silvicultural activities, and how costly are these activities? How does the probability of damage after natural hazard, such as insect damage, windfall or wildfire, differ between alternative management systems?

Existing literature has addressed some, but not all of these questions. One important topic for further research, which is inadequately addressed in present papers, is the economic effects of the spatial distribution of trees. It is typical for most uneven-aged stands that tree regeneration concentrates in patches, which leads to irregular spatial arrangement of trees of different sizes. Spatial stand growth models (Kurttila 2001; Pretzsch et al. 2002) and possibly gap models (Bugmann 2001) would serve as appropriate tools for accounting for the spatial irregularities and their economic consequences in any-aged formulations. Further, spatial growth models might enhance the characterization of habitat quality and the impacts of alternative harvest strategies on habitat quality.

One striking feature in the geographical coverage of the economic studies of uneven-aged management is the scarcity of case studies specified for the fragile soils of the tropics (Boscolo et al. 1997; Mendoza et al. 2000; Conrad et al. 2005). The interesting and important feature of sensitive forest ecosystems is the dependence of sustained production capacity on the type and intensity of management (Favrichon 1998). The potential for environmentally friendly forest management is at its highest in such conditions.

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Chapter 7
Modelling Continuous Cover Forests

Jerome K. Vanclay

1 Introduction

Continuous cover forestry (CCF) is desirable for many reasons, but this silvicultural approach does add to the complexity of predicting forest growth and timber yield. Many well established techniques such as yield tables and age-based growth models – techniques that are well-established and known to be useful and reliable – are not applicable in the all-aged forests that develop with continuous cover forestry. Thus CCF requires the use of alternative techniques that do not require knowledge of the age of trees, and admit the possibility of trees of all sizes and of many species. While this is not problematic conceptually, it poses some practical challenges in gathering data and calibrating the model, especially if the model deals with interspecific competition and other species interactions.

A further complexity is the need to predict regeneration. Most models for plantations and even-aged stands accept initial stocking as an input, and regeneration does not need to be predicted. However, prediction of regeneration is central to CCF, and a CCF model used for long-term simulations must be able to predict the amount and species of regeneration. This can be a complicated undertaking, and the complexity increases with the number and diversity of species (Vanclay 1992; Weiskittel et al. 2011). Because the nature of regeneration is usually dependent on stand conditions in the immediate vicinity, long-term simulations in CCF require a modelling approach that utilizes spatial data (e.g., Newnham and Smith 1964) or simulated small gaps (e.g., *JABOWA*, Botkin et al. 1972; *Sortie*, Pacala et al. 1993).

The many examples of models for CCF make a comprehensive review with case studies a daunting exercise that this chapter does not attempt, as recent comprehensive reviews are offered elsewhere (Hasenauer 2006; Pukkala 2009; Weiskittel et al. 2011).

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_7,

2 Modelling Approaches for CCF

It is possible to model CCF using stand-based approaches such as transition matrices (Hool 1966; Bosch 1971) and stand table projection (Buongiorno and Michie 1980), but the utility of these approaches is limited. These were popular approaches before the advent of computers because they provide practical information with little computational effort (Vanclay 1994a, b). However, since these methods were pioneered, both information needs and computational possibilities have advanced, and most forest managers and modellers prefer to use individual-based rather than class-based models in CCF situations. Class-based approaches work best when a stand can be described with few states (hence, with a limited range of sizes and number of species), but several studies show that many of the assumptions may be questionable in CCF (Hulst 1979; Binkley 1980; Roberts and Hruska 1986). The approach may still have utility where data and computational resources are limited, and further guidance may be found in standard texts (Vanclay 1994a, b; Weiskittel et al. 2011). Despite these limitations, matrix approaches have been used to investigate a range of management options for CCF, including conversion from even-aged stands (e.g., Rojo and Orois 2005) and sustainable harvests (Lopez et al. 2007).

Individual-based approaches may take two forms – cohort-based and single-tree models. A single tree model typically simulates a sample plot of finite area, and simulates the appearance (germination or recruitment), the increment and the death of each individual tree within the plot. This approach is often used in spatial models, which simulate not only each individual tree, but often explicitly model spatial competition, sometimes down to the level of tracing rays of sunlight and their interception by individual leaves (e.g., Groot 2004). The disadvantage of single tree models is the need to model mortality, and to identify when a tree disappears from the simulation. The difficulty of predicting mortality precisely, coupled with the consequences of such a prediction on the remaining trees in the model, mean that single tree models are often stochastic, which may in turn limit the utility of these models for forest management (Vanclay 1991a). In Fig. 7.1, one of the challenging parts of the model, and the discriminator between cohort and individual tree models is the term $p \times n$: in an individual tree model, the n is always integer, and often 1, whereas in cohort models n can take any real number. This in turn implies that the survival probability p in an individual tree model is binary (0 or 1).

An alternative approach that overcomes the need for stochastic modelling of mortality is cohort-based modelling, where each individual tree is represented as a triplet comprising identity, size and abundance, where identity usually involves taxon (and sometimes spatial location), size is stem diameter and/or tree height, and abundance represents stocking (stems/plot) (Vanclay 1994a, b). Depending on the granularity of stocking, cohorts may represent a single hectare (so that stocking is an integer, effectively a single tree model), or hundreds of hectares of forest (Vanclay 1991a). While cohort models have been proven successful in several contexts (e.g., Vanclay 1994b, Ong and Kleine 1995), they may not offer the spatial precision

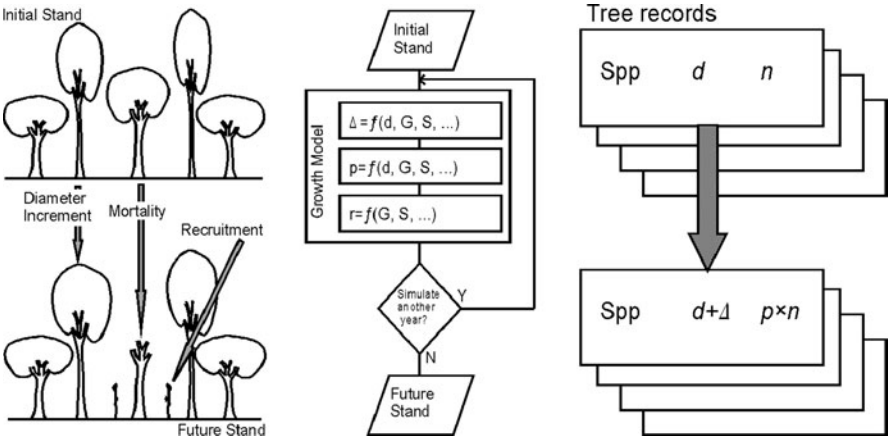

Fig. 7.1 Model components and their representation as tree records for a forest stand. Growth is modelled by incrementing the diameters in each record ($d + \Delta$) and mortality is accommodated by reducing expansion factors ($p \times n$)

needed to explore the nuances of various forms of CCF, such as different outcomes arising from individual tree selection and group selection harvesting systems.

Both cohort-based and individual tree models rely on similar components, which we will examine in turn.

3 Model Components

In a parsimonious model, it is customary to identify and model several attributes of each tree. In models that do not simulate competition explicitly, it is common to model tree size (usually stem diameter) and stocking or mortality. In contrast, models that simulate competition explicitly often model additional attributes including tree height, crown width and crown depth. In either case, a generic concept ‘size’ is refined into an attribute that is non-ambiguous, and can be both simulated and measured.

While the great majority of models take a parsimonious and explicit approach to finding the smallest number of simple relationships to describe observed changes, some modellers seek to explain observations in more detail and depth, to simulate components of tree growth in ways closer to the underlying physiology. Such models are often called physiological or mechanistic models (Landsberg 1986; Battaglia and Sands 1998). The attraction of these models is that some components of the models begin to approach stable underlying principles that may be generic and applicable to many species and situations. The disadvantage is that these approaches often require many parameters (e.g., 47 parameters for each species in the 3PG model, Sands and Landsberg 2002), including some that may be difficult to estimate.

The art of modelling is to choose a suitable level of detail, that simultaneously retains the scientific principle of falsification (able to be tested and refuted), that enables data collection and model calibration to be timely and efficient, and that achieves a utility sufficient for the model to be used to inform forest management and other user requirements. These demands are diverse, and different situations require diverse approaches.

4 Useful Relationships

There is a bewildering range of relationships that may be used in the construction of a growth model, but experience has shown that a small number of proven relationships provide robust results (Vanclay 1994a). Some of the more important of these relationships are detailed below to offer guidance for novice model builders.

4.1 Size (Diameter)

The growth in size of individuals lies at the heart of most growth models, and can involve the growth in height (e.g., Mitchell 1969), diameter or other parts of the trees, but it is perhaps most common to model diameter increment (Vanclay 1994a, b). Similarly, models can predict change in size (i.e., increment) or future size, and can predict diameter or some transformation of diameter such as cross-sectional area. Each approach has adherents, but the differences between these alternatives tend to be small, provided that the usual statistical assumptions are satisfied.

Modellers may estimate increment directly for a range of stand conditions, implicitly dealing with competition, or they may explicitly use a modifier to reduce potential growth to account for competition (e.g., Arney 1985). One of the difficulties of the latter approach is the need to obtain an independent and reliable estimate of potential growth. One approach is to rely on trees subjectively considered free of competition, but a better alternative is to estimate the potential growth and the modifier simultaneously.

One empirical equation that has been widely used to model diameter increment is based on a simple relationship (Wykoff 1990; Vanclay 1991b), with additional terms to include competition and site productivity:

$$\ln \Delta d = \beta_0 + \beta_1 \ln d + \beta_2 d^k + \beta_3 G + \beta_4 G_{>d}$$

where d is tree diameter, Δd is diameter increment, G is stand basal area, $G_{>d}$ is basal area in larger trees, and k is 1 (Vanclay 1991b) or 2 (Wykoff 1990). This equation is easy to calibrate, and predicts an increment pattern very similar to other equations with a strong biological basis (e.g., Bertalanffy 1942), but which are more difficult to calibrate (Ratkowsky 1983). Other equations that offer robust predictions are discussed by Weiskittel et al. (2011).

4.2 Competition and Species Interactions

In simple monospecific even-aged forests, competition is relatively easy to deal with, and simple stocking guides offer a useful way to estimate competition and manage forest stands (e.g., Reineke 1933; Newton 1997; Pretzsch and Biber 2005; Vanclay 2010). But dealing with competition becomes much more complicated in forest stands that are uneven-aged or comprise multiple species. Basal area in larger trees (Wykoff 1990; Vanclay 1991b; Vanclay 1994a, b) is a convenient non-spatial index of competition that is applicable and effective in complex forest stands. When spatial data (tree positions) are available, many more options are possible. Vanclay (1994a, b) classified spatial competition indices into six variants: the competitive influence zone (Ek and Monserud 1974), area potentially available (Moore et al. 1973), horizontal or vertical size–distance (Lemmon and Schumacher 1962), sky view (Bowman and Kirkpatrick 1986) and light interception (McMurtrie and Wolf 1983) approaches. Some of the more promising amongst the many alternatives available include the size-distance approaches based on the formulae of Miina and Pukkala (2000) and Hegyi (1974), but the specific calibration of these indices for individual species remains an important topic for further research.

Despite a wealth of literature on modelling intra-specific competition, there is no clear paradigm for modelling inter-specific competition and facilitation. While there is evidence of both facilitation (Forrester et al. 2006) and allelopathy (Blanco and Kimmins 2009), the dominant modelling approach tends to rely on a concept of strong and weak competitors built on competition indices pioneered for monospecific forests (e.g. Bristow et al. 2006), overlooking the possibility that species relationships may change over time (Forrester et al. 2011) and may not be consistent across species (Lhotka and Loewenstein 2011). To date, most models concentrate on direct species interactions, and overlook the indirect effects such as the effects of species competition on soils (Rothe and Binkley 2001).

4.3 Site Productivity

Forest modelling has a long tradition of recognising the importance of site productivity, and of using simple unidimensional indices such as site index in the prediction of plantation growth. While the utility and limitations of these indices is well established for plantation situations (Skovsgaard and Vanclay 2008), there is no dominant paradigm for dealing with site productivity in stands managed as CCF (Vanclay 1992) and many research questions remain outstanding. There are some indications that hyperspectral remote sensing may offer an efficient way to integrate the many dimensions of site productivity into an index amenable for stand growth modelling in CCF (Vanclay and Preston 1990; Turner et al. 2004), but progress with reliable site productivity assessment within CCF will depend on quality data drawn from long-term monitoring (Skovsgaard and Vanclay 2008). Long-term monitoring data, where available, provides a useful basis to formulate growth indices that may be used as a basis for calibrating other more empirical approaches (Vanclay 1989), and can be used to recalibrate and customise growth models (Trasobares and Pukkala 2004). Where dendrometric approaches cannot be used directly, indicator species and site descriptors may offer a practical way to estimate site productivity (Bergès et al. 2006).

4.4 Mortality

In even-aged forests, the self-thinning line provides an effective way to estimate mortality in forest stands, but the concept is of limited utility in CCF. With CCF, the best option is to predict tree survival from the resources deemed to be available to each tree. There are many ways to do this at the stand level (e.g. Vanclay 1991c) or individual tree level (Weiskittel et al. 2011). Most approaches tend to use a logistic model to predict survival from tree size and competitive status using variables such as relative size, basal area in larger trees, and crown ratio. Reviews (e.g. Hawkes 2000, Weiskittel et al. 2011) suggest that there is no single best way to deal with mortality, although there is an emerging consensus that empirical equations tend to perform better than theoretical (Bigler and Bugmann 2004b) and mechanistic approaches (Hawkes 2000).

Some models also deal with irregular mortality such as that arising from wildfire, pests and disease (e.g., Kobziar et al. 2006; Vega et al. 2011).

4.5 Regeneration and Recruitment

With industrial plantations, stocking is given and there is no need to model regeneration, making for a simple and robust model, but the modelling of regeneration is critical for models intending to model CCF over long time intervals. Depending on the context, regeneration models may begin with flowering and pollination, with seeding, regeneration, or recruitment (Weiskittel et al. 2011), but the most common approach in models used for informing forest management is to begin with established regeneration, often when saplings reach breast height (1.3 m) or a larger height threshold. Such models often entail two stages, reflecting the probability of a regeneration event and the abundance of regeneration given that an event occurs. The difficulty of predicting regeneration has spawned a diversity of approaches, and it is difficult to recommend any particular approach because the most promising approach depends on the forest type involved. For instance, Vanclay (1992) predicted the probability of a regeneration event using logistic equations, Hasenauer et al. (2001) used artificial neural networks, and Vickers et al. (2011) used an expert system to initiate regeneration. Similarly, modellers are divided about how to deal with regeneration once it is predicted: Vanclay (1992) recruited regeneration directly into the main model, whereas Monserud and Ek (1977) maintained a separate regeneration submodel, recruiting to the main model when trees were estimated to have reached 7.6 metres height. Miina and Heinonen (2008) offered a recent example of a stochastic regeneration simulator. The enduring principles that should guide modellers is the principle of parsimony (as simple as possible), of science (refutable), and of utility (fit for purpose).

4.6 Merchantability and Hollow Formation

It is insufficient simply to predict the existence and size of trees in a forest stand when CCF is practiced, because if is also important to estimate the proportion of merchantable timber, and to estimate some of the habitat services offered by individual trees. Despite this important need, there are relatively few such models. Strub et al. (1986) and Vanclay (1991d) offer some of the few examples of the modelling of merchantability of individual tree stems.

CCF is often adopted in favour of other silviculture alternatives because of the greater provision of environmental services, so it is important to be able to estimate progress towards these goals. In many situations, one of the key goals is the availability of hollows suitable for hollow-dependent fauna, and thus it may be useful for a CCF model to explicitly model some characteristics of tree hollows in the stand under simulation. Given the importance of these aspects, there are surprisingly few examples of such models, but representative examples are offered by Ball et al. (1999) and Ranius et al. (2009). Pukkala et al. (2005) offered a model for the spread of butt rot in even-aged conifers.

4.7 Physiological Relationships

A review of models for CCF would be incomplete without mention of mechanistic models and the relationships therein, but the task is complicated by the vast diversity of approached offered by the various adherents. Lacointe (2000) offered a comprehensive review of carbon allocation, and Weiskittel et al. (2011) offered a synthesis from a forest management viewpoint. The 3PGpjs variant (Sands 2004a) of the 3PG model (Sands and Landsberg 2002) is a widely-used open-source model that has been well documented (e.g. Almeida et al. 2004), including advice on the data gathering and model calibration procedures (Sands 2004b) needed to adapt this model for new species. This model has been used widely (e.g., Roxburgh et al. 2006; Coops et al. 2011), but primarily for even-aged plantations.

5 Validation and Implementation

Model design is but one aspect of a reliable model, and model calibration and evaluation are equally important in ensuring a serviceable and reliable model. Much has been written on model evaluation (e.g., Vanclay and Skovsgaard 1997; Weiskittel et al. 2011) and readers are directed there for technical aspects of this process which is the same for CCF models as with other models. It suffices to underscore that good data and reliable relationships are necessary, but insufficient to ensure a reliable model without careful calibration and effective evaluation.

Many of the classic growth models were implemented with thousands of lines of computer code, often Fortran (e.g., Botkin et al. 1972) or C++ (Congleton et al. 1997), and this code is sometimes re-used (Salminen et al. 2005). However, efficiencies can be gained through object-oriented programming (Sequeira et al. 1991), the adoption of modular structures (Reynolds and Acock 1997), and the use of visual modelling environments such as STELLA (Costanza et al. 1998) and Simile (Muetzelfeldt and Massheder 2003). Garcia (2003) has also offered practical suggestions for reducing the dimensionality of simulation models.

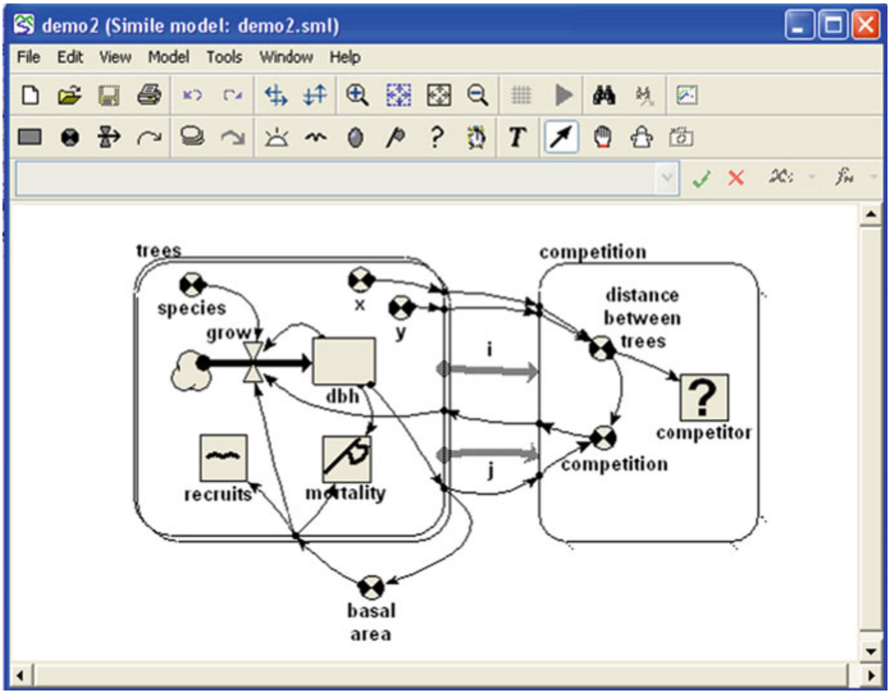

Fig. 7.2 An spatially-specific individual tree model implemented in the *Simile* modelling environment

In the past, much effort was devoted to finessing computer code to reduce memory requirements and run times, but with the advent of modern computing, these aspects are less critical and the primary consideration should be ease of understanding and of maintenance (enhancing, updating and adapting to new computing facilities). These aspects are often overlooked, but can greatly affect the utility of a model and should not be neglected. Fortunately, modern computing resources such as visual modelling environments greatly facilitate this aspect of modelling.

Visual modelling environments such as *Simile* (Muetzelfeldt and Massheder 2003), offer great potential for the rapid development, prototyping and testing of simulation models (Vanclay 2003). Figure 7.2 illustrates one example that was simple and quick to compile, but that implemented a sophisticated individual tree model to illustrate the capability of *Simile* and similar systems. This figure looks like an explanatory diagram, but is actually a model that simulates when the 'play' button (►) is pressed; it looks deceptively simple, but is in fact a sophisticated individual tree model. The point of this figure is to illustrate that powerful tools such as *Simile* make advanced modelling concepts accessible to a broader range of model builders and model users. Other more complex examples of spatially-explicit models of mixed species forests have been presented by Vanclay (2006).

6 Conclusion

There are well-established precedents for modelling uneven-aged and mixed-species forests that offer useful guidance for modelling continuous cover forests, but many challenges remain. Fertile areas for further research include site productivity assessment and the modelling of regeneration and species interactions (both synergism and allelopathy). Despite these challenges, there are good precedents for modelling timber production from CCF systems, but there remains a need for further research and development in modelling non-timber products and environmental services from these forests, and to scale up to evaluate landscape-scale implications of management options (Pretzsch et al. 2008).

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Chapter 8
Bootstrap Simulation, Markov Decision Process Models, and Role of Discounting in the Valuation of Ecological Criteria in Uneven-Aged Forest Management

Mo Zhou, Joseph Buongiorno, and Jingjing Liang

1 Introduction

Besides the market value of timber, forests provide substantial nonmarket benefits, especially with continuous-cover silviculture, which have long been acknowledged by forest managers. They include wildlife habitat (e.g. Bevers and Hof 1999), carbon sequestration (e.g. Dewar and Cannell 1992), biodiversity (e.g. Kangas and Kuusipalo 1993; Austin and Meyers 1999), landscape diversity (e.g. Gobster 1999), etc.

Several studies have explored the trade-offs between economic and noneconomic objectives. For example, Liang et al. (2006) find with a response surface approach that annual production is negatively related to tree size diversity. Seely et al. (2004) use a hierarchical decision-support system to combine multiple management objectives such as gross profit, carbon storage, and snag density. Zhou and Buongiorno (2006) and Zhou et al. (2008a, b) explore with Markov decision process (MDP) models how to pursue uneven-aged management under risk with multiple criteria including discounted net returns, landscape diversity, and wildlife habitat. With discounted financial returns in the objective function and undiscounted criteria as constraints, such methods offer a means of computing the shadow price (opportunity cost) of the undiscounted criteria.

Discounting net financial returns has been common practice in forest management, dating back to the ground-breaking Faustmann's formula (Faustmann 1849) which values lands under perpetual even-aged management. Chang (1981) derives the optimal cutting cycle for uneven-aged management from a formulation analog

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_8,

to the Faustmann formula. Forest rent and internal rate of return have been proposed as alternative economic criteria (e.g. Bentley and Roger 1966; Bierman 1968), but both are internally faulty or problematic (Newman 1988).

Buongiorno (2001) shows that Faustmann's formula can be viewed as a particular case of a more general MDP model with stochastic stand growth and prices, applicable to both even-aged and uneven-aged forests. Examples of applications of MDP models to forest management include Lembersky and Johnson (1975), Kao (1984), Kaya and Buongiorno (1987, 1989), and Lin and Buongiorno (1998). Real options approaches have also been applied to deal with discounted financial objectives with stochastic timber prices (e.g. Newman and Yin 1995; Plantinga 1998; Insley 2002).

However, disagreements lie in whether or not to discount nonmarket values in the same way as economic returns. Hartman (1976) broadens the general Faustmann formula to include the discounted environmental benefits as a return in the valuation stream (Newman 1988). Howarth (2009) argues that nonmarket values provided by public goods (such as public forests) should be discounted at a risk-free rate, just like financial returns. Still, most of the studies incorporating ecological criteria in forest management do not discount them, especially if they are not expressed in monetary terms (for an exception see Boscolo et al. (1997) which discount tons of carbon stored in forest stands).

However, discounting is just a weighting scheme which gives more importance to the present than to the future. Although it is usually used to discount future benefits and thus encourage resource depletion (Clark 1988), discounting could be used to discount future stock and thus promote conservation. By giving more weight to the present state of a forest (say its biodiversity) than to its future state, discounting "may work in favor of efforts to preserve natural resources in the future" (Gowdy 1996).

The focus of this chapter is to explore the effect of discounting on some characteristics of forests that are deemed valuable by ecologists, without arguing in favor or against discounting per se. We do this with stochastic simulation models of forest stands and MDP models for optimization. MDP models recognize risk in the growth of forest stands and in the price of wood with transition probabilities between stand states and price levels. Expected rewards from decisions include financial incomes from cutting trees and the ecological characteristics of the resulting stand states. Optimal policies from optimization of MDP models tie harvest decisions to stand states and price levels.

The objective is to quantify the effect of maximizing the discounted or undiscounted value of selected ecological criteria on the short and long-term characteristics of forests. We use the Douglas-fir/western hemlock forests in the United States Pacific Northwest as a case study, with the assumption of continuous-cover forestry. These forests provide timbers of superb quality and have the highest productivity in North America. They are also critical habitats to many wildlife species including reptiles, birds, and mammals. The ecological criteria examined in this chapter include the stand basal area, species and size diversity measured with Shannon's index (Shannon 1948), and percentage of the forest in late seral stage (Hummel and Calkin 2005).

2 Methods and Data

2.1 Markov Decision Process Models and Bootstrap Simulation

Markov decision process models deal with sequential optimization of discrete-time stochastic processes (Feinberg and Schwartz 2001). The Markovian property states that the conditional probability of any future system state, such as the physical state of a stand of trees, depends only on the present state of the system (Hillier and Lieberman 2005). It is important to note that this does not necessarily imply that the system is "memoryless". Instead, it reflects the truism that the system state at a future point in time $t + 1$ can be predicted only with the information that is available at time t. This information, defining the system state at t can be quite extensive and it could in particular include past behavior of the system. Thus, Markov chain models are general and powerful representations of stochastic processes (Buongiorno and Gilless 2003, p. 352). Markov chains allow the simplification of complex, multi-dimensional stochastic processes and make their optimization easier or at all possible (Holling et al. 1986; Insley and Rollins 2005).

One difficulty of Markov models is the determination of the transition probabilities between stand states. This can be rarely done from direct observation of changes in actual stands because there are usually too few observations in some stand states, even with extensive data set like the one available for this study. However, the transition probabilities can be readily estimated with simulation models of stand growth of a suitable form to capture the complexity of stand growth, mortality and recruitment (Kaya and Buongiorno 1989). In such a simulation, bootstrapping can be used to advantage because it leaves the data as is, without imposing a particular distribution on the shocks, as is the case when multivariate normal distributions of the shocks are used instead (Liang et al. 2006).

The final MDP model being optimized consists of state variables, decisions or actions by state, transition probabilities between states conditional on decisions, and expected rewards by state and decision (Winston 1991, p. 1026). Here we restrict our discussion to MDP models with finite states over an infinite planning horizon. The objective can be either the expected average reward or the total discounted rewards over an infinite period. The optimization of a MDP model can be done with linear programming or dynamic programming.

2.2 Forest Type Under Examination

The forest type studied here is Douglas-fir/western hemlock (*Pseudotsuga menziesii*/*Tsuga heterophylla*) forests in the Pacific Northwest of the United States. This forest type is the source of the highest quality wood in North America, exceeding the biomass and productivity of any temperate or tropical forest (Franklin 1988).

The dominant species are Douglas-fir with a wood of superior mechanical properties (Barbour and Kellogg 1990), and western hemlock which, although less valuable than Douglas fir, is also commercially important (Burns and Honkala 1990; Zhou et al. 2008a).

Douglas-fir/western hemlock forest (Photo: Jingjing Liang)

Even-aged silviculture is most common in this forest type. However, there is growing interest in alternative silvicultural methods (O'Hara 1998; Miller and Emmingham 2001). Intense attention is also being paid to late-seral stands, for their inherent value for wildlife habitat and fonts of biodiversity (Hummel and Calkin 2005). Zhou et al. (2008b) study the long-run effects of implementing diameter caps policy for Douglas-fir/western hemlock forests – keeping all trees equal to or above 41 cm of diameter at breast height, in the context of uneven-aged/continuous cover forestry. Their finding suggests that diameter caps policy, while halving the net present value of timber, would significantly increase the area of late seral forests and of the availability of spotted-owl nesting habitat.

Late-seral Douglas-fir/western hemlock forest (Photo: Jingjing Liang)

2.3 Simulation and Markov Model of Forest Growth and Management

To “bring the real world into the laboratory” (Holling et al. 1986), we used a detailed stochastic simulation model of forest growth. This model was compacted into an equivalent Markov model of forest growth by bootstrapping. Decisions and attendant ecological rewards were then introduced to create a MDP model that could be readily optimized by linear programming.

In this MPD model of forest growth and management, the state variables were forest stand states; decisions were harvests given a state; transition probabilities defined changes between states after harvest or without harvest. Expected rewards were ecological criteria arising from stand states and decisions.

The Markov model to predict forest growth was built from 14,794 plots in Oregon and Washington (Zhou et al. 2008b). These plots were all classified in the Douglas-fir/western hemlock forest type. They were mostly in western Washington and Oregon, the Blue Mountains and Columbia Plateau of northeastern Oregon, and the Colville National Forest in northeastern Washington (Fig. 8.1). More than 60% of the plots were located on Federal lands within the range of the Northern Spotted Owl (NWP Regional Ecosystem Office 2005).

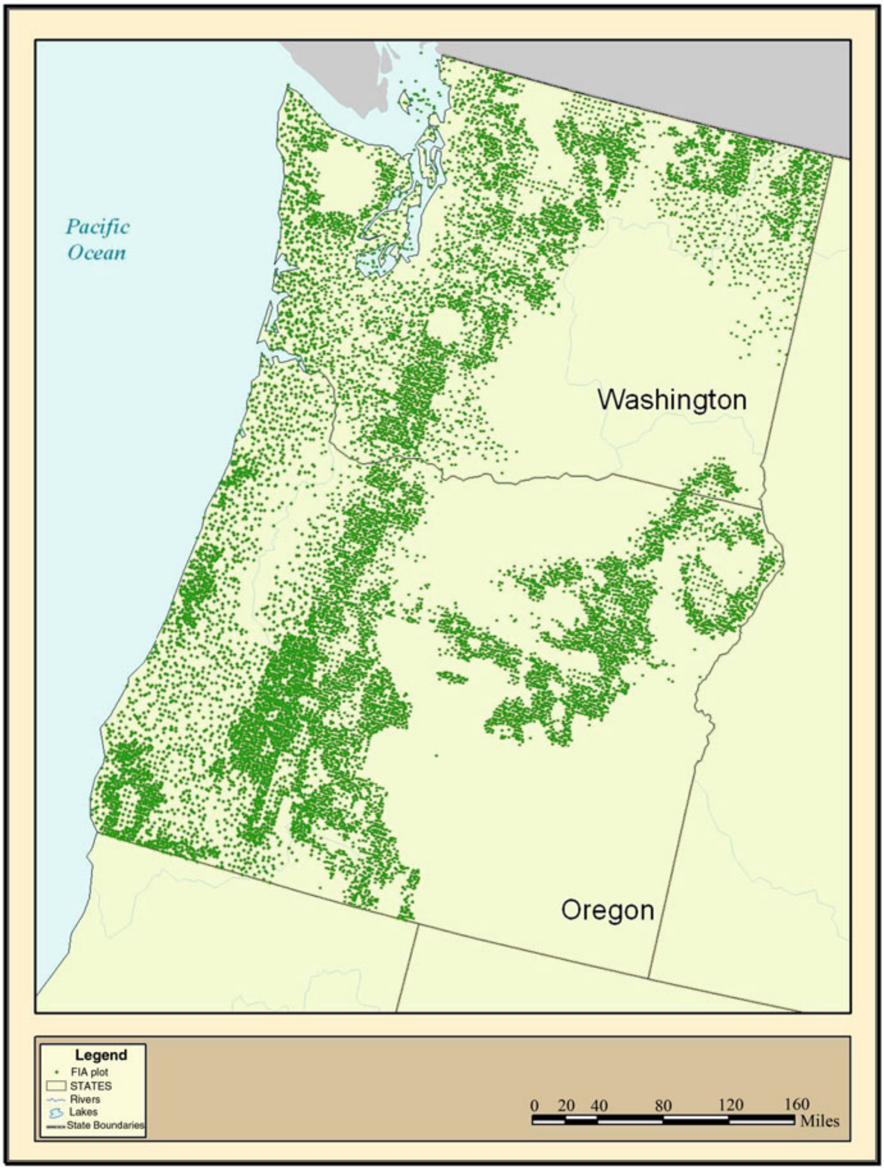

Fig. 8.1 Geographic distribution of the 14,794 FIA plots

The MDP model recognized two species groups. The shade-intolerant species consisted mostly of Douglas-fir (*Pseudotsuga menziesii*, 41% of the number of trees larger than 5.08 cm (2 in.) in diameter according to the Forest Inventory and Analysis (FIA) data), red alder (*Alnus rubra*, 11%), and ponderosa pine (*Pinus ponderosa*, 7%). The shade-tolerant species consisted of western hemlock

Table 8.1 Frequency of tree species in 14,794 FIA plots

Common name	Scientific name[a]	%
Shade-intolerant species		65.34
Douglas-fir	*Pseudotsuga menziesii*	41.27
Red alder	*Alnus rubra*	10.85
Ponderosa pine	*Pinus ponderosa*	7.10
Lodgepole pine	*Pinus contorta*	1.94
Western larch	*Larix occidentalis*	0.92
Black cottonwood	*Populus balsamifera* ssp.*trichocarpa*	0.72
Pacific madrone	*Arbutus menziesii*	0.63
Incense-cedar	*Calocedrus decurrens*	0.44
Oregon white oak	*Quercus garryana*	0.42
Western juniper	*Juniperus occidentalis*	0.26
Oregon ash	*Fraxinus latifolia*	0.25
Noble fir	*Abies procera*	0.18
Quaking aspen	*Populus tremuloides*	0.15
Western white pine	*Pinus monticola*	0.14
Sugar pine	*Pinus lambertiana*	0.05
Jeffrey pine	*Pinus jeffreyi*	0.01
Shade-tolerant Species		34.66
Western hemlock	*Tsuga heterophylla*	18.80
Western redcedar	*Thuja plicata*	5.88
Bigleaf maple	*Acer macrophyllum*	3.09
Grand fir	*Abies grandis*	2.09
Sitka spruce	*Picea sitchensis*	1.44
Pacific silver fir	*Abies amabilis*	1.32
White fir	*Abies concolor*	0.94
Mountain hemlock	*Tsuga mertensiana*	0.42
Engelmann spruce	*Picea engelmannii*	0.34
Subalpine fir	*Abies lasiocarpa*	0.17
Port-Orford-cedar	*Chamaecyparis lawsoniana*	0.12
Pacific yew	*Taxus brevifolia*	0.03
Alaska yellowcedar	*Chamaecyparis nootkatensis*	0.02
Redwood	*Sequoia sempervirens*	0.01
Total		100.00

[a] Nomenclature per Little (1979)

(*Tsuga heterophylla*, 19%), western redcedar (*Thuja plicata*, 6%), and bigleaf maple (*Acer macrophyllum*, 3%). Table 8.1 shows the detailed distribution of trees by species. Shade-tolerant and shade-intolerant trees were further divided into three size categories: small ($10\text{ cm} \le \text{dbh} < 25\text{ cm}$), medium ($25\text{ cm} \le \text{dbh} < 41\text{ cm}$), and large ($\text{dbh} \ge 41\text{ cm}$).

For each species-size class the basal area was low (indicated by 0) if it was less than the threshold – the average basal area over the plots used to build the model, or high (indicated by 1) otherwise. The threshold for shade-intolerant species was 5.85, 5.37, 5.39 m^2/ha for small, medium, and large trees, and 3.25, 2.48, and 2.84 m^2/ha for shade-tolerant small, medium, and large trees.

Thus, each stand state was represented by a string of six digits, such as 100011. The first three digits referred to the basal area of shade-intolerant trees in the small, medium, and large trees, while the last three digits referred to the same for the shade-tolerant trees. With two levels in each of the six species-size categories, there were $2^6 = 64$ possible stand states. Figure 8.2 shows the expected basal area by tree size and species group in each stand state.

Stand growth was described by a matrix giving the probability that a stand would move from one state to another in 1 year (Table 8.2). For example, in 1 year, a stand in state 1 would stay in state 1 with a probability of 0.79, move to state 3 with probability 0.01, move to state 5 with probability 0.06, and so on.

The matrix of transition probabilities between stand states, $\mathbf{T} = [p(s'|s)]$, where $p(s'|s)$ is the annual probability of moving from state s to s', was obtained by stochastic simulation with bootstrapping. The simulation was based on a stochastic nonlinear matrix growth model (Liang et al. 2006):

$$\mathbf{y}_{t+1} = \mathbf{G}\mathbf{y}_t + \mathbf{r} + \mathbf{u}_{t+1} \tag{8.1}$$

where $\mathbf{y}_t = [y_{ijt}]$ is a column vector of the number of trees per *ha* of species group i (Douglas-fir and other shade-intolerant species, or western hemlock and other shade-tolerant species), and diameter class j (nineteen 5.1 cm diameter classes ranging from 7.6 to 99.1 cm and above), at time t. $\mathbf{G}$ is a matrix of transition rates, $\mathbf{r}$ is a column vector representing the recruitment between year t and $t+1$. Both $\mathbf{G}$ and $\mathbf{r}$ depend on $\mathbf{y}_t$, making the model nonlinear.

The elements of $\mathbf{G}$ an $\mathbf{r}$ were obtained with equations that predict individual tree growth, mortality, and stand recruitment as functions of tree and stand characteristics. The stand characteristics include species diversity, size diversity, stand basal area, and tree size (Liang et al. 2005). Diversity was measured with Shannon's index (Shannon 1948). The equations were estimated with individual tree and stand data from permanent plots in western Washington and Oregon. The detailed model structure and the related parameters are in the Appendix. Validation tests comparing the deterministic predictions with observations are reported in Liang et al. (2005).

The stochastic shock vector $\mathbf{u}_{t+1}$ on a particular plot was the difference between the observed and predicted number of trees, $\mathbf{y}_{t+1}$. Bootstrap simulation (Liang et al. 2006; Zhou and Buongiorno 2006) was used to calculate the transition probabilities between stand states. Each replication gave the state at $t+1$ given the state at t. The state at t was a plot taken randomly with replacement from the 14,794 FIA plots. The state at $t+1$ was obtained with model (1) by drawing one shock vector, $\mathbf{u}_{t+1}$, randomly with replacement from the set of all the shock vectors. 150,000 replications gave stable estimates of the complete annual transition matrix between stand states (Table 8.2).

For each predicted stand state we also recorded the stand basal area per ha, the tree species and tree size diversity measured with Shannon's index (Shannon 1948), and whether the state met the definitions of late seral forest (LSF) (Hummel and Calkin 2005). Stand basal area, the cross-sectional area of the stem of all the trees at breast height of a forest stand, is a useful indicator of stand volume and density. The need to keep a rich biological diversity has been an enduring concern

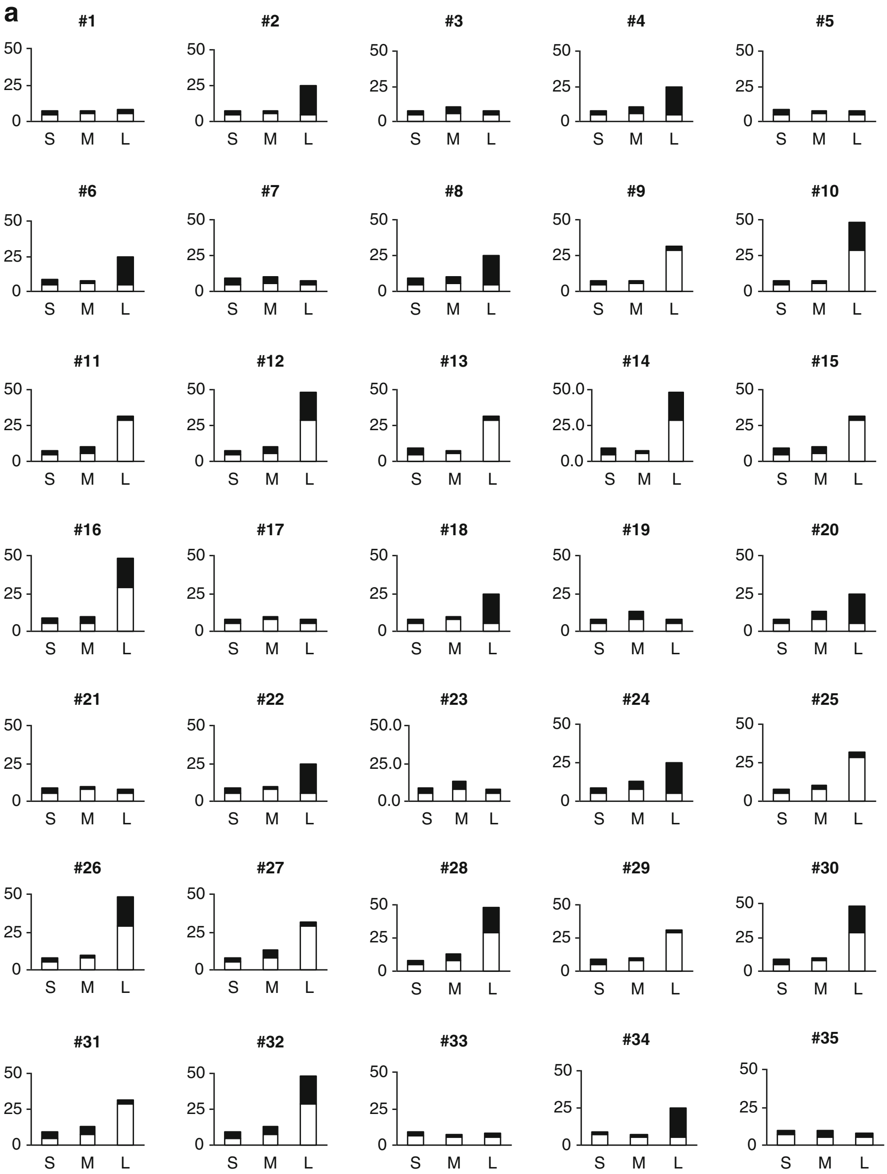

Fig. 8.2 (**a**) Expected basal area (m^2/ha) in small (*S*), medium (*M*), and large (*L*) pine trees (☐) and hardwood trees (■), by stand state #. (**b**) Expected basal area (m^2/ha) in small (S), medium (M), and large (L) pine trees (☐) and hardwood trees (■), by stand state #

of modern forest management (Hansen et al. 1991). Stands with greater vertical stratification – size diversity and rich in species composition provide critical habitats for many wildlife species (Ambuel and Temple 1983; Liang et al. 2006), and are

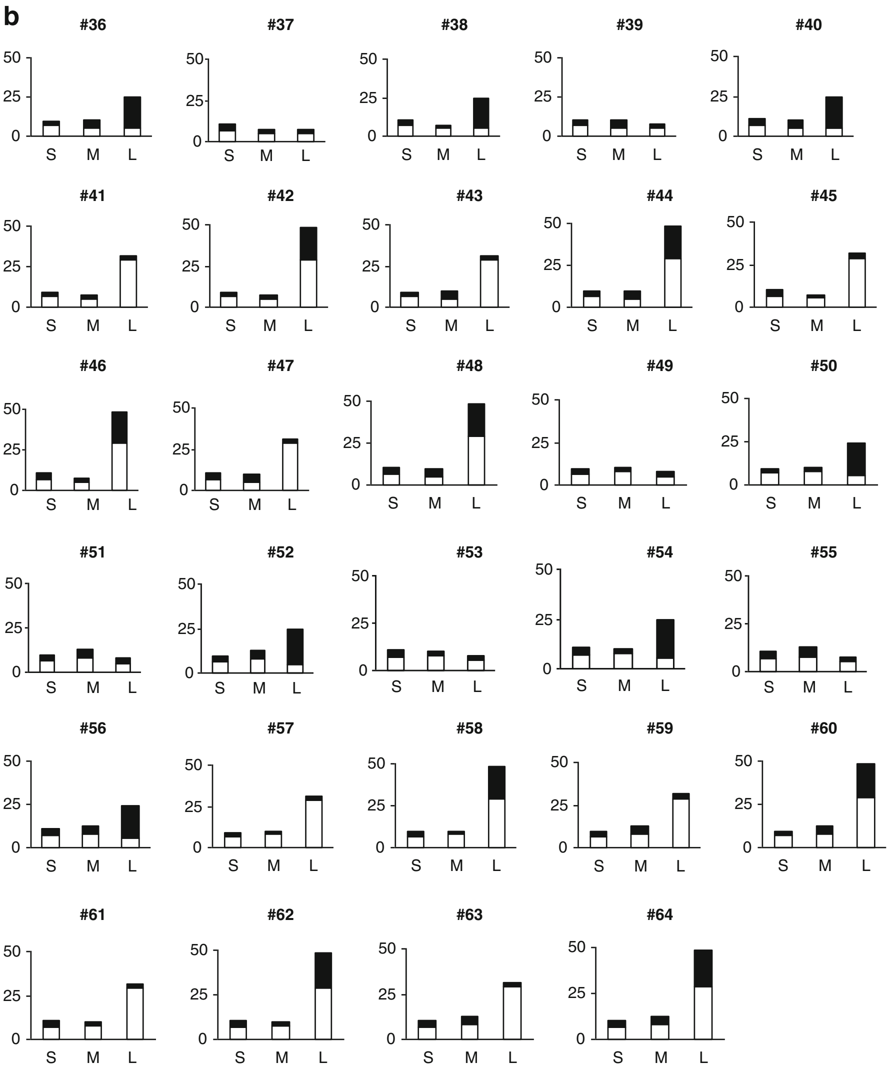

Fig. 8.2 (continued)

more aesthetically desirable (Guldin 1996). Late seral (old and mature) forests are crucial to the conservation of temperate biodiversity in the Pacific Northwest (Jiang et al. 2004).

The resulting expected values of the criteria, calculated as the mean over the 150,000 replications are in Table 8.3, together with the frequency of each stand state in the data. The expected basal area by stand state ranged from about 18 m^2/ha for state 1 (000000) with low basal area in all species and size categories, to 67 m^2/ha for state 46 (101101) with high basal area in the smallest and largest tree classes of both species groups. The expected basal area of large trees, 41 cm (16 in.) and

Table 8.2 Transition probabilities between stand states (Zhou et al. 2008a, b)

State #	State at t	State # at $t+1$ year (transition probability)
1	000000	1(.79), 3(.01), 5(.06), 17(.01), 33(.11)
2	000001	2(.78), 4(.03), 6(.07), 10(.02), 14(.01), 18(.04), 34(.04)
3	000010	1(.02), 3(.80), 4(.02), 7(.07), 11(.01), 19(.03), 35(.03)
4	000011	2(.01), 4(.88), 8(.05), 12(.01), 20(.02), 36(.02)
5	000100	1(.04), 5(.82), 7(.07), 21(.02), 37(.04)
6	000101	2(.03), 4(.01), 6(.82), 8(.04), 10(.01), 14(.02), 22(.03), 34(.01), 38(.04)
7	000110	3(.03), 5(.01), 7(.84), 8(.03), 15(.01), 23(.03), 39(.04)
8	000111	4(.02), 8(.91), 16(.01), 24(.02), 40(.03)
9	001000	9(.83), 10(.04), 11(.03), 12(.01), 13(.03), 25(.02), 41(.03)
10	001001	10(.88), 12(.02), 14(.04), 26(.02), 42(.03)
11	001010	9(.04), 11(.76), 12(.07), 15(.02), 27(.05), 43(.03)
12	001011	10(.04), 12(.84), 16(.04), 28(.02), 44(.04)
13	001100	9(.02), 13(.83), 14(.03), 15(.04), 29(.04), 45(.03)
14	001101	10(.03), 14(.88), 16(.03), 30(.02), 46(.04)
15	001110	11(.02), 13(.01), 15(.83), 16(.08), 31(.02), 47(.03)
16	001111	12(.02), 14(.02), 16(.90), 32(.02), 48(.04)
17	010000	1(.02), 17(.81), 19(.03), 21(.03), 25(.04), 49(.06)
18	010001	2(.04), 18(.76), 20(.05), 22(.04), 26(.04), 50(.05)
19	010010	3(.03), 17(.03), 19(.76), 20(.02), 23(.04), 27(.05), 51(.04)
20	010011	4(.04), 18(.02), 20(.81), 24(.04), 28(.03), 52(.05)
21	010100	5(.03), 17(.03), 21(.77), 23(.07), 29(.03), 53(.05), 55(.01)
22	010101	6(.04), 18(.04), 22(.77), 24(.05), 26(.01), 30(.04), 54(.05)
23	010110	7(.04), 19(.03), 21(.01), 23(.78), 24(.03), 31(.05), 55(.05)
24	010111	8(.04), 20(.02), 22(.01), 24(.85), 32(.02), 56(.04)
25	011000	9(.02), 25(.84), 26(.03), 27(.02), 29(.04), 57(.04)
26	011001	10(.05), 26(.84), 28(.02), 30(.04), 58(.04)
27	011010	11(.02), 25(.03), 27(.80), 28(.06), 31(.04), 59(.03)
28	011011	12(.05), 26(.04), 28(.82), 32(.04), 60(.03)
29	011100	13(.03), 25(.03), 29(.82), 30(.03), 31(.03), 61(.04)
30	011101	14(.06), 26(.03), 30(.84), 32(.03), 62(.04)
31	011110	15(.01), 27(.02), 29(.01), 31(.85), 32(.05), 63(.04)
32	011111	16(.05), 28(.03), 30(.02), 32(.85), 64(.04)
33	100000	1(.04), 33(.85), 35(.01), 37(.03), 49(.06)
34	100001	2(.05), 6(.01), 18(.01), 34(.74), 36(.05), 38(.06), 42(.01), 50(.08),
35	100010	3(.02), 19(.01), 33(.03), 35(.75), 36(.02), 39(.04), 51(.10),
36	100011	4(.05), 34(.03), 36(.80), 40(.05), 44(.02), 52(.05)
37	100100	5(.03), 33(.03), 37(.78), 39(.05), 53(.08)
38	100101	6(.07), 8(.01), 34(.01), 36(.01), 38(.75), 40(.06), 46(.03), 48(.01), 54(.05), 56(.01)
39	100110	7(.04), 35(.02), 37(.01), 39(.80), 40(.02), 47(.01), 55(.09)
40	100111	8(.05), 24(.01), 36(.02), 38(.01), 40(.85), 48(.01), 56(.04)
41	101000	9(.04), 41(.73), 42(.07), 43(.02), 45(.03), 57(.08), 59(.01)
42	101001	10(.05), 42(.86), 46(.04), 58(.02)
43	101010	11(.03), 12(.01), 41(.04), 42(.01), 43(.73), 44(.08), 47(.05), 59(.02)
44	101011	12(.06), 42(.04), 44(.81), 46(.01), 48(.04), 60(.02)

(continued)

Table 8.2 (continued)

State #	State at *t*	State # at $t + 1$ year (transition probability)
45	101100	13(.04), 15(.01), 41(.04), 45(.77), 46(.05), 47(.03), 61(.06)
46	101101	14(.05), 42(.03), 46(.88), 48(.02), 62(.02)
47	101110	15(.02), 16(.01), 43(.03), 45(.01), 47(.77), 48(.07), 61(.01), 63(.05)
48	101111	16(.04), 44(.02), 46(.02), 48(.88), 64(.02)
49	110000	17(.03), 33(.01), 49(.87), 51(.02), 53(.04), 57(.02)
50	110001	18(.03), 34(.01), 50(.82), 52(.03), 54(.05), 58(.04)
51	110010	19(.03), 35(.01), 49(.04), 51(.81), 52(.02), 55(.04), 59(.03),
52	110011	20(.04), 36(.02), 50(.02), 52(.86), 56(.04), 60(.02)
53	110100	21(.02), 37(.01), 49(.03), 53(.84), 55(.06), 61(.03)
54	110101	22(.02), 38(.01), 50(.02), 54(.87), 56(.03), 62(.04)
55	110110	23(.03), 39(.01), 51(.02), 53(.01), 55(.85), 56(.03), 63(.04)
56	110111	24(.03), 40(.02), 52(.02), 54(.01), 56(.90), 64(.02)
57	111000	25(.03), 41(.01), 57(.87), 58(.03), 59(.02), 61(.04)
58	111001	26(.03), 42(.02), 58(.88), 60(.02), 62(.04)
59	111010	27(.03), 57(.04), 59(.83), 60(.04), 63(.04)
60	111011	26(.01), 28(.05), 44(.02), 58(.04), 60(.81), 64(.06)
61	111100	29(.03), 57(.04), 61(.85), 62(.02), 63(.04)
62	111101	30(.03), 46(.03), 58(.04), 62(.87), 64(.02)
63	111110	31(.04), 47(.01), 59(.02), 61(.03), 63(.85), 64(.04)
64	111111	32(.04), 48(.03), 60(.02), 62(.02), 64(.88)

A state is defined by the level of basal area by tree species class and size class. For example state #58 (111001) means that basal area is high in the small, medium, and large shade-intolerant trees, low in the small and medium shade-tolerant trees, and high in the large shade-tolerant trees

above, was lowest for state 1 and highest for state 10 (001001) with high basal area in the large trees of the shade-intolerant and shade-tolerant species.

States 12 and 10 had the highest probability of having the structure of a late seral forest. In both states the shade-intolerant species (mostly Douglas-fir) had high basal area in large trees, and low basal area in the small and medium trees. In both states 12 and 10, the shade-tolerant trees (mostly western hemlock) had low basal area in the small trees, and high basal area in the large trees.

A decision means harvesting a stand and thus changing its state. For example, for stand state #4 (000011) – high basal area in medium and large shade-tolerant trees, a decision could be to do nothing, or to thin the large shade-tolerant trees to low basal area and thus move to state #3 (000010) – high basal area only in medium shade-tolerant trees, or to thin the medium shade-tolerant trees and move to state #2 (000001) – high basal area only in large shade-tolerant trees, or to thin the medium and large shade-tolerant trees and thus move to state #1 (000000) – low basal area in all six categories.

Each decision resulted in an immediate return, R_{sd}, the ecological criterion obtained from state s and decision d. After the decision, the residual stand would change state over 1 year according to the transition probabilities in Table 8.2. So, a stand would move from pre-decision state s to state s' after 1 year with probability $p(s'|s,d)$ depending on the decision, d.

Table 8.3 Probability of stand states in the study area, with their tree species diversity, tree size diversity, basal area, and probability that a stand state met late-seral forest (LSF) criteria (Hummel and Calkin 2005; Zhou et al. 2008a, b)

Stand state #	Initial prob.	Tree species div.	Tree size div.	Basal area (m^2/ha)	LSF prob.
1	0.066	1.13	2.69	18.2	0.00
2	0.002	1.33	2.81	50.4	0.01
3	0.011	1.25	2.74	28.7	0.00
4	0.011	1.32	2.80	50.2	0.06
5	0.031	1.23	2.69	20.9	0.00
6	0.002	1.34	2.85	54.0	0.01
7	0.025	1.28	2.74	30.3	0.00
8	0.027	1.32	2.83	53.7	0.02
9	0.004	1.07	2.82	49.3	0.00
10	0.024	1.15	2.78	63.6	0.52
11	0.003	1.15	2.83	49.8	0.00
12	0.021	1.19	2.80	63.5	0.83
13	0.004	1.11	2.86	52.2	0.00
14	0.024	1.16	2.80	65.0	0.18
15	0.005	1.17	2.85	52.3	0.00
16	0.043	1.21	2.84	63.9	0.14
17	0.015	1.13	2.75	32.3	0.00
18	0.002	1.33	2.81	50.4	0.00
19	0.007	1.19	2.75	37.9	0.00
20	0.007	1.35	2.79	52.9	0.04
21	0.008	1.17	2.79	35.9	0.00
22	0.002	1.34	2.85	53.8	0.00
23	0.012	1.24	2.76	39.7	0.00
24	0.016	1.34	2.80	53.8	0.04
25	0.013	1.10	2.82	50.9	0.01
26	0.018	1.16	2.82	62.4	0.06
27	0.008	1.15	2.81	51.4	0.12
28	0.018	1.21	2.81	62.0	0.10
29	0.011	1.12	2.85	53.8	0.00
30	0.017	1.19	2.84	64.5	0.05
31	0.016	1.17	2.83	54.8	0.00
32	0.032	1.21	2.84	63.1	0.19
33	0.055	1.08	2.68	21.1	0.00
34	0.001	1.33	2.86	53.5	0.00
35	0.005	1.25	2.73	31.1	0.00
36	0.004	1.34	2.84	53.7	0.01
37	0.016	1.19	2.69	26.3	0.00
38	0.001	1.34	2.85	58.7	0.00
39	0.012	1.26	2.73	34.0	0.00
40	0.011	1.33	2.85	56.4	0.00
41	0.002	1.06	2.86	55.6	0.00
42	0.013	1.15	2.82	66.5	0.02
43	0.001	1.15	2.87	54.3	0.00

(continued)

Table 8.3 (continued)

Stand state #	Initial prob.	Tree species div.	Tree size div.	Basal area (m^2/ha)	LSF prob.
44	0.011	1.19	2.85	65.3	0.02
45	0.002	1.11	2.87	57.7	0.00
46	0.017	1.18	2.84	67.3	0.00
47	0.003	1.16	2.88	54.7	0.00
48	0.028	1.22	2.86	66.8	0.00
49	0.041	1.12	2.74	35.3	0.00
50	0.003	1.31	2.83	57.4	0.00
51	0.013	1.19	2.74	40.7	0.00
52	0.010	1.34	2.80	56.6	0.00
53	0.024	1.16	2.76	38.7	0.00
54	0.004	1.35	2.86	61.9	0.00
55	0.026	1.22	2.74	44.0	0.00
56	0.025	1.34	2.81	58.9	0.00
57	0.022	1.09	2.85	55.1	0.00
58	0.025	1.17	2.85	65.0	0.01
59	0.011	1.14	2.83	56.1	0.05
60	0.015	1.20	2.85	64.3	0.02
61	0.019	1.13	2.86	57.9	0.00
62	0.023	1.19	2.87	66.6	0.00
63	0.022	1.17	2.85	58.5	0.00
64	0.037	1.22	2.87	65.8	0.00

2.4 Optimizing Discounted Criteria

The following linear program (d'Epenoux 1963; Hillier and Lieberman 2005, p. 921) was used to find the policy (harvest by stand state) that maximized the discounted value of a specific criterion over an infinite time horizon.

$$\max_{y_{sd}} \sum_{s} \sum_{d} R_{sd} . y_{sd}$$

$$s.t.$$

$$\sum_{d} y_{s'd} - \frac{1}{(1+r)} \sum_{s} \sum_{d} y_{sd}\, p\left(s' \,|s, d\right) = \pi_{s'} \quad s' = 1, \ldots, N$$

$$y_{sd} \geq 0, \quad for\; s = 0, 1, \ldots, N. \tag{8.2}$$

where R_{sd} was the immediate ecological criterion resulting from stand state s and decision d. The unknown, y_{sd}, was the discounted expected time in state s and decision d. The constant r was the discount rate, and $\pi_{s'}$ was the initial probability of each of the N initial states, s'.

Given the solution of (8.2), the policy that maximized the discounted criterion was obtained from:

$$p(d|s) = \frac{y_{sd}}{\sum_{d} y_{sd}} \tag{8.3}$$

where y_{sd} was the probability of decision d given state s, under the best policy. The policy was deterministic ($p(d|s) = 0$, or 1). Furthermore, the best policy was independent of the initial condition $\{\pi_{s'}\}$ (Hillier and Lieberman 2005, p. 921).

The steady-state probability of state s and decision d, needed to determine the long-term expected effect of the best policy, was obtained as follows. The transition probability matrix with the best policy was:

$$\mathbf{T}^* = \mathbf{DT} \tag{8.4}$$

where **D** was the decision matrix of the best policy and **T** the transition probability matrix without decision.

The steady-state probabilities of stand states under the best policy, $\mathbf{P} = [p_s]$, were then obtained by solving the system of simultaneous equations:

$$\mathbf{P} = \mathbf{PT}^* \tag{8.5}$$

Then, the probability of state s and decision d for the best policy was:

$$p_{sd} = p_s p(d|s) \tag{8.6}$$

2.5 *Optimizing Undiscounted Criteria*

Without discounting, the objective function was the expected value of the ecological criterion, over an infinite horizon. The following linear program was used to obtain the best undiscounted policy (Winston 1991, p. 1034):

$$\begin{aligned}
&\max_{z_{sd}} \sum_{s} \sum_{d} R_{sd} . p_{sd} \\
&s.t. \\
&\sum_{d} p_{s'd} - \sum_{s} \sum_{d} p_{sd}\, p\left(s' |s, d\right) = 0 \quad j = 1, \ldots, N \\
&p_{sd} \geq 0, \\
&\sum_{s} \sum_{d} p_{sd} = 1
\end{aligned} \tag{8.7}$$

where the unknown, p_{sd}, was the steady-state probability of state s and decision d.

The best policy was given by:

$$p(d|s) = \frac{p_{sd}}{\sum_{d} p_{sd}} \tag{8.8}$$

where $p(d|s)$ was the probability of decision d given state s under the best policy. As in the discounted case, the best decision was deterministic ($p(d|s) = 0$, or 1).

The steady-state probability of stand state s under the best policy was:

$$p_s = \sum_{d} p_{sd} \tag{8.9}$$

2.6 *Effects of Policies*

The immediate impact of a policy on a particular stand was R_{sd}, the ecological criterion resulting from stand state s and decision d. The steady state probability of a stand state, p_s, derived above could also be interpreted as the expected fraction of a large forest area that would be in a particular state in the long-run, under the chosen policy.

The long-term or steady-state effects of a policy on the ecological criteria (tree species diversity, tree size diversity, stand basal area, and late-seral forest probability) were computed as the expected value of the undiscounted criteria:

$$E(R) = \sum_{s} \sum_{d} p_{sd} R_{sd} \tag{8.10}$$

To assess the interim effect of a policy, the probability of stand state s after n years of applying a policy was calculated from:

$$\pi_n = \pi \mathbf{T}^n \tag{8.11}$$

where $\boldsymbol{\pi} = [\pi_s]$ was the row vector of probability of the initial state. As n increased, $\boldsymbol{\pi}_{\mathrm{n}}$ converged to $\mathbf{P} = [p_s]$, the steady-state probability of stand states.

For a particular policy, the expected cutting cycle, $E(c)$, was the expected interval between states that called for harvest. As p_s was the steady state probability of state s under that policy, the probability of being in a state that called for harvest, p_c, was:

$$p_c = \sum_{s \in S} p_s \tag{8.12}$$

where S was the set of all states that called for harvest under the policy. Then,

$$E(c) = \frac{1}{p_c} \tag{8.13}$$

was the expected cutting cycle (Kaya and Buongiorno 1989).

3 Results

3.1 Best Policies With and Without Discounting

The same discount rate of 3.8%, used previously by Zhou et al. (2008a, 2008b) to discount economic returns, was also used to discount the ecological criteria. With discounting, the maximum value of the criteria over an infinite time horizon (but not the corresponding best decisions) depends on the initial condition. For the stand states, the initial condition was defined by the frequency of stand states in the forests under study (Table 8.3).

Models (8.2) and (8.7) were used to maximize the discounted and undiscounted ecological criteria, respectively. The best policies were defined by the stand state that would be left after harvest, given the stand state at decision time (Table 8.4).

With or without discounting the policies are stationary (independent of the time of the decision), deterministic (the probability of making a decision in a particular state is 0 or 1), independent of the initial state, and dependent only on the stand state at decision time (Hillier and Lieberman 2005, p. 911 and 921).

For example, if the objective was to maximize discounted species diversity and the stand was in state 21 (010100) – high basal area in medium shade-tolerant and small shade-intolerant trees, the best decision was to cut the stand to state 5 (000100) – high basal area only in small shade-tolerant trees, by reducing the basal area of medium shade-intolerant trees (Table 8.4). If instead species diversity were not discounted, the best decision was to cut the same stand to state 17 (010000) – high basal only in medium shade-intolerant trees, reducing the basal area in small shade-tolerant trees.

When maximizing tree species diversity, 18 of the 64 stand states called for different decisions depending on whether discounting was or was not done. In maximizing tree size diversity, the decisions with and without discounting were different for five states only. When basal area was maximized, all but two stand states called for the same decisions with or without discounting. To maximize discounted LSF, 13 stand states required different decisions depending on whether LSF was discounted or not.

3.2 Immediate Impact of Discounting

Table 8.5 shows how discounting changed the ecological criteria of the stand that remained immediately after a harvest. For 20 stand states there was a difference in at least one of the criteria.

For example, when maximizing tree species diversity, for a stand in state #45 (101100) – high basal area in small and large shade-intolerant trees and small shade-tolerant trees, the policy with discounting called for cutting the stand to state #5 (000100) – high basal area only in small shade-tolerant trees, with an expected

Table 8.4 Decisions that maximized discounted (D) or undiscounted (U) ecological criteria

	Species diversity		Size diversity		Basal area (m^2/ha)		Late seral forest (%)	
Stand state	D	U	D	U	D	U	D	U
1	–	–	–	–	–	–	–	–
2	–	–	–	–	–	–	–	–
3	–	–	–	–	–	–	–	–
4	2	2	2	2	**2**	**3**	2	2
5	–	–	–	–	–	–	–	–
6	–	–	–	–	–	–	2	2
7	–	–	–	–	–	–	–	–
8	6	6	6	6	6	6	2	2
9	–	–	–	–	–	–	–	–
10	2	2	9	9	–	–	–	–
11	**3**	–	–	–	–	–	–	–
12	2	2	11	11	–	–	–	–
13	**5**	–	–	–	–	–	9	9
14	6	6	13	13	–	–	10	10
15	**7**	–	13	13	–	–	–	–
16	6	6	13	13	14	14	12	12
17	–	–	–	–	–	–	–	–
18	–	–	–	–	–	–	–	–
19	3	3	–	–	–	–	–	–
20	–	–	**18**	–	18	18	–	**18**
21	**5**	**17**	–	–	–	–	–	**17**
22	–	–	–	–	–	–	18	18
23	7	7	–	–	–	–	–	**19**
24	20	20	22	22	22	22	**20**	–
25	**9**	**17**	–	–	9	9	9	9
26	18	18	–	**25**	10	10	10	10
27	**3**	–	11	11	–	–	–	–
28	20	20	11	11	12	12	12	12
29	**5**	–	13	13	–	–	9	9
30	22	22	13	13	14	14	10	10
31	**7**	**15**	13	13	–	**15**	15	15
32	20	20	13	13	14	14	12	12
33	1	1	–	–	–	–	–	–
34	–	–	–	–	–	–	2	2
35	3	3	–	–	–	–	–	–
36	–	–	34	34	–	–	**2**	–
37	**5**	–	–	–	–	–	–	–
38	–	–	–	–	–	–	–	–
39	7	7	–	–	–	–	–	–
40	38	38	38	38	38	38	38	38
41	–	–	–	–	–	–	–	–
42	34	34	41	41	–	–	10	10
43	–	–	–	–	–	–	–	–
44	36	36	43	43	42	42	12	12

(continued)

Table 8.4 (continued)

	Species diversity		Size diversity		Basal area (m^2/ha)		Late seral forest (%)	
Stand state	D	U	D	U	D	U	D	U
45	**5**	**41**	–	–	–	–	41	41
46	38	38	45	45	–	–	10	10
47	**7**	**15**	–	–	–	–	43	43
48	38	38	47	47	46	46	12	12
49	**17**	–	17	17	–	–	**17**	–
50	18	18	34	34	–	–	–	–
51	3	3	19	19	–	–	**19**	–
52	**20**	–	34	34	50	50	**20**	**50**
53	**5**	**49**	21	21	–	–	**21**	–
54	–	–	–	–	–	–	50	50
55	7	7	**23**	–	–	–	**23**	–
56	54	54	54	54	54	54	**20**	**50**
57	**41**	**49**	41	41	41	41	41	41
58	18	18	**41**	–	42	42	10	10
59	43	43	43	43	43	43	43	43
60	**20**	**52**	43	43	42	42	12	12
61	**5**	**49**	45	45	45	45	**41**	–
62	54	54	–	**45**	46	46	10	10
63	**7**	**15**	47	47	47	47	43	43
64	54	54	47	47	46	46	**12**	–

Numbers refer to stand states resulting from a decision (harvest). Those in bold indicate decisions that differed if the criterion was discounted. Hyphens mean no harvest

species diversity of 1.23 (Table 8.3). Meanwhile, maximizing undiscounted species diversity called for cutting the stand to state 41 (101000) – high basal area in small and large shade-intolerant trees, with an expected species diversity of 1.06. Thus discounting led to a 16% immediate gain in expected species diversity. For most stand states, discounting led to higher expected tree species diversity immediately after harvest, but for stand states #25 (011000) – high basal area in medium and large shade-intolerant trees, and #57 (111000) – high basal area in all three sizes of shade-intolerant trees, it led to 5% lower expected tree diversity, which presumably was compensated by higher expected tree species diversity from subsequent decisions, leading to a higher discounted tree species diversity over an infinite time horizon.

When maximizing tree size diversity only for states #20 (01001) – high basal area in medium shade-intolerant trees and large shade-tolerant trees, and #55 (110110) – high basal area except for large shade-intolerant and shade-tolerant trees, did the policy with discounting lead to a slightly higher index of expected tree size diversity immediately after harvest.

Maximizing discounted stand basal area led to decisions that increased the expected post-harvest basal area only in pre harvest states #4 (00011) – high basal area in medium and large shade-tolerant trees, and #31 (011110) – high basal area except for small shade-intolerant and large shade-tolerant trees. State #4 called for

Table 8.5 Immediate impact of policies that maximized discounted criteria relative to those that did not discount

State		Max Tree species diversity	Tree size diversity	Basal area	Late seral forest[a]
4	000011	0.00	0.00	0.76	0.00
11	001010	0.09	0.00	0.00	0.00
13	001100	0.11	0.00	0.00	0.00
15	001110	0.09	0.00	0.00	0.00
20	010011	0.00	0.01	0.00	0.04
21	010100	0.09	0.00	0.00	0.00
25	011000	−0.05	0.00	0.00	0.00
27	011010	0.09	0.00	0.00	0.00
29	011100	0.10	0.00	0.00	0.00
31	011110	0.09	0.00	0.05	0.00
37	100100	0.03	0.00	0.00	0.00
45	101100	0.16	0.00	0.00	0.00
47	101110	0.09	0.00	0.00	0.00
49	110000	0.01	0.00	0.00	0.00
52	110011	0.01	0.00	0.00	0.04
53	110100	0.10	0.00	0.00	0.00
55	110110	0.00	0.01	0.00	0.00
56	110111	0.00	0.00	0.00	0.04
57	111000	−0.05	0.00	0.00	0.00
58	111001	0.00	0.00	0.00	0.00
60	111011	0.01	0.00	0.00	0.00
61	111100	0.10	0.00	0.00	0.00
63	111110	0.09	0.00	0.00	0.00
64	111111	0.00	0.00	0.00	0.83

[a] Increase in probability that a stand meets late seral forest criteria

harvest to state #2 (00001) – high basal area only in large shade-tolerant trees, with discounting, compared to state #3 (00010) – high basal area only in medium shade-tolerant trees, without discounting, for an immediate 76% gain of expected post-harvest basal area. State #31 called for no harvest with discounting, and harvest to state #15 (001110) – high basal area in large shade-intolerant and small and medium shade-tolerant trees, without discounting, for a 5% gain in expected basal area immediately after harvest.

In maximizing LSF probability, although the best decisions with discounting differed from those without discounting for 13 stand states (Table 8.4), discounting led to a stand with higher probability of LSF characteristics only for four stand states. The largest difference was for state #64 (111111) – high basal area in all six species-size categories, for which discounting led to stand state 12 (001011) – high basal area in large shade-intolerant and medium and large shade-tolerant trees with a 0.83 probability of LSF characteristics, while not discounting maintained state 64, with a 0.00 LSF probability.

Table 8.6 Long-term effects of maximizing discounted or undiscounted ecological criteria on expected forest characteristics

Expected forest characteristics	With discounting	Without discounting
	Max tree species diversity	
Tree species diversity	1.33	1.34
Tree size diversity	2.82	2.83
Basal area (m^2/ha)	54.16	54.85
Late seral forest (%)	1.01	0.76
Cutting cycle (year)	9.40	10.35
	Max tree size diversity	
Tree species diversity	1.11	1.11
Tree size diversity	2.85	2.85
Basal area (m^2/ha)	53.19	53.12
Late seral forest (%)	0.00	0.04
Cutting cycle (year)	7.90	7.87
	Max basal area (m^2/ha)	
Tree species diversity	1.16	1.17
Tree size diversity	2.80	2.81
Basal area (m^2/ha)	65.13	65.32
Late seral forest (%)	21.58	21.47
Cutting cycle (year)	22.33	22.32
	Max late seral forest area (%)	
Tree species diversity	1.16	1.17
Tree size diversity	2.78	2.79
Basal area (m^2/ha)	63.26	63.33
Late seral forest (%)	61.56	61.60
Cutting cycle (year)	9.87	9.87

3.3 Long-Term Effects of Discounting

Regardless of the immediate impact of a policy, the long-term effect would be different due to the cumulative effect of successive decisions over a long period of time. We present first the effects of discounting over an infinite time horizon, and then the transient effects during the first five decades.

To determine the forest characteristics that would result in the long term from discounted or undiscounted policies we computed the undiscounted expected value of the criteria, over an infinite time horizon with Eq. 8.10. Being undiscounted, this expected value was independent of the initial condition. Equation 8.13 gave the expected cutting cycle.

Discounting tree species diversity, tree size diversity, basal area, and LSF probability had a negligible effect on their long-term expected value (Table 8.6). Discounting changed the expected cutting cycle only when maximizing tree species diversity, shortening it by about 1 year.

Table 8.7 shows the expected long-term composition of the forest, defined by the distribution of stand states that would result from maximizing each ecological

Table 8.7 Long-term frequency (%) of stand states in the forest landscape, resulting from the maximization of discounted ecological criteria

	Max:			
Stand state	Tree species diversity	Tree size diversity	Basal area (m^2/ha)	Late seral forest (%)
1	0.0	0.0	0.0	0.0
2	12.1(10.6)	0.0	0.1	0.5
3	0.0	0.0	0.0	0.0
4	1.3	0.0	0.0	0.0
5	0.0	0.0	0.0	0.0
6	16.4(15.1)	0.1	0.1	0.0
7	0.0	0.1	0.0	0.0
8	0.9	0.0	0.0	0.0
9	0.0	12.6	0.1	0.2
10	0.4	0.6	21.9	58.1
11	0.0	6.3	0.0	0.1
12	0.0	0.7	4.8	30.3
13	0.0	28.3	0.1	0.0
14	0.5	1.0	29.0	2.6
15	0.0	1.8	0.1	0.1
16	0.0	0.2	1.1	1.6
17	0.0	0.0	0.0	0.0
18	13.0	0.0	0.0	0.2
19	0.0	0.0	0.0	0.0
20	14.5(10.3)	0.0	0.0	0.1
21	0.0	0.0	0.0	0.0
22	7.1	0.0	0.0	0.0
23	0.0	0.1	0.0	0.0
24	1.2	0.0	0.0	0.0
25	0.0	2.2(2.7)	0.0	0.0
26	0.8	0.5	0.5	1.1
27	0.0	0.5	0.0	0.0
28	0.5	0.0	0.1	0.8
29	0.0	1.6	0.0	0.0
30	0.3	0.1	0.6	0.1
31	0.0	0.0	0.0	0.0
32	0.0	0.0	0.0	0.0
33	0.0	0.0	0.0	0.0
34	3.5	0.0	0.0	0.0
35	0.0	0.0	0.0	0.0
36	1.4	0.0	0.0	0.0
37	0.0	0.0	0.0	0.0
38	6.8	0.0	0.0	0.0
39	0.0	0.1	0.0	0.0
40	0.6	0.0	0.0	0.0
41	0.0	11.6	0.1	0.1
42	0.0	1.1	15.7	2.0

(continued)

Table 8.7 (continued)

	Max:			
Stand state	Tree species diversity	Tree size diversity	Basal area (m^2/ha)	Late seral forest (%)
43	0.0	5.5	0.1	0.1
44	0.0	0.5	0.6	1.5
45	0.0	12.3	0.1	0.0
46	0.2	0.7	23.0	0.1
47	0.0	7.4	0.0	0.0
48	0.0	0.6	0.6	0.1
49	0.0	0.0	0.0	0.0
50	1.5	0.0	0.0	0.1
51	0.0	0.0	0.0	0.0
52	0.9	0.0	0.0	0.0
53	0.0	0.0	0.0	0.0
54	14.5(17.5)	0.0	0.1	0.0
55	0.0	0.0	0.0	0.0
56	0.6	0.0	0.0	0.0
57	0.0	1.2	0.0	0.0
58	0.1	0.0	0.4	0.0
59	0.0	0.3	0.0	0.0
60	0.0	0.0	0.0	0.0
61	0.0	1.0	0.0	0.0
62	0.6	0.0	0.5	0.0
63	0.0	0.5	0.0	0.0
64	0.0	0.0	0.0	0.0
Total	100.0	100.0	100.0	100.0

In parentheses are the frequencies with undiscounted criteria that were significantly different

objective, with discounting and without discounting (in parentheses) over an infinite time horizon. Alternatively, this can be interpreted as the long-term probability that a particular stand would be in any given state, independently of its initial condition.

In maximizing tree species diversity, discounting raised the frequency of stand states #2 (000001) – high basal area only in large shade-tolerant trees, #6 (000101) – high basal area in small and large shade-tolerant trees, and #20 (010011) – high basal area in medium shade-intolerant and medium and large shade-tolerant trees. It lowered the frequency of state 54 (110101) – high basal area except for large shade-intolerant and medium shade-tolerant tree, by 3%, the largest change.

In maximizing tree size diversity, only stand state #25 (011000) – high basal area in medium and large shade-intolerant tree, would be less frequent with discounting, but by less than 1%. When maximizing basal area or LSF probability, discounting had no effect on the frequency distribution of stand states.

Table 8.8 Effects of maximizing discounted or undiscounted ecological criteria on expected forest characteristics after 10, 20, or 50 years, for a forest with the initial distribution of stands in Table 8.2

	With discounting			Without discounting		
Forest characteristics	10 years	20 years	50 years	10 years	20 years	50 years
	Max. species diversity					
Species diversity	1.29	1.31	1.33	1.28	1.31	1.33
Size diversity	2.79	2.81	2.82	2.80	2.81	2.82
Basal area (m^2/ha)	44.81	49.17	52.43	46.36	50.08	53.10
Late seral forest (%)	0.99	1.20	1.38	0.52	1.05	1.15
	Max. size diversity					
Species diversity	1.15	1.13	1.11	1.15	1.13	1.11
Size diversity	2.83	2.85	2.85	2.83	2.85	2.85
Basal area (m^2/ha)	49.46	52.33	53.14	49.47	52.26	53.07
Late seral forest (%)	0.11	0.09	0.06	0.09	0.06	0.04
	Max. basal area					
Species diversity	1.18	1.18	1.17	1.18	1.18	1.17
Size diversity	2.80	2.80	2.81	2.79	2.80	2.81
Basal area (m^2/ha)	53.86	57.26	62.65	53.26	56.48	61.94
Late seral forest (%)	9.97	12.54	17.73	10.15	12.41	17.46
	Max. late seral forest					
Species diversity	1.18	1.17	1.17	1.18	1.17	1.17
Size diversity	2.79	2.79	2.79	2.78	2.79	2.79
Basal area (m^2/ha)	55.31	60.73	63.01	55.42	60.64	62.89
Late seral forest (%)	40.35	53.67	60.45	39.28	52.95	60.21

3.4 Transient Effects of Discounting

Table 8.8 shows the effects of discounting ecological criteria on the expected characteristics of the forest after 10, 20, and 50 years, given the initial forest condition defined by the distribution of stand states in Table 8.3.

The probability of each stand state after 10, 20, and 50 years, following the policies that maximized the discounted or undiscounted criteria, was derived with Eq. 8.9.

Other things being equal, discounting had little influence on the expected forest characteristics. The main difference was that, with the initial conditions used here, maximizing discounted LSF probability led after 10 years to an expected LSF frequency that was 3% higher than maximizing undiscounted basal area. After 50 years the expected value of the forest characteristics was practically the same as with an infinite time horizon, shown in Table 8.6.

4 Summary and Conclusion

This chapter has presented general methods to develop management strategies that optimize ecological or economic criteria in continuous cover forestry. They rely on Markov decision process models, with transition probability matrices generated by bootstrap simulation of forest stand growth.

As an application, we dealt with the question of whether ecological costs and benefits should be discounted in the same way that financial costs and benefits are discounted in standard financial analysis. Without taking any position in this debate, this chapter investigated some of the purely ecological consequences of discounting ecological criteria.

To this end, Markov decision process models were developed, with infinite time horizons and discounted or undiscounted objective functions. The data were from Douglas-fir/western hemlock forests in the U.S. Pacific Northwest, with the assumption of continuous-cover forestry. The Markov probabilities of transition between stand states were obtained by bootstrap simulation.

The ecological criteria examined were: tree species diversity, tree size diversity, stand basal area, and probability of LSF characteristics. As the present value of the discounted criteria (but not the optimum policies) depend on the initial condition, the current distribution of stand states in the study area was used as initial condition. The discount rate for the ecological criteria was comparable to typical discount rates for financial returns.

In this context, we found that in maximizing expected tree species diversity over an infinite time horizon, 18 out of 64 stand states would call for different decisions with discounting. For tree size diversity, the decisions differed for five stand states. For basal area all but two states called for the same decision. For LSF frequency, discounting led to different decisions in 13 stand states.

However, when decisions differed with discounting, for only a few stand states were the criteria substantially different immediately after the decision. Thus, discounting would matter only for a few stand states, which in the study area had a low frequency within the overall forest landscape.

Given this initial forest landscape, the only criterion that differed substantially after a decade of applying a policy with discounting, compared to not discounting, was the expected frequency of LSF. But the difference disappeared in less than 50 years.

In the long-run (infinite time horizon), the initial conditions do not matter, and it was found that discounting or not discounting led to very similar values of expected tree species diversity, tree size diversity, stand basal area, and late seral forest.

In this as in many previous applications, the Markov decision process model has been found to be a very powerful means of investigating decision making under risk in forest management. It is a very appealing metaphor of the decision making process. At any point in time, the best decisions should, and can only be, based on the information available at that moment. The outcomes of various decisions, including doing nothing, are not known exactly and can only be thought of in terms of probabilities. High system complexity, such as the growth of a multi-species, uneven-aged forest which may be first modeled with simulation models, can then be reduced to Markov transition probability matrices for easy optimization leading to simplified decision-making rules.

Acknowledgements This work has been supported in part by financial support to Joseph Buongiorno by the USDA Forest Service Southern Forest Experiment Station.

Appendix

The matrix **G** and the vector **r** in the deterministic part of the growth model (8.2) have the following form (Liang et al. 2005):

$$\mathbf{G} = \begin{bmatrix} \mathbf{G}_1 & & & \\ & \mathbf{G}_2 & & \\ & & \mathbf{G}_3 & \\ & & & \mathbf{G}_4 \end{bmatrix}, \quad \mathbf{G}_i = \begin{bmatrix} a_{i1} & & & & \\ b_{i1} & a_{i2} & & & \\ & \ddots & \ddots & & \\ & & b_{i,17} & a_{i,18} & \\ & & & b_{i,18} & a_{i,19} \end{bmatrix}$$

$$\mathbf{r} = \begin{bmatrix} \mathbf{r}_1 \\ \mathbf{r}_2 \\ \mathbf{r}_3 \\ \mathbf{r}_4 \end{bmatrix}, \quad \mathbf{r}_i = \begin{bmatrix} r_i \\ 0 \\ \vdots \\ 0 \end{bmatrix}$$

where a_{ij} is the probability that a tree of species i and diameter class j stays alive and in the same diameter class between t and $t+1$. $i=1$ for Douglas fir, 2 for other shade tolerant species, 3 for western hemlock, and 4 for other shade-tolerant species. b_{ij} is the probability that a tree of species i and diameter class j stays alive and grows into diameter class $j+1$, and r_i is the number of trees of species group i recruited in the smallest diameter class between t and $t+1$, with a time period of 1 year. Recruitment is zero in the higher diameter classes. The b_{ij} probability is equal to the annual tree diameter growth, g_{ij}, divided by the width of the diameter class. Diameter growth is a function of tree diameter D_j (cm), stand basal area B (m^2/ha), site productivity Q (m^3/ha/year), tree species diversity H_s, and tree size diversity H_d.

$$g_{1j} = 0.7860 + 0.0124D_j - 0.0001D_j^2 - 0.0107B + 0.0267Q + 0.0658H_s - 0.2426H_d$$

$$g_{2j} = 0.6104 - 0.0038D_j + 0.0001D_j^2 - 0.0080B + 0.0170Q + 0.0707H_s - 0.0693H_d$$

$$g_{3j} = 0.9026 + 0.0148D_j - 0.0001D_j^2 - 0.0107B + 0.0061Q - 0.0250H_s - 0.1750H_d$$

$$g_{4j} = 0.5851 + 0.0081D_j - 0.00003D_j^2 - 0.008B + 0.0178Q + 0.1285H_s - 0.1441H_d$$

The expected recruitment of species i is represented by a Tobit model:

$$r_i = \Phi(\boldsymbol{\beta}_i \mathbf{x}_i/\sigma_i)\boldsymbol{\beta}_i \mathbf{x}_i + \sigma_i \phi(\boldsymbol{\beta}_i \mathbf{x}_i/\sigma_i)$$

with:

$$\boldsymbol{\beta}_1 \mathbf{x}_1 = -21.9317 - 1.2996B + 0.0971N_1 + 0.8007Q + 11.119H_s - 6.8020H_d$$
$$\boldsymbol{\beta}_2 \mathbf{x}_2 = -23.6333 - 0.8293B + 0.0975N_2 + 0.2032Q + 8.4122H_s - 5.9733H_d$$
$$\boldsymbol{\beta}_3 \mathbf{x}_3 = -30.8842 - 0.9359B + 0.0926N_3 + 0.6699Q + 14.693H_s - 9.8919H_d$$
$$\boldsymbol{\beta}_4 \mathbf{x}_4 = -34.5350 - 0.7512B + 0.0924N_4 + 0.7375Q + 8.0361H_s - 2.2701H_d$$

where N_i is the number of trees per hectare in species group i; Φ and ϕ are respectively the standard normal cumulative and density functions, and the standard deviations of the residuals are, $\sigma_1 = 23.5417$, $\sigma_2 = 22.4354$, $\sigma_3 = 27.3244$, $\sigma_4 = 23.0297$.

The probability of tree mortality per year, m_{ij}, is a species-dependent probit function of tree size and stand state:

$$m_1 = \frac{1}{10.5}\Phi(-2.1103 - 0.0356D_j + 0.0002D_j^2 + 0.0081B - 0.0200C + 0.0059H_s + 0.5110H_d)$$
$$m_2 = \frac{1}{10.5}\Phi(-1.4063 - 0.0204D_j + 0.0002D_j^2 + 0.0053B - 0.0147C + 0.0022H_s + 0.1411H_d)$$
$$m_3 = \frac{1}{10.5}\Phi(-3.1746 - 0.0416D_j + 0.0003D_j^2 + 0.0156B - 0.0230C + 0.3252H_s + 0.4192H_d)$$
$$m_4 = \frac{1}{10.5}\Phi(-1.5188 - 0.0093D_j - 0.0000D_j^2 - 0.0042B - 0.0303C - 0.2721H_s + 0.3528H_d)$$

The probability that a tree stays alive and in the same size class from t to $t + 1$ is, then:

$$a_{ij} = 1 - b_{ij} - m_{ij}$$

The expected single tree volume v_{ij} (m^3) is a species-dependent function of tree and stand characteristics:

$$v_{1j} = -0.6116 - 0.0119D_j + 0.0012D_j^2 + 0.0132B + 0.0293Q - 0.1249H_s - 0.0330H_d$$
$$v_{2j} = -0.5487 + 0.0038D_j + 0.0009D_j^2 + 0.0044B + 0.0173Q + 0.0136H_s + 0.0358H_d$$
$$v_{3j} = -0.4621 - 0.0141D_j + 0.0013D_j^2 + 0.0072B + 0.0104Q - 0.0659H_s + 0.0516H_d$$
$$v_{4j} = -0.1401 - 0.0170D_j + 0.0011D_j^2 + 0.0087B + 0.0146Q - 0.2471H_s + 0.0125H_d$$

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Chapter 9
Resource Assessment Techniques for Continuous Cover Forestry

Michael Köhl and Thomas Baldauf

1 Introduction

In the world's temperate and boreal regions roughly one fourth of the forest area is covered by mixed species stands (UN-ECE/FAO 2000). According to the MCPFE-Report on the State of Forests in Europe (Fig. 9.1), half of the European forest area is covered by mixed forests with two to three species (MCPFE 2007). Continuous cover forestry (CCF) is characterized by selective harvesting and natural regeneration, resulting in uneven-aged, multi-layer and multi-species forests. The resulting variability and diversity constitute particular conditions that need to be considered in forest planning and inventory.

Forest inventories in Europe have developed over more than 200 years and have strongly been influenced by the objectives of forest management. Historical situations such as the timber shortage at the end of the nineteenth and beginning of the twentieth century were promoting single species and even-aged stands, and hence forest management planning inventories were optimized for the assessment of those stand types. Nowadays sustainable forest management is seen "as a dynamic and evolving concept" that "aims to maintain and enhance the economic, social and environmental values of all types of forests, for the benefit of present and future generations" (UN 2008). The multi-functional role of forests is rated high by the public opinion, and maintaining the multiple services of forests has became a significant objective in forest management planning (Bachmann et al. 1998; MCPFE 1993, 2007; Gadow et al. 2007).

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6_9,

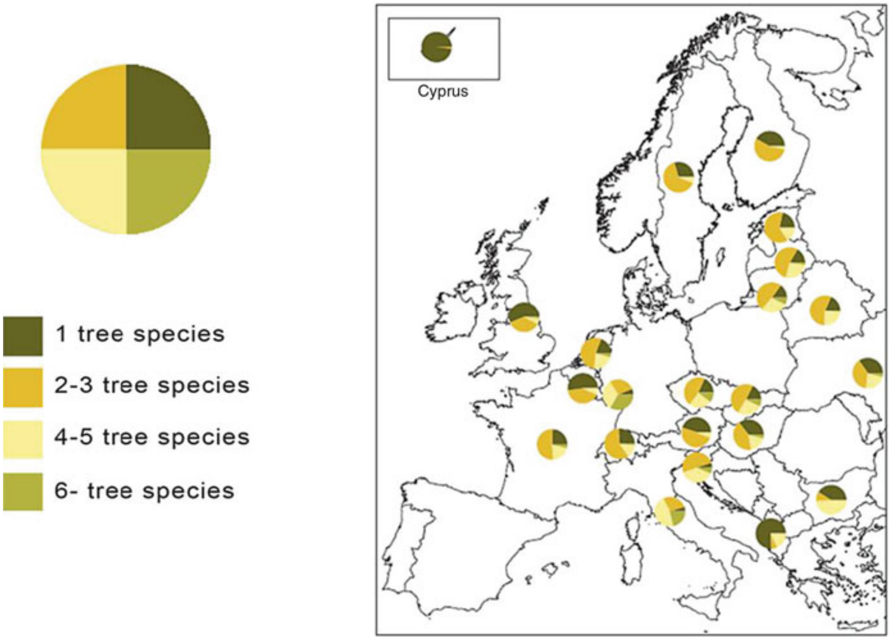

Fig. 9.1 Share (%) of the forest area by number of tree species for MCPFE countries, 2005 (from MCPFE 2007)

CCF presents rather different sampling populations, which are characterized by high variability and diversity of most population elements. The management of CCF systems requires information about complex forest structures in both space and time. Thus, the information needs must be satisfied but the heterogeneous population characteristics make it necessary to develop specific survey systems for continuous cover forests.

Monitoring CCF systems has attracted worldwide attention after the sixteenth session of the Conference of the Parties of the United Nations Framework Convention on Climate Change (UNFCCC), held in Cancun in 2010. This conference adopted REDD (Reducing Emissions from Deforestation and Forest Degradation in Developing Countries) as a programme to create financial values for carbon stored in forests. REDD + extends REDD beyond the mere conservation of forests to the sustainable management of forests. The implementation of REDD + requires "a much more straightforward measurement of forest carbon, one that can be easily applied by projects with minimal – or ideally, no – reference to outside quantification experts" (Munden Project 2011).

2 Information Needs and Specific Problems in Inventorying and Monitoring CCF

Traditional forest resource inventories focused on the assessment of the sustainability of wood production providing information on attributes such as basal area, timber volume, wood assortments, timber value and volume increment. Today, the information need is much wider and includes issues such as ecosystem services and functions (ESF), carbon pools in forests, biomass and fuelwood, and timber quality (Lund and Smith 1997; IPCC 2003; IPCC 2006; Köhl et al. 2006). Information on forest resources needs to address the seven thematic elements of sustainable forest management, which are (1) extent of forest resources; (2) forest biological diversity; (3) forest health and vitality; (4) productive functions of forest resources; (5) protective functions of forest resources; (6) socio-economic functions of forests; and (7) legal, policy and institutional framework (UN 2008). Most of these elements can be assessed by in-situ forest inventories.

Even-aged, single species forest ecosystems are "homogenized" by standard management practices such as planting, thinning and harvesting operations. From a survey point of view special assessment concepts are required to sample the heterogeneous populations of CCF; a fact that was already recognized at the end of the nineteenth century by Biolley and Gurnaud, who developed methods for monitoring selectively cut forests in France and Switzerland and made the estimation of change a key issue. The increment measured over the previous planning period was taken as the basis for deriving the allowable cut during the coming period. Gurnaud (1878), working in France, elaborated rules for the application of repeated measurements in estimating increment; this procedure is known as the method of control. These rules were first applied in Switzerland by Boilley in 1890 to the Couvet forest in the Jura mountains (Biolley 1921). Through his developments on control sampling, Schmid (1967) gave a decisive impetus to the application of sampling procedures in European forest management planning surveys (Schmid-Haas et al. 1984).

In CCF systems, which focus on the establishment and maintenance of near-natural forest management, information on diversity is a key-issue. Diversity is for categorical data what variance is for metric data but diversity is a wider concept since is also includes variation in continuous variables. According to Crow et al. (1994) diversity is made up of three components: (1) compositional diversity, (2) structural diversity and (3) functional diversity. These three components can be assessed on the genetic, species, ecosystem and landscape scales. Monitoring CCF systems mainly focuses on the assessment of the species and ecosystem scales. It is widely accepted that, due to the heterogeneity of CCF systems, information in terms of statistical point estimates (e.g. volume per hectare, number of species within a stand) is not sufficient, since those estimates do not provide information for the spatial domain and the strucutral diversity of forest stands.

The variance patterns of homogeneous and heterogeneous forests vary. Table 9.1 presents an example of the Swiss national forest inventory showing, among other

Table 9.1 Sampling estimates from the Swiss NFI (source: WSL 1999)

	Volume			Number of stems			Volume growth			Harvest		
Forest type	$\bar{X}$, m^3/ha	see %	CV	$\bar{X}$, n/ha	see %	CV	$\bar{X}$, m^3/ha/a	see %	CV	$\bar{X}$, m^3/ha/a	see %	CV
Even-aged high forest[a]	405.5	1	0.67	482	1	0.67	10.2	2	1.34	7.9	3	2.08
Uneven-aged high forest[b]	359.9	6	1.36	506	6	1.36	8.5	7	1.59	4.1	11	2.50
Selectively cut forests	289.1	8	1.33	436	8	1.33	6.8	10	1.66	3.7	16	2.66

see: standard error of estimate, *CV* coefficient of variation; see and CV describe the between-inventory-plot variation of concentric, fixed area (500 m^2/200 m^2) plots

[a] Trees forming the upper canopy layer have similar DBHs and belong to one development class

[b] Trees forming the upper canopy layer belong to more than one development class

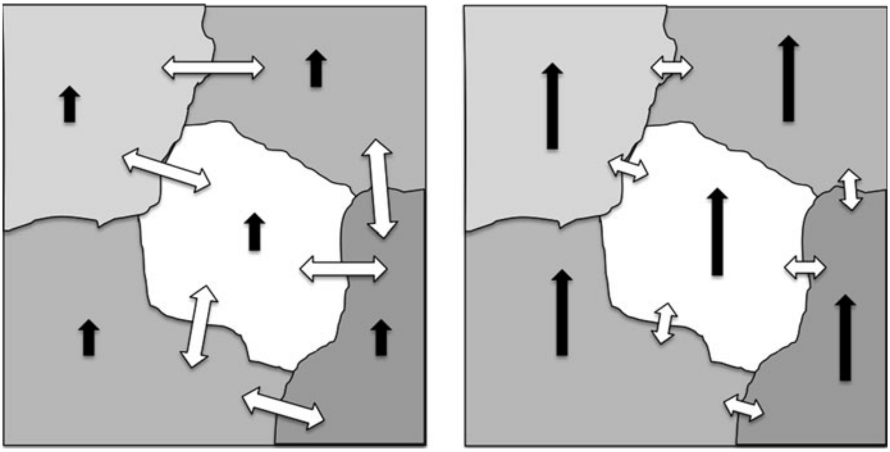

Fig. 9.2 Between-stand (*white arrows*) and within-stand (*black arrows*) variance of forests consisting of homogeneous (*left*) and heterogeneous (*right*) stands

things, the percent standard errors for volume, number of stems, volume growth and harvests for even-aged and uneven-aged high forests and selectively cut forests.

In CCF systems the within-stand variability is generally larger than the between-stand variability, while in even-aged, single species forests the between-stand varibility exceeds the within-stand variability (Fig. 9.2). Stand inventories require thus a higher sampling intensity in heterogeneous than in homogeneous forests. The situation is different in large area, extensive inventories, where CCF systems need larger plot sizes to capture the within-stand variability but fewer sample plots may suffice than in forests consisting of homogeneous stands. Gray (2003) and Král et al. (2010) describe the implications of the local variability in stand structure on sampling in mature costal Douglas fir and beech dominated forests.

Sampling and estimation procedures applied in the inventory and monitoring of CCF systems have to provide statistical key parameters that describe the population as a whole. In addition, information on the size distributions (e.g. diameter or height distributions) and spatial distribution of trees is required to describe the structural variability. Plot-based field surveys are an ideal tool to assess attributes on small scales with a high level of thematic sensitivity, but generally they do not allow one to produce maps. Remote sensing technologies are known to be an efficient tool to provide geo-referenced information. However, using the image products that are currently available at an operational level only a limited number of attributes can be derived. These attributes do not satisfy all information needs for CCF systems.

Several sampling designs have been described to combine remote sensing and field data, such as stratified sampling, multi-stage or multi-phase sampling (Bickford 1959; Bowden et al. 1979; Köhl et al. 2006). These combined surveys aim at increasing the cost-efficiency of survey systems by providing statistical key parameters with a reduced sampling error, but they do not facilitate mapping spatial information. Mapping of attributes which cannot be captured by remote sensing techniques requires thus alternative techniques such as the k-nearest neighbor method or geo-statistical methods (Corona 2010).

Forest surveys providing information on the productive function of forests utilize mainly data on interval or absolute scales. Statistical inference on these scales can easily be handled by estimation procedures using mean square approaches. The specific information needs associated with CCF systems render data describing complex forest structures on nominal and ordinal scales necessary and require non-parametric statistical approaches. Error estimation procedures are complex, as multiple confidence intervals have to be calculated simultaneously for classes and require the adjustment of the global level of significance (Goodman 1964, 1965; Quesenberry and Hurst 1964; Miller 1981).

3 Inventory and Monitoring Concepts for CCF Systems

The shift of information needs from productive functions to non-wood goods and services and ecological aspects of forests initiated the development of new forest resource assessment approaches, which facilitate the cost-effective assessment of CCF systems. New developments in remote sensing techniques and sensors are promising in terms of providing spatial information. The information needs concerning ecological aspects of forests are, however, often below the spatial resolution of most remote sensing tools. This holds especially true for CCF systems due to their small scale heterogeneity, resulting in a large amount of mixed pixels. Many attributes needed for monitoring CCF cannot be captured by remote sensing and have to be assessed in the field. It is beyond the scope of this chapter to provide a comprehensive survey of methods describing the structural diversity and spatial patterns of forests. Some methods to describe and assess the structure of CCF systems are described in Chap. 2 of this book. Instead a selection of approaches utilizing field assessments, remote sensing, combined survey systems and geo-statistical methods will be presented and discussed in the light of their potential for operational applications.

3.1 Terrestrial Surveys

The maintenance and enhancement of species diversity and other aspects of structural diversity became a major objective in sustainable forest management. Forest variability indices offer a tool for supporting decisions on management and silvicultural treatments. Neumann and Starlinger (2001) comment that “methods applied in assessing different types of diversity are as manifold as the ways of calculating measures of diversity”.

Most diversity indices are based on neighborhood methods (Pielou 1975). They can easily be applied in field surveys and provide measures that allow comparisons of different sites or the development over time. Examples of those indices are the Shannon Index, Simpson Index or Pielou Index (Magurran 1988), or the more recently published forest variability index (Das and Nautiyal 2004), the *Uniform Angle Index* (Gadow et al. 1998) and the *Tree Species Spatial Diversity* (Hui et al. 2011).

Where decisions are guided by forest variability indices, the understanding of those indices becomes crucial. Indices for species diversity and stand structure have been compared in several studies (Magnussen and Boyle 1995; Köhl and Zingg 1996; Gleichmar and Gerold 1998; Franc 1998; Neumann and Starlinger 2001). Köhl and Zingg (1996) found inconsistent estimates for changes of species diversity when indices are applied at successive occasions. Neumann and Starlinger (2001) compare species diversity and stand diversity indices in a wide range of ecological conditions in Austria. They found many correlations between the indices and arrive at the conclusion "that the method of calculation is not so important" given that a consistent set of elements of a forest is studied.

Beside the spatial structure of a forest ecosystem monitoring species diversity becomes an important objective in near-natural ecosystems. Random or systematic allocation of sampling units involves the risk of missing rare species. Thompson (1990) describes adaptive cluster sampling as an efficient method for the assessment of species with low abundance. Adaptive cluster sampling allocates sampling units in two steps: (1) an initially fixed number of sample plots is randomly or systematically distributed over the sampling area; (2) in each plot where rare species are found, the neighboring plots are measured. In any of these additional plots where the species of interest is found, another set of neighboring plots is established. The procedure is continued until no further plots are found where the species of interest occures. The procedure is an efficient approach to sample rare species (Roesch 1993), but operational applications of adaptive cluster sampling are few due to unknown a-priori sample sizes and cumbersome field assessments.

New vistas for the assessment of forest related attributes have been opened up by terrestrial laser scanning (TLS). TLS provides very detailed 3-D digital models of trees and canopies by very high laser pulse densities. Figure 9.3 shows a laser scanning device (FARO Laser Scanner Photon 120) mounted on a tripod and two point clouds each captured from a single viewpoint. TLS has been used to estimate leaf area (Huang and Pretzsch 2010), stem diameters (Maas et al. 2008), sweep and lean (Thies et al. 2004), tree value (Murphy et al. 2010), fibre quality (van Leeuwen et al. 2011), forest canopy structures (Parker et al., 2004) or fuelwood quantity (Loudermilk et al. 2007). The estimation of standing timber volume based on TLS data has been described by Hopkinson et al. (2004), who used allometric functions with stem diameter and tree height as input variables, and Maas et al. (2008), who used estimates of bole size and taper.

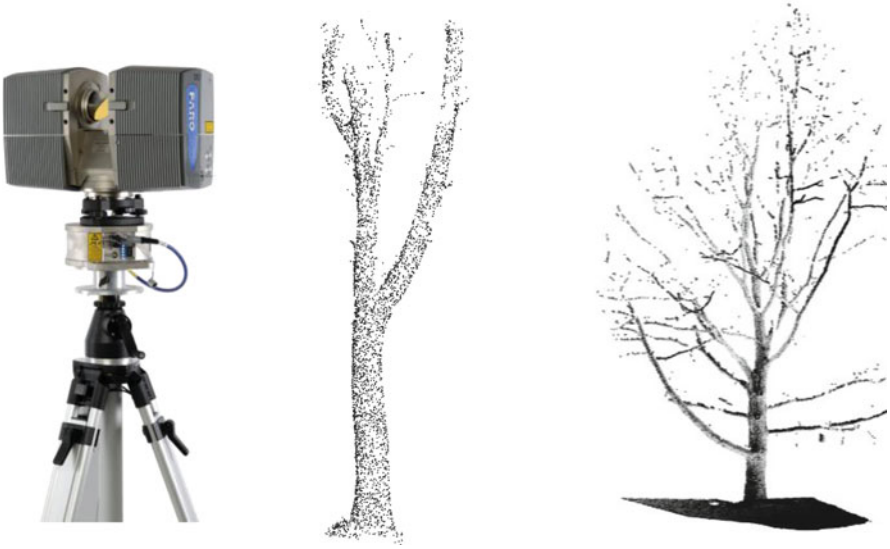

Fig. 9.3 Terrestrial laser scanner (FARO Laser Scanner Photon 120; Waitkus, vTI) and two examples of recorded point clouds, showing a tree trunk and a tree in total (Courtesy of Dr. Andre Iost, Institute for World Forestry, Hamburg)

3.2 Remote Sensing Techniques

In many publications, multispectral remote sensing data have been described as an ideal tool for extensive, large area forest resource assessments. A specific challenge for remote sensing applications is the small-scale heterogeneity, which characterizes CCF systems and may result in a large amount of mixed pixels. Bodmer (1988), Kellenberger (1996) and Fuller et al. (2003) elaborate on the effect of mixed pixels on classification accuracies and the feasibility to detect changes. Holmgren and Thuresson (1998) describe the limited suitability and utilization of satellite remote sensing for forest management planning purposes on the strategic, tactical and operational level. Recent sensor developments allow one to overcome many deficiencies of former remote sensing systems and open new perspectives for operational application.

Köhl and Lautner (2001) describe the use of airborne hyperspectral data of the Australian HyMap sensor in a test site located in the Ore Mountains, Germany. By utilizing hyperspectral remote sensing sensors image data were collected using many (from 30 to more than 200) narrow and adjacent spectral bands. Despite the fact that hyperspectral imagery is an extension of multispectral imagery the tools applied for image analysis and interpretation differ from the well known approaches in multispectral image analysis. A hyperspectral scene can be seen as an image with a spectrum of gray values, which are available for each pixel. For a given geographical area the data can be viewed as a cube, having two dimensions that

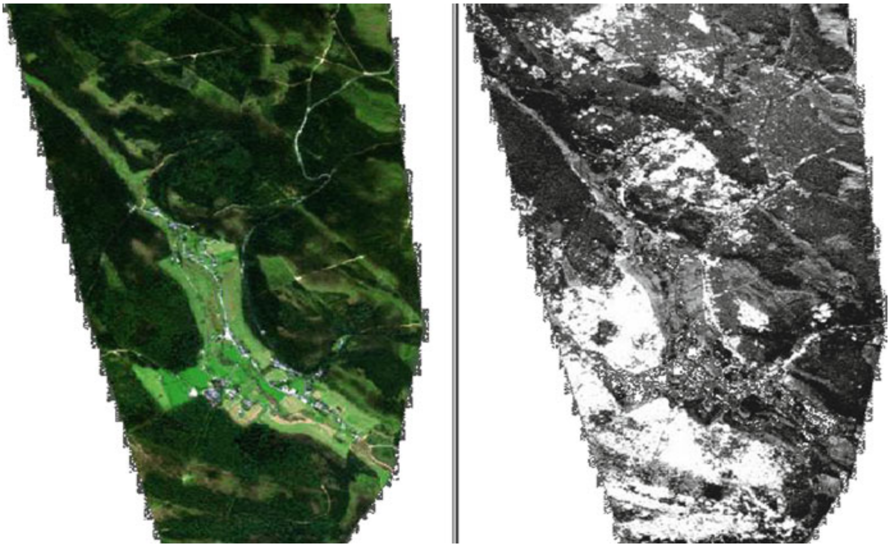

Fig. 9.4 HyMap ratio image (*left*) and proportion of 'broadleaved species'. The lighter the *grey* values the higher is the proportion of broadleaved species in the pixel (source: Häussler et al. 1999)

represent the spatial position and one that presents wavelengths. The image spectra can be compared with known spectra from field or laboratory experiments (so-called end-members). The data enable one to detect and map the spectral signatures of objects. An advantage of hyperspectral data is the possibility to handle mixed pixels by spectral unmixing or endmember analysis. Both analysis approaches foster the potential of hyperspectral data to provide detailed information on small areas. Figure 9.4 presents a ratio image of the original hyperspectral data and the endmember abundances for broadleaved species.

In the last decade the use of active remote sensing systems has gained increasing attention. Active remote sensing systems utilize instruments that send a pulse of energy to the earth's surface. The energy pulse is reflected from materials such as leaves, branches, stems and soil, and the reflected energy is received and recorded by the instrument. For forest inventory and forest management applications RADAR and LIDAR techniques have been described. LIDAR (Light Detection And Ranging), also called laser scanning or laser altimeter, is an active remote sensing technique that uses pulses of laser light while RADAR (Radio Detection And Ranging) uses electromagnetic waves. Forest attributes are calculated from these data mainly be means of regression estimates. Koch (2010) presents an optimistic review of current developments and future perspectives of laser scanning and synthetic aperture radar RADAR.

A variety of LIDAR systems has been used that successfully derived biomass, timber volume, basal area, or canopy height from LIDAR data (Garcia et al. 2010; Hyde et al. 2007; Lefsky et al. 2005; Lucas et al. 2008; Morsdorf et al. 2009;

St.-Onge et al. 2008; Zhao et al. 2009). The use of LIDAR has proven to be a valuable method for deriving forest characteristics relevant to forest management. Particularly for estimating tree and canopy height for both forest stands and individual trees a reliable suite of techniques exist (Magnussen et al. 1999; Persson et al. 2002; Vauhkonen 2010). Mehtätalo and Nyblom (2009) use a model based approach incorporating LIDAR data in combination with information on the shape of tree crowns to estimate individual tree stocks. Furthermore, Korpela et al. (2010) report high species classification accuracies for individual trees. Regarding CCF, Gaulton and Malthus (2010) present a promising approach to map canopy gaps using LIDAR data. Disadvantageous for operational applications of LIDAR remote sensing is the fact that most LIDAR systems studied in the context of forest assessments are not space-borne but flown at low altitudes, may render a frequent data acquisition from small areas costly, reducing the utility of airborne LIDAR systems (Parker et al. 2004).

In addition to airborne LIDAR systems, there are also operational space-borne RADAR systems. RADAR systems can be used at at any time of day and under nearly any weather condition, as the platforms carry their own active energy source and only few atmospheric constituents (i.e. clouds or other air-borne particles) interfere with detection at the wavelengths used for RADAR remote sensing. According to Luckman et al. (1998) these advantages explain the frequent application of radar-data in the tropics and mountainuous regions.

The wavelengths used with RADAR are about four to five orders of magnitude longer than those used in optical remote sensing. In contrast to optical sensors, which record reflections of the objects' surface, RADAR is sensitive to small twigs and leaves (X- and C-Band), and its long wavelength is only reflected by larger structures (L-Band). These conditions allow radar-data to provide information about structures beneath the forest canopy. The polarisation of RADAR signals in a vertical or a horizontal direction is an additional feature of these sensors that can be used for accentuating the backscatter from objects with particular orientations, such as tree boles in a recent clearcut (Kasischke et al. 1994).

Recent forest inventory experiments with profiling radar have shown to provide valuable data for stand-level forest inventories (Carleer and Wolff 2004; Hyyppä et al. 2000; Leckie et al. 2004). Baldauf and Köhl (2009) utilize TerraSar-X data to automatically detect and identify harvesting of individual trees in a vast tropical forest area. Such information is mandatory for monitoring forest degradation in natural forests or silvicultual treatments in managed forests.

3.3 Mapped Information

The ability to provide geo-referenced data is one of the outstanding advantages of remote sensing compared to field assessments. The information content of forest stand maps based on remote sensing data can be increased by combining field assessments and remote sensing imagery (Corona 2010). This approach is especially

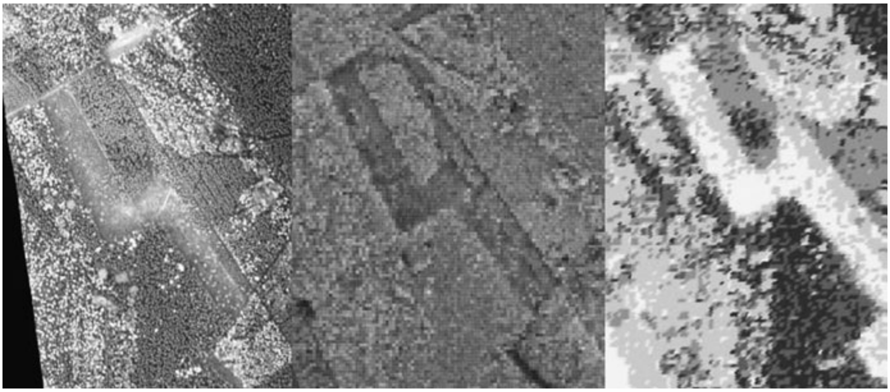

Fig. 9.5 HRSC-A image (*left*), the HyMap image (*middle*) and a map of the basal area per hectare derived by the kNN-method (*right*) (source: Häussler et al. 1999)

useful where maps are required that have to show the spatial pattern of attributes that are either not directly assessable in remote sensing imagery or that cannot be assessed with appropriate accuracy. Mapped output can be obtained by relating terrestrial samples to the spectral information of pixels via the "k nearest neighbor (kNN) –method". For the entire set of pixels without associated ground assessments the k nearest neighbors in the spectral image space are determined among those pixels, which coincide with the location of field samples. The values of attributes assessed on the ground at the location of the k nearest pixels are weighted by the distances in the spectral image space and assigned to the respective pixels for which no ground information is available. Pixel estimates are plotted to produce maps that show the spatial distribution of attributes assessed on the ground in the resolution of the remote sensing data.

The kNN-method was introduced into forestry by Kilkki and Päivinen (1987). Since then the method was further developed and has found numerous applications (Tomppo 1991; Nilsson 1997; Tomppo et al. 1999; Franco-Lopez et al. 2001; Tomppo and Halme 2004; Stümer and Köhl 2005; Finleya and McRoberts 2008). Kim and Tomppo (2006) and Magnussen et al. (2009) contributed to an advanced analysis of the statistical properties of kNN-estimates. While they refer to a model-based approach, Baffetta et al. (2009) investigate the statistical properties of kNN-estimators in a design-based framework.

Figure 9.5 presents the results of mapped basal area estimates calculated by means of the kNN-method using ground plots and hyperspectral images for a stand in the forest district Bärenfels located to the southeast of Dresden/Germany (Häussler et al. 1999).

Mapped information can be obtained by means of *geo-statistical methods* in situations where information from terrestrial surveys are available but there are no remote sensing data or other, geo-referenced wall-to-wall data,. These methods utilize the concept of regionalized variables and describe spatial dependencies

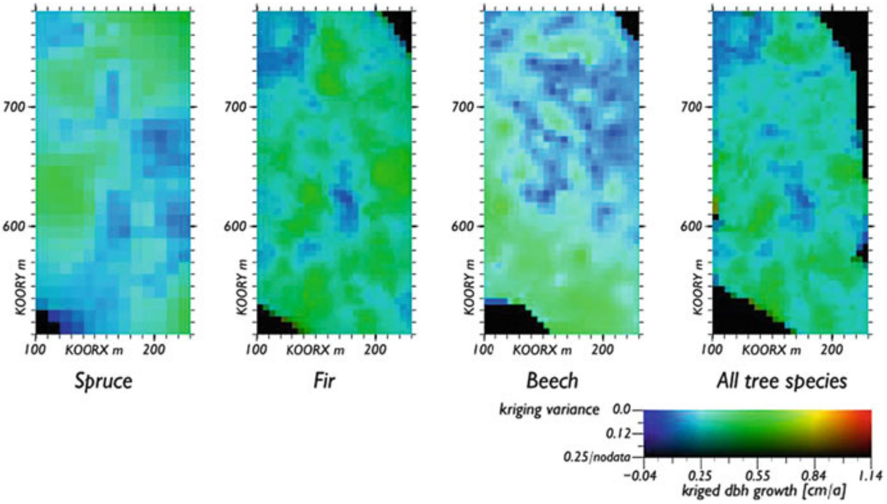

Fig. 9.6 Kriged values of dbh-growth in a selectively cut stand between 1976 and 1987 (Köhl et al. 2000)

between the realizations of random variables by means of variograms. A variogram is a graphical display of the variance of observed differences over the distance between the observations. If there are spatial dependencies the variance between the observed differences on two points normally increases with increasing distance until at a specific range a maximum value, i.e. *sill*, is reached. Empirical models can be fitted to variograms and used for deriving local estimates by means of *Kriging*. Kriging is a set of methods that have desirable qualities of statistical estimates. At its simplest form, Kriging is no more than a method of weighted averaging of the observed values of a property within a neighborhood. A detailed description of geo-statistical methods can be found in Journel and Huijbregts (1978) or Clark (1979). While geo-statistical methods have their roots in mining, one of the pioneers who applied those methods in forestry is Mandallaz (1993, 2000). He used geo-statistical methods in the context of two-phase inventory concepts for estimating stand related variables.

On the stand level, geo-statistical methods can be applied to detecting spatial patterns of tree attributes. Köhl et al. (2000) applied geo-statistical methods in a selectively cut stand located in the Bernese Pre-alps in Switzerland, which is stocked by uneven-aged spruce, fir and beech. They found that the growth rates of individual trees show a pronounced spatial pattern (Fig. 9.6).

Neural networks provide an alternative approach for gathering mapped information in situations where traditional statistical analysis techniques fail due to violations of the underlying constraints and assumptions. Neural networks are a field of research that deals with the "biological know-how" and mimics human thinking and reasoning. They are an ideal tool for the analysis of complex data sets due to the abilities to tolerate errors and uncertainty, and to accumulate and handle knowledge. One of the most advanced neural network applications are Kohonen's self-organizing maps (Kohonen 1982, 2001), which merge a high degree of biological plausibility with topology preserving features. Köhl and Jensen (1993) analyzed data of the Swiss forest health monitoring program by self organising maps; they could reproduce spatial patterns and utilize different spatial information layers for causal inference. Stümer et al. (2010) applied self organizing maps for estimating aboveground woody biomass and forest carbon stock in a test site located in Thuringia, Germany. Their approach utilized in-situ national forest inventory data and satellite remote sensing data (Landsat 7 ETM+) and provided maps showing a high-resolution spatial distribution of forest carbon stocks as well as statistical estimates of carbon stock and growing stock.

3.4 Change Estimation

Analysing resource trends over time is essential for sustainable forest management. Many inventory concepts have been developed and operationally implemented that allow for an efficient and statistically sound estimation of changes; Examples are CFI (Continuous Forest Inventory, Bickford 1959), SPR (Sampling with Partial Replacement, Ware and Cunia 1962), or updating methods (Hahn and Hansen 1983). In most operational applications the only measure of uncertainty included is sampling error, though other sources of uncertainty exist such as measurement error, classification error, model prediction errors, or frame errors (Lessler and Kalsbeek, 1992). Several studies focused on the impact of these additional sources of error on estimates (Gertner and Köhl 1992; McRoberts et al. 1994; Kangas 1998; Gal and Bella 1995; Westfall and Woodall 2007). Measurement varibility effects on the estimation of volume growth changes were studied by Canavan and Hann (2004) and Westfall and Patterson (2007). Westfall and Patterson (2007) found, by means of simulation studies, that the additional error due to measurement variability was small compared to sampling error, but bias introduced a substantial increase in total error.

Different change detection methods are applied to assess land-cover changes by digital remote sensing techniques. Where changes are calculated by direct comparisons of any thematic maps, classification accuracies have a strong implication on the reliability of change estimates. Fuller et al. (2003) conclude that the "measurement of small to medium scale changes over large areas requires levels of precision in mapping which are near impossible to achieve with satellite image classification alone".

More advanced digital change detection techniques have been presented that utilize multivariate statistical approaches. With respect to the algorithms applied they can be grouped into the following categories: classification (e.g. post-classification comparison), algebra (e.g. rationing, regression), change vector analysis, transformation (e.g. multivariate alteration detection), and hybrid methods (Nori et al. 2008). Nielsen et al. (1998) and Canty (2010) present Multivariate Alteration

Detection (MAD) transformation that is based on the canonical correlation analysis. MAD uses two sets of multivariate observations (e.g. digital image data assessed at two different points in time) and transforms them into a difference between linear combinations of the original variables. Occasionally, MAD has been applied to assess forest area changes (Nori et al. 2008), but deserves due to its superior features for land-cover change detection more attention in the future (GOFC-GOLD 2010).

4 Conclusions

Maintaining the traditional multi-functional role of forests is the leading principle in sustainable forest management. Due to a transition in silvicultural methods and increasing claims of the public to the recreational and protective functions of forests there is a shift of information required for forest management planning. Statistical key-parameters such as mean values, totals, proportions or ratios may be sufficient for managing even-aged single species stands, which have a homogeneous spatial distribution for most attributes. CCF aims to maintain near-natural forest ecosystems favoring uneven-aged mixed species populations, natural regeneration and natural selection processes. The resulting forests show high diversity and variability, and heterogeneous spatial patterns. Thus, simple first order statistics are no longer sufficient for management planning in the context of CCF, but have to be supplemented by distributions, sometimes also by mapped information.

For the inventory and monitoring of CCF, the augmented information needs result in a couple of consequences: (1) the set of attributes to be assessed increases, (2) data on nominal or ordinal scale are gaining importance and require non-parametric estimation procedures and (3) handling and analyzing geo-referenced data for mapped output becomes a major issue.

During the transition period from homogeneous, even-aged, single species forests to heterogeneous CCF systems there is a significant need for information on changes. Growth and yield models developed for single-species, even-aged stands are associated with an unknown prediction error when applied to mixed species forests, as they may not take into account, in a proper way, the competition between trees and differences in spatial distributions. Monitoring not only timber growth but structural changes becomes is also a a great challenge and a major issue of forest resource assessments at successive occasions.

Appropriate assessment methods for monitoring CCF systems have to be developed that leave the beaten track of ordinary least squares estimates by utilizing new tools in the areas of geo-statistics, non-parametric or multivariate statistic, GIS or remote sensing, to provide spatially explicit data.

Acknowledgements We want to express our thanks to Dr. André Iost, Dr. Bernhard Kenter, Konstantin Olschofsky, and Daniel Plugge, all Institute for World Forestry, Hamburg, Dr. Charles T. Scott, USDA Forest Service, Northeastern Research Station, Newtown Square, PA, USA, and to Prof. Dr. Steffen Kuntz, Infoterra GmbH, Friedrichshafen, Germany, for helpful input and comments.

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T. Pukkala and K. von Gadow (eds.), *Continuous Cover Forestry*, Managing Forest Ecosystems 23, DOI 10.1007/978-94-007-2202-6,