Tropical Forestry

Sven Günter
Michael Weber
Bernd Stimm
Reinhard Mosandl *Editors*

Silviculture in the Tropics

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Tropical Forestry

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Sven Günter • Michael Weber • Bernd Stimm •
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Editors

Silviculture in the Tropics

With 49 Figures and 59 Tables

Springer

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Preface

In 1998, Dawkins and Philip published their important book *Tropical Moist Forest Silviculture and Management. A History of Success and Failure*. In that book they stated: “The main message from the past is that natural forest management can be done; conservation and production are not incompatible; in fact, in some of the circumstances the only way to conservation will be through the management of the production of goods and services from the forest.”

Convinced of the trueness of this statement silviculturists working in all tropical areas of the world have contributed to this book. It reflects the efforts of 46 authors to specify the state of the art in tropical silviculture. The book aims at combining two complementary aspects of tropical silviculture: first we would like to emphasize its role as integrative scientific discipline linking ecological and socioeconomic dimensions, and second we call attention to its role as an instrument to satisfy the manifold demands of the multiple forest users, from livelihood over timber production to provision of environmental services. Therefore, this book contains contributions from scientists, as well as from representatives of international institutions. While the first parts of the book focus on forest users and new aspects in silviculture, the latter parts are structured according to forest types. For maintenance of high scientific standards, all articles are peer reviewed. All ten parts of the book (with the exception of introduction and conclusions) are subdivided into one review chapter giving an overview and introduction into the topic, followed by case studies from Africa, Asia, and Latin America highlighting special aspects.

We hope this book will be useful to scientists, practitioners, and students working in tropical forestry, especially to those managing tropical forests for the provision of goods and environmental services in supporting them to do this in a wise, adaptive, and sustainable manner and under concern of human needs and social requirements.

February 2011 Freising

The Editors

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Abbreviations

a.s.l.	Above sea level
ATSC	Australian Tree Seed Centre
BMP	Best management practice
CAI	Current annual increment
CBD	Convention on Biological Diversity
CDM	Clean Development Mechanism
CF	Community Forestry
CFM	Collaborative Forest Management
CFUG	Community Forestry User Group
CIFOR	Center for International Forestry Research
CoP	Conference of Parties (Organized by United Nations)
CSIRO FFP	CSIRO Forestry and Forest Products
DBH	Diameter at breast height
DFCC	District Forest Coordination Committee
DFRS	Department of Forest Research and Survey
dsm^{-1}	Deci siemens per meter
FAO	Food and Agriculture Organization
FCT	Future crop tree
GBH	Girth at breast height
GFRA	Global Forest Resource Assessment
GHG	Green house gas
GM	Genetic modification
ha	Hectare
IAD	Institutional analysis and development
IDH	Intermediate disturbance hypothesis
ILO	International Labor Organization
INGO	International Non Governmental Organization
IPM	Integrated pest management
ITTO	International Timber Trade Organization
IUCN	International Union for Nature Conservation

LAI	Leaf area index
LLP	Long-lived pioneer
m	Meter
MAHI	Mean annual height increment
MAI	Mean annual increment
MMAI	Maximum mean annual increment
MCD	Minimum cutting diameter
MDF	Mixed dipterocarp forest
MED	Minimum exploitable diameter
MFD	Minimum felling diameter
MOFSC	Ministry of Forest and Soil Conservation
NGO	Non Governmental Organization
NPLD	Non-pioneer light demanders
NPP	Net primary production
NPV	Net present value
NTFP	Non-timber forest product
PPFD	Photosynthetic photon flux density
PST	Partial shade-tolerant
REDD	United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries
RGR	Relative growth rate
RIL	Reduced-impact logging
R-PIN	Readiness Plan Idea Note
SFM	Sustainable forest management
SMFE	Small and medium-sized forest enterprises
TAS	Traditional agroforestry system
TCBFM	Traditional community based forest management
TFD	The Forest Dialogue
TST	Total shade-tolerant
UNEP	United Nations Environment Programme
UNU-WIDER	United Nations University-World Institute for Development Economics Research
y	Year

Part I
Introduction

Chapter 1
Introduction to Silviculture in the Tropics

Sven Günter

Abstract This chapter provides an introduction to the book *Silviculture in the Tropics*. The development of the scientific discipline of silviculture is closely related to the evolvement of the term "sustainability" from stable provision of wood in the eighteenth century to the provision of environmental services and non-timber forest products nowadays. Silviculture as a scientific discipline aims at mediating between natural sciences and societal disciplines. Several definitions of silviculture in this context are presented in the text. Many principles of silviculture in temperate ecosystems are generally valid in the tropics too. However, one main difference from temperate silviculture is the exorbitant biodiversity of most of the tropical forest ecosystems. This makes silvicultural planning and interventions much more complicated, on the one hand, and compatibility with the aims of conservation of biodiversity much more important, on the other hand. Since many people in the tropics in contrast to those in most countries in temperate ecosystems depend on forests for their subsistence and livelihood, silvicultural goals should match the aims of rural development and reducing poverty. The second part of this chapter provides an overview of the chapters in the book, which is subdivided into eight main parts, each consisting of an introductory overview chapter, accompanied by some case studies from different tropical continents. Parts II and III set the stage for the following more specific parts. Part II deals with the different demands of users towards forests, whereas Part III deals with the multiple new aspects in modern forestry with strong impact on silviculture, from conservation of biodiversity to use of non-timber forest products to modeling approaches in science and practice. Parts IV–VI deal with silviculture in natural humid forests, dry forests and special ecosystems such as mangroves and mountains. Parts VII–IX discuss forests with stronger human interventions: secondary forests and planted forests for productive purposes and for restoration. The book ends with a final, concluding chapter.

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1.1 Tropical Forests and Changing Requirements

Almost half of the world's forests are located in tropical countries (FRA 2010). Latin America[1] has the largest tropical forest area (810 million hectares), followed by Africa[2] (627 million hectares) and Asia[3] (489 million hectares). However, the extent of annual deforestation is still enormous, mainly owing to conversion of forests into agricultural land, especially in the tropics. More than seven million hectares of forests are lost every year. An alarming loss of 2.2 million hectares per year is attributed to Brazil and 0.7 million hectares are lost in Indonesia each year. Also, the relative rates of deforestation are highest in the tropics: 9.7% in the Comoros, 5.8% in Togo and 4.0% in Nigeria. The increasing demands on land use, especially for food and energy, will probably aggravate this dilemma in the future. Although direct employment of people in the forestry sector in relation to the whole labor force of a country is usually below 1% (FRA 2010), the dependence of people on forests for their livelihood is much higher. If deforestation in the tropics continues, an increasing number of poor people will lose even their low existence base, and the global human population will have to face the negative effects in relation to climate, biodiversity and other environmental services. Of course, silviculture alone cannot stop deforestation, but it can contribute to more sustainable management of the renewable forest resources and hence can mitigate negative effects of deforestation and climate change.

The requirements of societies towards forests, forest services and products differ in time and between cultures. Although early in human history the main aims were collecting non-timber forest products (NTFPs) and hunting, in recent centuries the main focus was on harvesting timber. Especially since the Brundtland Report of the World Commission on Environment and Development (1987), the requirements and aims have changed to meeting social objectives for values other than timber (Benskin and Bedford 1995). Today, new challenges arise for forest management and silviculture by the need to fulfill both the demand for products such as NTFPs and timber on a local or regional scale and the needs for environmental services such as conservation of biodiversity and mitigation of climate change on a global scale.

1.2 Definitions and Concepts of Silviculture

The two main pillars of silviculture are reflected by the Latin words *silva* (meaning "forest") and *cultura* (meaning "cultivation"). Silviculture in this very basic sense hence describes cultivation of forests, without implying any qualitative criteria or

[1]Without Argentina and Chile.

[2]Without Mediterranean countries.

[3]Only South Asia, Southeast Asia and Oceania.

thresholds for best management practices. Thus, in its very literal meaning, silviculture is rooted in social sciences as well as in natural sciences. However, it is notable that "culture" is sometimes defined as the opposite of "nature" (Liebsch 2004). The composition of two apparently antipodal terms indicates the overarching objective of silviculture of balancing culture and nature. Silviculture in its literal meaning consequently aims at mitigating and balancing the objectives of conservation of forest ecosystems and functions and anthropogenic uses. Dawkins and Philip (1998) stated that the basis for silvicultural objectives is defined by social requirements within the limits of what is technically possible.

Managing forests without considering the impacts of interventions on an ecosystem, as a kind of "one-way-management," is as old as humankind. The increasing human population and pressure exceeding the regeneration capacity of forest ecosystems necessarily causes destruction of wilderness and loss of biodiversity, e.g., by overhunting of animals and intensification of agriculture several thousand years ago (Eastwood et al. 2007; Horan et al. 2003) or exploitation of high-timber-value species such as mahogany in Central America starting some 100 years ago (Lamb 1966). With forest goods and services becoming scarce (not exhausted), it is essential to look for a long-term balance between the needs of humans and conservation of nature. The beginning scarcity of natural resources can be considered as an alarming signal from nature for humans to modify and adapt their silvicultural activities. Increasing awareness that forests are limited natural resources is hence the basis for silviculture in the context of sustainable forest management. A proactive balancing between apparently detrimental objectives requires a sound analysis of ecological and economic processes and optimization of trade-offs and interactions in order to avoid exploitation of natural resources. Thus, measuring and quantifying the signals of scarcity are important instruments for silviculture.

Price is frequently considered to be a good indicator of scarcity or shareholders' perception of scarcity. "The conventional economic approach seeks to maximize the present value of a stream of aggregate benefits less costs" (Toman and Ashton 1996). The difficult task of including public goods and services in the microeconomy is one example of "imperfect markets" (Stiglitz and Walsh 2010). This term indicates the possible limitations of purely market driven forestry to achieve sustainability. Additionally, the current revenues are based on decisions and silvicultural operations carried out decades and sometimes even hundreds of years ago. Silviculture today, in turn, has to set the course for economically profitable and ecologically sustainable management in the future. A further problem of markets is strong time preferences of forest users, since future yields are less reflected in current prices than yields today. Discounting is a common instrument to overcome this problem, but it is questionable if all economic and ecological risks can really be represented correctly by discount rates. Unfortunately, it is still common practice to manage tropical forests without sound knowledge of sustainable yields or significant impacts of human interventions on ecosystem functions and services. Forest management under uncertainty and without considering risks, on the one hand, and lack of ecological knowledge (especially regarding yields and long-term damage to the remnant stand), on the other hand, may hardly set the stage for sustainability in the future (Knoke 2010). Besides economic indicators, globally comparable ecological indicators for disturbance of ecosystem functions and for defining thresholds of "responsible management" are required (Raison et al. 2001). Several international organizations are working on transparent and reproducible lists of indicators of sustainability. However, bridging this gap of missing knowledge may be the major challenge for foresters and economists in the future.

Despite economic and ecological dimensions, social aspects play a key role in sustainable forestry too (Weber-Blaschke et al. 2005). Dawkins and Philip (1998) gave an illustrative example: "what happens to a swiss forester if his precautions against protective functions of alpine forests such as avalanches, fail and result in a loss of human life?" The answer of the forester is: "I have to walk behind the coffins to the graveyard with the villagers." In this example, the protective functions are less driven by economic processes than by social control. On a global scale this may raise the question of who will take responsibility for global climate change or loss of biodiversity due to human pressure. Although social control may lead to effective management of protective functions on a local scale, the global mechanisms of social control for achieving "responsible management" are still unsolved (Toman and Ashton 1996).

In the context of sustainable forest management, several definitions of silviculture have been proposed. In the following I will highlight just a few of them:

- Silviculture "is sometimes called the growing side of the forestry business: the cultivation of woods or forests; the growing and tending of trees as a department of forestry (in Oxford English Dictionary 2nd edition, 1989)" (Dawkins and Philips 1998).
- The art of producing and tending a forest; the application of knowledge of silvics in the treatment of a forest; or the theory and practice of controlling forest establishment, composition and structure, and growth (Smith et al. 1996).

Although these definitions mainly refer to application of activities in the forest, the following ones include aims and objectives. They integrate the requirements of society towards the forests. These more comprehensive definitions explicitly imply ecosystem functions and products and services far beyond timber production. They create a link to sustainable forest management. In the following definitions, the scientific character of silviculture is highlighted instead of the rather descriptive and artistic point of view above. They are generally valid for the tropics too.

- In his book *Silviculture in the Tropics*, Lamprecht (1986) cites Leibundgut: "Today, silviculture considers the forest as ecosystem. It aims at regulating all life processes in an ecologically stable forest and organizing its establishment and regeneration in a way that all needs related to forests are fulfilled best possible and sustainably, i.e. in a permanent and rational manner."
- Silviculture investigates the consequences of decisions about the treatment of forest ecosystems in order to fulfill present and future human needs (Knoke 2010).

- Silviculture is designed to create and maintain the kind of forest that will best fulfill the objectives of the owner and the governing society. The production of timber, though the most common objective, is neither the only nor necessarily the dominant one (Smith et al. 1996).

1.3 Main Differences Between Silviculture in the Tropics and Temperate Zones

There are two main aspects from both natural and social dimensions with a strong impact on silvics and management in the tropics that have to be stressed with more emphasis:

- The high number of tree species complicates botanical identification in the field. Mostly, fertile samples are required for correct identification. Additionally, higher tree diversity is usually accompanied by a lower number of harvestable individuals per hectare, with some exceptions (e.g. in Southeast Asia, where dipterocarps with high timber value dominate the upper canopy in many cases, peat swamp forests, mangroves). Further, biodiversity is recognized as a global value, but up to now does not provide economic benefits to tropical land owners. Thus, balancing conservation and economic interest will be of higher importance than in temperate forests.
- Many countries in the tropics are developing countries or countries in transition. In addition to all forest functions in temperate zones, tropical forests have to fulfill subsistence needs in many cases and suffer from higher human pressure. They are frequently converted into alternative land-use forms which provide either food or cash crops with higher economic returns, at least from a non-sustainable and short-term point of view. Further, many governments have poor or almost no control over the forests and cannot balance conflicting land-use interests properly. Users' needs, of course, are different from and often much more dynamic than those of users in temperate ecosystems. According to the above-mentioned definitions, silviculture in the tropics therefore requires much more careful integration of the social and political dimensions. Decisions and treatments which have long-term effects on the ecosystem should consider that the users' needs may change rapidly.

1.4 Focus and Structure of the Book

The book addresses scientists as well as professionals from the fields of tropical forestry, conservation and landscape management. Each part starts with a general overview chapter as an introduction to the topic and which summarizes the state of the art. Case studies from different tropical regions in each part give more detailed insight into special regional, technical or social aspects. All chapters have been peer-reviewed.

To provide a broad overview of the different concepts of tropical silviculture, the book is designed as a participative coproduction by authors from all regions of the world, i.e. authors from Africa, the Americas, Asia, Europe and Oceania. It is scientifically based, but is addressed at application. Therefore, authors from non-scientific institutions which aim at finding practical solutions for balancing human interests with conservation such as the Food and Agriculture Organization of the United Nations (FAO), Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ), Deutscher Entwicklungsdienst (DED) and Stichting Nederlandse Vrijwilligers (SNV) contributed several articles.

Forest users are given greater attention in this book than in other books related to tropical silviculture. This is attributed to the consideration that the requirements of humans finally set the frame and define the aims for management and silviculture. In Part II (with a review in Chap. 2 by Kotru and Sharma) we include three case studies representing different points of view covering "classical" forestry management by concessions, community forestry and the latest developments in access and benefit sharing.

Part III (with a review in Chap. 6 by Weber) introduces current global aspects which may have a strong influence on silviculture in the future. On the one hand, short-rotation plantations are emerging around the world, aiming at the highest possible yields in the shortest possible intervals (see Chap. 9 by Onyekwelu). On the other hand, there is increasing demand for buffer zone management mitigating the increasing human pressure on protected areas and conservation of biodiversity and genetic resources (see Chap. 7 by Putz and Chap. 8 by Finkeldey), for example by including NTFPs in silvicultural practices (see Chap. 10 by Vantomme). The demands for both maximization of timber and conservation do not necessarily fit together and require appropriate management and decision-supporting tools. Growth and yield predictions are essential prerequisites for sustainable management. Thus, it is surprising that wood production and, even worse, the financial consequences of forest management and silviculture are poorly assessed in many areas of the tropics (see Chap. 11 by Knoke and Huth).

After forest users and new aspects in tropical silviculture have been introduced, the book is subdivided into parts on (semi-)natural forests and planted forests according to the FAO's classification (FAO 2006). The (semi-)natural forests are discussed in four book parts. Whereas Part IV (with a review in Chap. 12 by Ashton and Hall) refers to humid forests with emphasis on timber-rich forests of Southeast Asia, Part V (with a review in Chap. 16 by Fredericksen) covers the drier ecosystems with a more pronounced dry season or less rainfall, or both. Here, the main focus is on the neotropics. However, the chapters on both wet and dry forests cover the whole of the tropics and are accompanied by case studies from all tropical regions. Changes in dry season length and precipitation among forest types and regions are continuous and any classification will cause abrupt interruptions in some cases. The authors and the editors roughly followed Holdridge's classifications, with exception of the Meliaceae. The latter are placed in Part IV, despite their distribution ranges covering both wet and dry forest formations. However, their silviculture tends to be more typical of humid forests. Additionally, the silviculture of the major tropical tree families Dipterocarpaceae and Meliaceae with highly appreciated timber can be compared directly in one book part.

Because it is not possible to cover all forest ecosystems in the world in this book, Part VI (with a review in Chap. 20 by Günter) considers two climatically azonal forest formations, mangroves and montane forests, both of them at opposite positions along the altitudinal gradient. Exemplarily to other forest ecosystems at ecological margins, the role of payments for environmental services is discussed in this part. Owing to increasing human pressure, secondary forests are an expanding forest formation worldwide. Therefore, special reference is given to this often neglected topic in Part VII (with a review in Chap. 23 by Akindele and Onyekwelu).

Parts VIII and IX on planted forests cover forest types with a stronger human component. The two parts are dedicated to two different aims: Part VIII (with a review in Chap. 27 by Onyekwelu et al.) provides an overview of the broad field of plantation forestry in terms of wood production, and Part IX (with a review in Chap. 30 by Weber et al.) refers mainly to planting for restoration purposes and rehabilitation of ecosystem functions. However, these two parts have smooth transitions: protective functions do not exclude wood production and wood production could be compatible with ecosystem functions.

Final conclusions are given in Chap. 34. On the basis of the contributions to this book, Günter et al. extracted five trends for modern tropical silviculture.

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Part II
Forest Users: Past, Present, Future

Chapter 2
Review
Forest Users: Past, Present, Future

Rajan Kotru and Sudhirender Sharma

Abstract From a frugal use of tropical forests by primitive indigenous communities a few centuries ago, their utility has grown to global significance with a wide array of goods and services sought by the world community. This evolutionary role of tropical forests, however, has come at a cost as these forests are under severe threat owing to persistent overuse. It is widely accepted that with the dawn of colonialism across the tropical belt, extraction of forest products for industrial use and infrastructure became intensive, and energy and livelihood demands of the growing population of forest-dependent communities soared. The resultant deforestation and forest degradation under state ownership was countered by handing over management to local communities. It has clearly emerged that tenure security is the key for getting communities committed to judicious management in the long run. With the increased demand for sustainable yield of goods and services, consultative processes amongst a range of stakeholders became important to minimise conflicts and influence policy and management in practise. Learning experience shows that for sustainable management of tropical forests state and community partnership is unequivocal, social inclusion and governance issues must be resolved, value addition of forest products must add to the local economy and employment, technical management must be simplified and the climate agenda must be addressed. Moreover, since sustainable forest management can no longer be seen in isolation from the politics and practise of other sectors regarding forests, it is inevitable that institutional capacities, learning and knowledge networks, participatory monitoring and advocacy forums are consolidated across vertical and horizontal levels of governance and relevant sectors.

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2.1 Context

2.1.1 Forest Use in the Tropics from a Historical Perspective

From a mere frugal use of intact tropical forest ecosystems by insignificant forest dweller populations barely a few centuries ago, these forests in the twenty-first century have evolved to a global natural asset for a broader delivery of goods and services sought by a range of stakeholders (e.g. subsistence farmer, wood industry, conservationists, traders). Since CoP 13 (e.g. the Bali road map in 2007), the status of "free for all" of tropical forests owing to their immense growth and biodiversity potential has found defining attention in the emerging challenge of countering greenhouse gas emissions as the root cause of climate change. Following it up, CoP 15 in Copenhagen (2009) highlighted the role of forests in sequestering greenhouse gases and made cost-effective and efficient forest management a prime agenda. However, this radical shift in the thinking on and use of tropical forests has come at a significant loss. The Global Forest Resources Assessment 2005 (FAO 2006) of the Food and Agriculture Organisation of the United Nations (FAO) mentions the net loss in forest area at the global level during the 1990s was an estimated 94 million hectares – an area larger than Venezuela and equivalent to 2.4% of the world's total forests. In another estimation for tropical forests, natural dense broad-leaved forest covers 1,260 million hectares, or 9% of Earth's total land area (Barbier and Burgess 2001). Despite increased awareness of the importance of these forests, deforestation rates have not slowed.[1] Analysis of figures from the FAO shows that tropical deforestation rates increased by 8.5% from 2000 to 2005 when compared with the 1990s, whereas loss of primary forests may have expanded by 25% over the same period. The rate of primary forest loss has doubled in Nigeria and Vietnam since the 1990s, whereas Peru's rate has tripled.

Although extensive, the world's forests have shrunk by some 40% since agriculture began 11,000 years ago. Three quarters of this loss occurred in the last two centuries as land was cleared to make way for farms and to meet the demand for wood. As a classic example of forest decimation, Haiti, with a forest cover estimated at 3% of all land area, has experienced severe degradation of its natural resources and a significant change in its land cover. Although deforestation in Haiti is obviously multifaceted, one issue emerges from empirical analysis in explaining deforestation: land tenure. A study was made on the causes of deforestation in Haiti, particularly in the Forêt des Pins Reserve, using the annual average area of cleared forest per household as the dependent variable. Data were collected with the use of a survey instrument administered to 243 farm households in 15 villages inside the

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[1]The Draft Global Forest Resource Assessment 2010 of the FAO reports that there is slowdown of the deforestation rate. However, South America and Africa are having a higher net annual loss of forests (2000–2010) and Asia, especially owing to afforestation in China, India, Vietnam and Indonesia, is showing a net gain. Between 2000 and 2005, Africa and South America experienced the largest net forest losses (21.87 and 19.01 million hectares, respectively).

reserve. Tobit regression results revealed that household size, education of the head of the household, land tenure regime and farm labour are important factors affecting land clearing.

Given the above account and accelerated changes in tropical forests occurring since the early 1960s, largely attributed to population and economic growth, the mechanisation of extraction techniques, and increasing means for transportation (ITTO 2006), the tropical forest ecosystems are rated as the most threatened forest ecosystems in the world (Millennium Ecosystem Assessment 2005). The grave implications of such devastation can be seen from the fact that these forests shelter nearly half of Earth's biodiversity, capture carbon, protect water, food and soil resources, and provide timber and other forest products for consumption and commercial use (FAO 1995, 2001). Subsequently, this has serious implications for an estimated 350 million indigenous and tribal peoples at least partly dependent on forests, including some 60 million who are substantially dependent on forests for their subsistence and livelihoods. These forests are particularly located in developing tropical countries and therefore are very important to the poor and women, who shoulder much of the burden of hauling wood and collecting and marketing forest products. Many such forest-dependent communities, ethnic minorities or farmers lack both land security and political representation (Wolvekamp 1999).

2.1.2 State Control of Tropical Forests

Transformation of the use and expectation from forests has historically started from very primitive tribal/indigenous communities living within or on the fringe of forest areas fulfilling their livelihood-oriented basic needs. Tropical forests thinly surrounded by humans were the ultimate local saviour socially, economically, culturally and spiritually. On the other hand, there are some areas where civilisation was built in harmony with the forest. Forest civilisation, developed by Indio people, which was destroyed by the European invasion, is a good example of coexistence between humans and nature. However, it has to be acknowledged that the low-population factor certainly helped the cause of balanced use of the forest. Similarly, for more than 400 years two distinct ethnic groups, the Chachi and Afro-Ecuadorians, through their respective cultural practises have managed forests sustainably, providing them with food, clothing, medicine and ritual necessities (Gamboa, in Colchester 2001). This umbilical relationship between tribal people and forests was first disturbed in the colonial era. Heske (1937) described dense forests in India as the ultimate edifice for the spiritual philosophy this country has given to the world. Colonial expansion in the mid-nineteenth century in India was marked by the establishment of railways spurring greater access to forest resources which were fed to industrial revolution back home. Hence colonial government claimed large tracts of forests as forest administration also was established in the 1860s. Since then, the issue of land rights and indigenous peoples, especially in the forestry sector in India, has been highly sensitive because many tribal communities have been divested of their customary rights for purposes such as large dams, mining, timber contracts or biodiversity conservation.[2]

The presence of colonial powers in the continents with tropical forests had the effect of causing reorganisation in local land use and power structures in response to colonial markets and government pressures. Even though the colonial powers did not seek massive changes in the ownership structure of land use and power, enough damage was done to break down the traditional structures (Vosburgh 2003). Nevertheless, colonial governments were strongly in favour of absolute proprietary rights of the state over the forest, and state monopoly. Not enough consideration was given to the fact that customary use of norms by local people was regulated by their indigenous institutions and by customary relations within and between villagers. All uncultivated land went to the state while discretion of rule prevailed. However, the exponential population growth since the beginning of the twentieth century is very much coherent with the mounting pressure on tropical forests as both locally growing populations and industrial needs of the developed world targeted these forest ecosystems. Hence, according to the FAO Forest Resources Assessment, Earth's forested area is in decline, mainly due to the conversion of forests to agricultural land (FAO 2005). With sovereignty of several erstwhile colonial states returning around the mid-twentieth century, the ownership of forests was consolidated centrally by the independent states, promoting the culture of control and command.

2.1.3 Emerging Set of Stakeholders and Conflicts

The major processes associated with deforestation are largely anthropogenic, including clearing land for agriculture and livestock production, human settlement, commercial logging, mining, hydroelectricity projects and military activities (Kaimowitz and Angelsen 1998; Allen and Barnes 1985; Bawa and Dayanandan 1997; Rudel and Roper 1997). Nevertheless, higher deforestation and forest degradation rates after the postcolonisation phenomenon indicate that centralised forest governance systems treated forests largely as "revenue cows" as emerging states tried to build on the new development paradigms of agriculture expansion, cattle ranching on clear-felled forests for meat production, industrial growth and massive infrastructure establishment. Development largely occurred in emerging urban centres as growing but alienated rural populations (e.g. in India, Bangladesh, Indonesia, Malaysia, Haiti, Democratic Republic of the Congo, Brazil) based on subsistence added to the anthropogenic pressure on the forest ecosystems. The "control and command" management of forests akin to centralised governance systems went hand in hand with the gradual alienation of authentic forest users

[2]In 2008 the Indian parliament passed the forest tribal rights bill but its implementation is still inadequate.

from traditional access to forest resources. One of the key features of this top-down system of forest use was the induction of "concessionaries" in good company with states as an agent to log the forests clean. That centralised systems of forest governance cannot be the harbinger for the rescue of tropical forests was very much evident from the "Rio Summit" in 1992. This summit was instrumental through the adopted Agenda 21, to endorse the participatory role of local communities in decision-making favouring sustainable forest management. It is also more than 40 years since discussions were initiated for an international tropical timber agreement, in an early attempt to align the conservation and development of tropical forests. Hence, for about two decades there has been a popular move to devolve forest governance from centralised government to a lower level of government (e.g. civil society, local governance bodies, private sector).

Global interest in sustainable management of tropical forests has emerged. Partly this is evident from the fact that the focus is on identifying principles, criteria and indicators on the basis of which sustainable forest management can be judged. As a result of renewed global attention to safeguard forest cover whilst sustainable use occurs, there are now many people with an interest or stake in forests. Transformation of the stake in a forest from a single user to multiple stakeholders in formal and informal institutions is therefore bound to generate clash of interests. This brings in the accessibility and rights issue of actual forest dependents for whom forests are the primary assets for supporting their livelihoods and local economy. Nepal's case is a classic example of shifting of ownership and with that the power of exclusive use of its forests from a "free for all status" prior to 1957 to a more people-oriented forest governance (see Fig. 2.1) after a

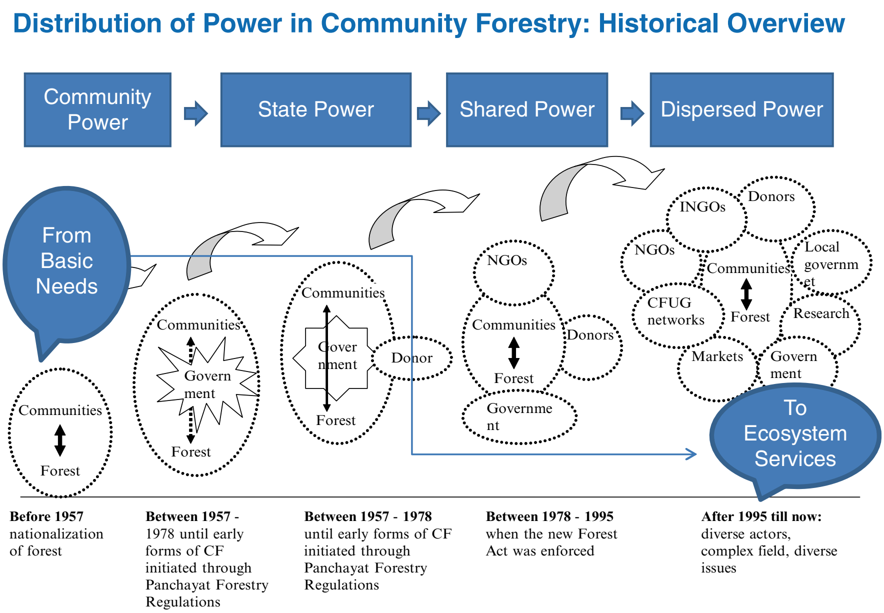

Fig. 2.1 Power distribution in Nepalese forestry, a historical view (after Ojha 2008)

period of strong state control. Figure 2.1 can also be seen in the context of a growing number of stakeholders having a stake in a power game regarding forest governance as well as their interests in sustainable forest management after an era of deforestation and forest degradation. From the sheer basic needs logic to paradigmatic focus on ecosystem services multi-stakeholdership is characteristic in the dispersed power situation. In turn, forests from the cradle of civilisation and culture have become objects of multiple interests and stakeholdership for the economy and conservation. Decision-making cannot be firmly unilateral or monopolistic but must be collective and consensual. The timeline below the picture in Fig. 2.1 also shows a clear increase in the number of interest groups or stakes. All these stakeholders have different rights and interests along a continuum of relevance for day-to-day forest management (Colfer 1995).

With increased population, increased consumption and higher demand for Earth's resources over the past century, forest governance has become a burning issue. This is also because the state's monopoly has simply not worked. Forest governance changes ushered in through decentralisation processes across the globe have resulted in several stakeholders articulating their interest and role to shape governance and with that forest management in a consultative mode. For instance, having two chief stakeholders, i.e. state and local communities, and adopting community forestry has boded well for Nepal because the historically high rate of forest loss of 0.5% annually (i.e. of forest and shrubland combined; DFRS in R-PIN Nepal 1998) since 1978–1979 has been slowed and there is formidable evidence suggesting that community-managed forest regimes lead to reduced deforestation and forest degradation. This was partly assessed for the hills, where community forest management modality is well anchored. Despite the genuine attention given to participatory forest management, addressing the drivers behind deforestation and forest degradation remains elusive. From the angle of poverty as one of the key drivers for such a situation, the complex connection between forests and human livelihoods has led to criticism that forests are poverty traps, as not enough wealth is generated for poor communities to escape poverty (The Forest Dialogue Review 2009). On the other hand, the barrier of an inadequate or weak enabling framework has meant that the value addition of forest products to promote business cases on pro-poor and socially inclusive forestry has not reached the "economies of scale" stage. In an interesting study, forest tenure distribution by tenure categories was analysed for 25 of the 30 most forested countries (Sunderlin et al. 2008), showing that 74% of the forest land is still with the state, and a mere 11% has been given to local communities for management.[3]

The challenge of reducing deforestation in the tropics as shown above is complicated by the fact that, in most cases, it results from a combination of social, economic, political, biophysical, historical and other factors, indicating that rather than one single mechanism, a mix of policies and approaches is required (Geist and Lambin 2001). Accordingly, policies aimed at curbing deforestation and forest

[3]As per the FAO's Draft Global Forest Resource Assessment 2010, up to 80% of the world's forests are publicly owned, but ownership and management of forests by local communities, individuals and private companies is rising.

degradation in the tropics range from strict preservation of undisturbed forest areas, to land-use policy reform, promotion of timber plantations, and regulation of forest use, through to market-based incentives for sustainable forest management. However, with the growing dominance of capitalism and democracy as global operating standards, the concept of private property lies at the heart of political and economic assumptions. Through its policy instruments the state is increasingly trying to harmonise its interests with the interests of local communities. On the other hand, local communities struggle to maintain a balance between their societies and forest environments when faced with rising populations, growing demands for basic needs and money, and increasingly strong external physical and psychological pressures (e.g. through outmigration of youth and deficient local labour) through forced state-led development concepts.

2.2 Tenure Security as a "Panacea"

In 1989, the FAO published the *Community forestry rapid appraisal of tree and land tenure*, which referred to tenure as a "bundle of rights" to land and trees. In the publication it was argued that forest initiatives need to develop a "tenure strategy" that constitutes an incentive for tree planting and forest management. Two decades later, tenure across the tropics has emerged as a fundamental issue in efforts to achieve sustainable forest management and to meet the needs of the rural poor, including the right to food (FAO 2006, 2007). Although most of the world's forests remain under public ownership and state control, especially in developing countries, a diversification of forest tenure arrangements is taking place as stakeholdership in forest sector multiplies, and as a result of that, in various regions of the world revised forest policies and laws are put in place. The nature of these new tenure settings differs considerably, reflecting the past and recent history of the countries, the different approaches selected by governments to improve forest management, and also the growing voices of local stakeholders demanding recognition of their rights and a role in decision-making. Many of the tenure reform processes such as privatisation, titling and restitution or redistribution of land are not adequately implemented because of a weak enabling environment, a lack of involvement of the beneficiaries in decision-making about the new tenure arrangements and poor communication. Inevitably, this creates insecurity, mistrust and conflict, increases the fragility of tenure and reduces interest in proper forest management. The analysis of different forest tenure arrangements, including those that are the result of tenure reform, shows that a number of important elements have to be in place to make them successful. These range from a supporting policy, legal and institutional framework to strengthening the capacities of all stakeholders involved, including the staff of state institutions initiating tenure reforms.

Current tenureship for forest land across the tropics has a colonial heritage and exists along the communal tenure system. As the issue of land awareness comes to the fore, engendered partly by population pressure, relative price changes and the commoditisation of land, conflicts develop: farmer–grazier, farmer–farmer, indigenous people–state, etc. These inconsistencies in the tenure system reduce the possibility of negotiating lasting solutions in land-related conflicts (Colfer et al. 2008). The social cost of this behaviour is limited not only to mutual distrust but also to opportunity costs of both time and financial resources mobilised by the parties in conflict to follow up legal procedures (Baye 2007). It is evident from the above account that the traditional land-use system is being altered by a global environment which imposes neoliberal reforms such as privatisation and liberalisation. The context is further reinforced as the commercialisation of agriculture, pastoral and forest products is reshaping relations between production and exchange, leading to new demands for access to and control over land and its related assets. Land tenure systems influence and are influenced by conflict situations, which engender insufficient access to primary assets – a situation that is the outcome of economic, social and political processes, and their interactions. These interactions are mediated through a wide range of both formal and informal arrangements, including tenure arrangements. Rapid changes in economies, environmental conditions and social structures demand institutions that can transform themselves to meet new priorities and shifting demands (FAO 2008); hence, there are uncertainties in arriving at the right mode of tenure.

2.3 Characteristics of Forest Users

2.3.1 From Devolution to Multistakeholdership

The previous points have elaborated a historical context of forest use in the tropics, which has now culminated in a forest sector that attracts wider interest and commitment from a network of actors ranging from policymakers to resource managers, and from advocacy groups to private companies. However, despite this change in stakeholder scenario, in many countries the state has maintained control over land and forest resources as part of its power base. In practise, however, “many governments continue to prove unable to carry out the responsibilities they give themselves. Policy options inappropriate to local contexts, weak institutional capacity to implement them and corrupt and rent-seeking behaviour all contribute to limit the effectiveness of state control” (Cotula and Mayers 2009). What usually drives governments to engage in tenure reform by granting management rights or ownership of forests or both to different stakeholders (private individuals, companies, communities or other local groups or to a combination of several of these) is the need to devolve management responsibilities to those who are closer to the forest and have a stake in its conservation or who may have better capacity for forest management than state institutions. A second objective may be to promote local economic development by providing opportunities for poor local people to generate income from the management of forest resources. Devolution of ownership or management rights or both may also be part of a general decentralisation process. In some circumstances, however, tenure reforms are the consequence of the realisation of the state's failure to prevent further degradation, rather than a search for more efficient and socially acceptable management approaches. The international community and also the emerging voices of farmers and communities in the countries themselves are putting national governments under growing pressure.

Increasing devolution of ownership or management rights ultimately results in more diversified forest tenure systems that are officially recognised. It helps legalise de facto existing tenure systems by providing more tenure security to those who depend on forests for their livelihoods or who use forest resources to generate income. As a result, more diversified tenure arrangements have the potential to contribute to poverty alleviation and to reducing deforestation and forest degradation. In situations where the capacity of state institutions is weak, especially at the local level, diversification of tenure systems involving local stakeholders may also contribute to more sustainable management of forests and to reducing deforestation and forest degradation. It should be emphasised that security of tenure is a necessary but not sufficient condition for effective forest management and has to be accompanied by an appropriate policy, legal and institutional framework. It also has to take into account the local context: simply introducing models from other countries has generally resulted in failure.

Land tenure and resource availability can play a critical role in the land-use decision-making process, resulting in different types of land-use changes. A study in Thailand investigated the role of land tenure security and farm household characteristics on land-use change in the Prasae Watershed using geographic information system and farm-level data. Conversion of forest to annual crops and subsequently to perennial crops was a typical land-use change from 1982 to 2004. Tenure insecurity was found to be associated with deforestation and forest encroachment. Insecure landholders adopt perennial crops to acquire basic land-use rights and entitlement to subsequent legal registration, whereas more secure land tenure is seen to have economic advantages for production and long-term investment. In case study 2.2 (Kotru 2009), rehabilitation of degraded forest land through the community's involvement in forest management brought a drastic and positive change to the local forest ecosystems. Although land tenure security can act as a crucial factor in land-use decision-making, farmers opt for different land-use options on the basis of characteristics such as farm size and available labour. It emerges from the above discussion that an effective policy should aim to improve both farm productivity and land quality while protecting the remaining forest.

Tenure reforms should be incorporated in a broader context that includes governance and regulatory frameworks; conducted in isolation they are bound to fail or have limited impact. Empowerment will not come from titling alone, and titling does not ensure the capacity to benefit from forest resources or their equitable sharing, but requires a lot of additional cross-cutting support. From the above historical account it emerges that sociopolitical, socioeconomic and environmental needs and compulsions have largely influenced how tropical forest ecosystems have been used. State-designed policy frameworks, in general, have increasingly adjusted to the emerging needs of inclusive participation of forest-dependent communities (e.g. the case studies in this chapter). The institutional analysis and development framework proposed by Ostrom (1990, 2005) as the core of community-based resource management theory in a way matches the current attempt at democratisation in the forest sector. Along a value chain of forest products and services, it tries to define the physical environment, attribution of communities to the action area, actors and action situations, thus generating patterns of interaction and outcomes. Figure 1 shows that from state dominion in the 1950s to the democratisation process of the mid-1990s, a wide spectrum of direct and indirect forest users are interested in the forest sector. The so-called multistakeholder process in the forest sector is an emerging paradigm articulated, for example, in a piloted District Forest Coordination Committee (DFCC) directive (MoFSC 2005). The aim of establishing DFCCs is to institutionalise the forest sector decentralisation process and to promote good governance in biodiversity conservation and forestry sector management. DFCCs are promoting multistakeholder representation in decision-making processes, raising ownership in forest sector programmes, capitalising social learning, managing their problems and disputes internally and thereby raising a sense of self-reliance through generating and mobilising locally available resources (Rana et al. 2009). Issues affecting the district forest sector are openly discussed and special attention is given to livelihood improvement and forest product distribution for the district population as a whole. The other aspect of multiple users becoming part of the forest sector is related to the recognition that third-generation issues (e.g. more income and employment, pro-poor and inclusive outreach, enterprise-oriented forestry) are yet to be addressed despite progress made in community-oriented approaches (e.g. learning from Nepal, Indonesia and India). Therefore, as the range of goods and services derived from forests has increased, forest users have undergone changes in their profiles, each exercising differential strategies to use and manage forests.

2.3.2 Main Stakeholders and Their Characteristics

Important stakeholders and methods to identify and define these are widely applied (Colfer 1995). The rationale behind this identification of stakeholders originates from the premise that all stakeholders have the common interest of sustainable forest management providing a flow of goods and services on a continual basis. Accordingly, stakeholders may be distinguished on the basis of their proximity to the forest, preexisting rights, dependency, etc. The categorisation adopted in the following sections takes a practical approach of significance emerging from the historical context described earlier, and as being direct stakeholders.

2.3.2.1 Forest Dwellers (e.g. in Brazil, India, Indonesia and Myanmar)

This type of user – often termed as “indigenous groups” – is clearly the most important stakeholder and is still prevalent across the rich cover of tropical forests and follows a livelihood strategy dependent on forests. Although this type of user may have a role as a hunter, gatherer, etc., the use is generally within the sustainability levels. This type of user is closely related to the aspect of “shifting cultivation” and is currently coming under extreme stress owing to reduction in forest cover. The situation is further complicated by increased control of forest cover by the state and alienation of indigenous forest users, state’s often unplanned development initiatives (roads, hydropower dams) in and around forest areas and overall restrictive policies of states to focus on conservation. Increasingly, such a user type is seen as an encroacher on the forest although there are also policy processes in operation where the rights of such tribal/ethnic groups are being secured (e.g. Tribal Forest Rights Directive in India, rights of forest-dependent ethnic groups in the proposed new constitution in Nepal, forest rights for ethnic groups in Brazil). It is interesting to note how fast indigenous peoples’ interests and rights are being recognised and applied by various countries in Asia and by international development agencies. Historically, different legal, economic and political situations have marginalised them from communal management of land in their ancestral domains. And current state policies, laws and development programmes generally do not accept the domains of indigenous peoples and attempt to divest such lands from communal management. However, there are reasons for optimism. Organisations of indigenous peoples and forest-dwelling communities are fast gaining voice and opportunity, and after decades of limited action many countries are beginning to consider far-reaching legal and policy reforms. There is a major opportunity to advance the rights and livelihoods of forest peoples by establishing the institutional foundations for sustained conservation and forest-based economic development.

2.3.2.2 Subsistence Users

These users have quasi-shifted from a purely forest dependent lifestyle to a more agrarian orientation where conversion of forests into agroforestry and homestead systems dominates. Although dependence on farming dominates, these users exist in all tropical countries using forests for subsistence. The International Labour Organisation (ILO[4]) estimates that for every job in the formal sector in forestry there is another one (or two) in the informal sector (ILO 2001). It is because of these users that degradation of such ecosystems can be immense (e.g. grazing, conversion of forest cover into agriculture). It is also here that community-based approaches have been initiated on a large scale. Owing to their better accessibility to the state’s

[4]ILO Convention 169 is a binding international treaty to exclusively deal with the rights of indigenous and tribal people.

delivery systems, this group is very well networked to advocacy institutions, markets, politicians and development programmes in general.

2.3.2.3 The State as a "Revenue Monger"

The state is certainly the main user and owner to date (e.g. 74% of forest land is still with the state). This use was primarily for generating revenues barely two decades ago, but increasingly a balance is being sought between conservation and production. Similarly, the state is increasingly realising that forest degradation cannot be controlled through command and policing but can only be controlled by inclusion and empowerment of forest dwellers and subsistence users (see Sects. 2.3.2.1 and 2.3.2.2). In the power game of authority over forests, states have started to yield management rights to immediate and primary users on the ground. Most of the state institutions have an old structure hardly adapted to the fast-changing forest sector scenario (e.g. climate change agenda, decentralisation process, private sector involvement). Hence, the capacities of such institutions to address the emerging needs of the sector have enormous deficiencies. Apart from this, a genuine aversion for change management brings about a resistance to reform processes, making adaptive structural and service delivery changes tedious and abnormally slow.

Nevertheless, the forest sector in previous decades has partly lost its instrumental role in providing revenues to the state as protection-oriented conservation strategies have unfolded.

2.3.2.4 Private Profit Makers/Concessionaries

Although users of this type may not be the owner of the forest, they wield a lot of influence in designing the management of some of the richest tropical forests around the globe (e.g. in Brazilian rain forests, Indonesian and Malaysian concessionaries, Cameroonian timber merchants). As service providers for generating forest revenues (mostly from timber), users of this type do not necessarily follow a sustainable-use principle. Very often, the role of this user type in association with the key decision makers of the state provided the bulk of corruptive practises that exist in the forest sector. The private sector is fast emerging as an important actor as initiatives for public–private partnerships bridging economic and conservation cooperation between the state and communities show potential for rural income and employment generation. However, issues of forest law enforcement, governance and trade have not yet unlocked the role of the private sector for the benefit of forest users, as an enabling framework to do so remains elusive.

2.3.2.5 Civil Society

In recent years, civil-society organisations as representatives of interest groups and networks from local to global levels of forest governance (e.g. Global Alliance on

Community Forestry, Greenpeace) have increased in significance. At the micro level these are often known as community-based organisations, which have become major players in forest-related issues in most countries, often challenging established positions and poor levels of accountability and transparency. Although differing in perspectives and approach, these groups focus attention on conserving biological diversity, extending protected areas, driving forest certification and improving forest governance to reduce illegal logging and to stress the connection between forests and livelihoods. As a global coalition, international agencies, e.g. the United Nations Environment Programme (UNEP[5]), regional and community organisations engaged in conservation, research and development, and civil societies are very influential as policy-influencing institutions working to encourage greater local, national and global commitment and action on pro-poor tenure, policy and market reforms. As partners, civil societies conduct work in specific areas of their regional and thematic expertise. These engage with a wide group of collaborators who participate in and support, for instance, rights-related activities around the world. Such a strategic coalition goes beyond the traditional set of international development actors to involve a wide spectrum of organisations, each of which provides a critical perspective in the larger chain of actors necessary to advance change. On the basis of their experience, it is found that empowerment and asset-based development are part of a process that is dependent on a set of enabling conditions, including security of tenure for access to and use of natural resources. These core beliefs of several civil societies thus focus on rights and governance, and form the foundation for programmes and activities. The decisions of policymakers and their attitudes towards reform are influenced by a number of actors at different regional, political and social levels. Often the facilitation role provided by civil societies to networks seeks to bring together strategic actors with the influence and knowledge to share and to advance tenure and policy related discussion mobilising reform processes at many levels and with many constituencies. This includes bringing together networks of senior policymakers from large forested countries, networks of policymakers at regional and national levels, and supporting networks of indigenous peoples and forest communities to make their voices heard in regional and international dialogue.

2.4 Current Forest Management Focus and Design

Currently, climate change and decentralisation aspects present a moving target, having the potential to drive change in existing relationships between and among producers and consumers of tropical forest products. The consequent increase in demand for ecosystem services is slated to transform forest conservation and

[5]UNEP is the designated authority of the United Nations for environmental issues at the global and the regional level. Its mandate is to coordinate the development of environmental policy consensus by keeping the global environment under review and bringing emerging issues to the attention of governments and the international community for action.

management significantly. To realise services from forests in the context of climate change, the forest stakeholders may need to return to the drawing board to increase the effectiveness of sustainable forest management. Existing institutional mechanisms, however, have thus far limited themselves to sustaining forest cover at levels that meet the demand for food, fibre and fuel. The shift in favour of enhancing environmental services will impact the existing political-economy of forest management. Maintaining such services poses challenges, especially where trade-off between the production of goods and the provision of services is precariously balanced. However, in low-income situations, sustainable forest management faces far more constraints, reflecting limited ability and willingness to pay for the additional costs involved in adhering to the environmental criteria. Consequently, in tropical areas, the proportion of forests that are sustainably managed remains very low (ITTO 2006).

In densely populated Africa, South Asia and Southeast Asia, forests are vulnerable to degradation caused by illegal logging, fuel wood collection, grazing and poaching. Community-based forest management has contributed to forest conservation, but skewed benefit-sharing has not allowed for maximum gains from community participation in forest management. The success of such approaches depends on establishing appropriate trade-offs between conflicting objectives (FAO 2009). This requires a robust institutional framework and good mediation skills to negotiate a lasting compromise. The current management is organised largely as described in the following sections.

2.4.1 State-Managed Forests

State-managed forests have regular operational plans developed from colonial times. These usually have a scientific basis and are prepared on the basis of production and protective uses, for which several silviculture systems are adopted. However, the enigma of state-managed forests in the tropics is the overall demanding pressures these are subjected (see point 1). Normally parallel departments are created or concessionaires are hired to do the technical management part (i.e. harvesting, logging, etc.). It is in this category also that conservation of forest areas with a strong regimen of protection is being practised. However, most of the conservation areas have been developed through alienation of original forest users/dwellers (e.g. in India, Nepal, Myanmar, Laos and Cameroon). Originally pristine or natural forests, these forests are being systematically converted into plantation forests with new tree species (e.g. exotic) mixed regularly. On the other hand, as Uebelhör and Drews (Chap. 4) report, it is increasingly recognised that indigenous peoples and local communities often have a deep understanding of their environment and their forest's ecology. This knowledge forms an important basis for the conservation of global biodiversity and for its sustainable use. The past two decades have seen a resurgence of interest in the many products and services of forests and so current management systems are challenged to address economic, social and ecological aspects of sustainable development. For a few decades such modalities of management have been emerging as "extractive systems" (e.g. rubber, palm oil), so forest land is often used for horticultural purposes and is used to deliver revenues in the shorter term.

2.4.2 Community-Based Modalities

Although a very limited forest cover along the tropical belt is managed with or by local communities, participatory forest management is fast appearing as a panacea for saving these forests. As Belcher et al. (2008) points out, throughout the tropics rural households are now involved in a wide range of systems for the management of forest resources. An interesting and valuable class of systems falls on the continuum between pure extraction and plantation management. These systems are fundamentally being promoted not only to involve local knowledge of communities in planning, implementation, monitoring and protection but also to make biodiversity–productivity trade-offs. There is often a trade-off between biodiversity (by some measure, often just a species count) and productivity (either the total value of production per hectare or the profit per hectare) in resource management. A case study from India (Chap. 3) is a classic example of production and protection aspects which can be addressed through community-based approaches. Nepal and Mexico through their characteristic "Community Forestry" modality have demonstrated that with community-based-management operational plans and their full implementation by the local communities (e.g. planting, harvesting, marketing of products) the forest cover (in the mid-hills) can be increased. However, issues of inequity, elite capture and exclusion of poor/disadvantaged groups are becoming evident. Nevertheless, one of the key arguments emerging in such a type of management is that community forestry is being promoted at the cost of destruction of state-owned forests.

The overall management decisions in both modalities described above are becoming complex as the number of stakeholders showing proactive interest in production and protection of value-added goods and services of forests is multiplying. The so-called multistakeholder processes are becoming important to include heterogeneous interests of differential actors. Collaborative forest management in Nepal's "biodiversity hotspots" in the tropical Terai forests is a good example of an evolving model for social inclusion and pro-poor focus. On the other hand, the concession system of Peru (Chap. 5) is expected to lead to sustainable forest management but has yet to show lasting results.

As Grossheim mentions (Chap. 5), the forest concession system was adopted by the Peruvian government at the beginning of the century, and has not yet achieved its purpose since it has not contributed significantly to the Amazon Region's rural development. However, technically speaking, the concession system is solid ground on which to improve sustainable forest management, and even more so if one considers the unsustainable forest use before 2000. Certainly, it also appears important to adjust the concession design in almost all its dimensions. However, the community-based approach if not complemented by other programmes such as income-generation activities and agroforestry initiatives ensuring short-term benefits will take time to make a positive difference for sustainable forest management.

Except for plantations owned by private companies, the role of the private sector is linked to several levels of value chains that emerge from use of forest products. Thus, in both of the modalities described above, the private sector may change its role to be a marketing agency, a harvesting company, or for value addition of raw products, etc. However, in most of the tropics, the role of the private sector in public–private partnerships is emerging fast.

It can be summed up from the above account that understanding the current political, economic, ecological, and social situations; the power relations among the various actors involved in forest management; the often unequal distribution of costs and benefits of forest exploitation; the discourses of science, neoclassical economics, sustainable forest management and national development; and the colonial and precolonial roots of current deforestation in these regions is becoming more important than ever. The current climate change discussion adds a very challenging dimension to future forest management as managing carbon is added to the menu of services fast-degrading tropical forests have to deliver. In a nutshell, this would mean that the major challenges revolve around addressing the wider field of forest governance and not just around government agencies, policies and regulations, but will include (adapted after Don Gilmour 2009[6]):

- The whole system of managing and governing (formal and informal).
- The process by which forest management decisions are made and implemented (power relations).
- The implementation of sustainable forest management in the tropics is fundamentally associated with a conflict over access to valuable resources. Managing this conflict constructively is critical to the outcomes.
- Many of the transformations discussed come about through conflict (small and large) and we do not yet understand enough about how change comes about at these critical moments – politics rather than policy.
- Influencing the carbon forestry debate to internalise the basic principles derived from sustainable forest management (e.g. to prevent co-option of participatory forest management by the carbon forestry agenda).

2.5 Emerging Paradigms

The emerging paradigm for tropical forests from the foregoing account is derived from formidable current and future challenges. Foremost is the challenge of how to mainstream multistakeholder processes without causing conflicts as well as seeking

[6]Adapted from a presentation given by Don Gilmour at the International Community Forestry Workshop (2009) in Nepal.

a balance between community and state ownership based on principles of sustainable production and protection. Two extreme situations reflect this: according to UNEP's Global Biodiversity Outlook 1 Report (2001), about 60%, and possibly closer to 90%, of all species are found in tropical moist forests; on the other hand, legislative instruments are being introduced to safeguard indigenous interests at the country level. Nevertheless, climate change and CoP 13 have brought tropical forests "back to business". Factors underlying forest land-use change and conversion in the tropics as demonstrated above are spread across vertical and horizontal levels of forest governance. Thus, factors such as economics, policy and institutions, technology, social and cultural dimensions, demographic aspects and others (natural factors such as soil quality, etc.) will determine sustainable forest management. As Thompson (Rametsteiner and Simula 2003, p. 88) explains, "Instead of seeing the world as frozen in a black box of equilibrium and harmony, we must think about the world as an ever-changing system poised at the edge of chaos". It follows that the sustainable forest management is a complex concept, "specifically designed to embrace and reconcile the different interests in forests" that include productive, ecological, economic, social, cultural and spiritual forest values. Domestic and international policies concerned with sustainable forest management employ instruments ranging from traditional "command-and-control" regulation to economic mechanisms that attempt to harness the power of market-driven incentives (Cashore and McDermott 2004). Yet, as Pearce (1998, p. 28) suggests, "while market mechanisms might be beneficially invoked for a range of forest values, they cannot eliminate altogether the need for regulation for some values such as the aesthetic appeal of landscapes and the cultural value of wilderness, which do not lend themselves well to economic instruments for forest management". Sustainable forest management is now more seen as a management regime that integrates and balances social, economic, ecological, cultural and spiritual needs of present and future generations. Nevertheless, the above definition of sustainable forest management also shows that interventions and milestones of the state and the immediate dweller are now no longer challenged only by firewood extraction and usufruct logic but are now also challenged by greater issues of income and employment generation, climate change vulnerability and, last but not least, by whom and how such a forest should be management-financed [including Reducing Emissions from Deforestation and Forest Degradation (REDD)]. However, approaches in the following sections are suggested to ensure that the challenges of insecure tenure, deforestation and slow forest sector reform are met and sustainable forest management happens.

2.5.1 State and Community Partnerships

State and community partnerships must improve as there is a strong conceptual basis (particularly the sociology of sustainable forest management) for moving ahead, but this is not always applied. The basic sociology (which addresses inclusion, etc.) is often lost in application of standardised implementation procedures. This also means that regulatory reform is slower than tenure reform (i.e. people can own the land but may be constrained from using it). Lessons pertaining to an enabling framework need to be identified and mainstreamed at the policy and management in practise levels. Various forms of participatory forestry are expanding globally and are now a recognised part of the forest management landscape. Participatory approaches must be universally applied to get maximum and effective cooperation of a wide range of stakeholders and first and foremost the local communities and their institutions.

2.5.2 New Financing Instruments

The financing of sustainable forest management is becoming a huge challenge. Increasingly, the role of payment for environmental services rendered by local resource managers is gaining momentum (e.g. based on carbon, watershed). For instance, payments for environmental services articulated through REDD and Clean Development Mechanism (CDM) approaches make economic sense in many (most?) situations, although it has to be recognised that such payments can only be an additionality to support sustainable forest management. On the other hand, these approaches may prove to be complex transactions and make sense only if the community is engaged and its rights and benefits are respected. However, carbon forestry has the potential to recentralise power if national governments control the management agenda.

2.5.3 Social Inclusion and Governance

Power relations and their clarity must be set at all levels of governance and especially between two main actors, government and civil society. In the case of the Changar example from India (Chap. 3), the increased role of women in forest management shows that power relations can be changed over time for the better, but since stakeholdership is becoming wider, we have to involve elites as well as the poor. Local elites need to take some power from national elites, as this creates space for local communities to occur. It is to be noted that here "trust" is recognised as a critical element of effective partnerships, but what does this mean in terms of building (and breaking) trust? If governments recognise a little bit of rights, they will get a little bit of conservation.

2.5.4 Economic Development

Although participatory forest management regimes were established in degraded areas and take many years to restore productivity, we will only see the real economic benefits in the coming decades (e.g. economic microenterprises across Asia-Pacific, small sawmills are appearing in Nepal). However, regulatory frameworks (forestry and trade ministries) hinder maximisation of the value chain of forest products. Economic development must be the underlying principle for leveraging cooperation from forest-dependent users/communities. It is clear that forest-based livelihood improvement does not and cannot equate to poverty reduction, as the factors and solutions are diverse and therefore complex. It has been established that forestry alone cannot solve the problems of poverty and exclusion and other public investment programmes will have to complement it (World Forestry Congress 2009).

2.5.5 Technical Management

Management of tropical forests has become a complex phenomenon. Use of indigenous knowledge with scientific logic of management has shown positive results but needs to evolve further. However, striking a balance between fulfilling the international conventions (CBD) and local livelihoods has shown tremendous progress and we can build on this. Technical management concepts in future will have to prove that these are biodiversity- and climate-smart whilst material yields for local communities are not curtailed. Scientific forest management jargon has to be replaced with adaptive management, understandable and practical at local levels.

2.5.6 Cross-Cutting Domain

Since sustainable forest management cannot be seen in isolation anymore from the politics and practise of other sectors regarding forests, it is inevitable that the state and donors will play ongoing critical roles in terms of building institutional capacities (e.g. community, state, private sector), information and knowledge networking, dissemination of knowledge (best practises), supporting advocacy on influencing policy and regulatory frameworks, and practise of management paced with accelerated needs for adjusting forest management.

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Chapter 3
Participatory Forest Management and Sustainable Development Outcomes in the Subtropical Himalayas: A Sequel of Environment, Economy and Equity through Social Empowerment

Rajan Kotru

Abstract For rehabilitation of degraded hills of the Shiwaliks in the Kangra district of Himachal Pradesh, India, the Changar Eco-Development Project initiative between 1994 and 2006 adopted a watershed development approach, of which participatory forest management involving local communities in decision-making on forest resource planning and implementation was the key component. The operative planning sequence of the participatory forest management (PFM) was (1) problem identification and potential assessment, (2) planning of forestry resources on common land and private land involving village communities and user groups, (3) forest management capacity building of user groups, (4) local institutional strengthening and (5) orientation of the forest department to PFM practises. The productive and protective potential of afforestation measures on degraded land to address ecological, economic and social problems has been demonstrated visibly. These areas are now managed and rehabilitated through adapted forest management practises led by local communities, as increased supply of forest products (e.g. fuel wood, fodder) and services (e.g. water recharging) is met. Moreover, the standing stock in plantations has gained a substantial economic value. However, such innovations will have to be backed up and scaled up by forest managers (e.g. including the forest department) to obtain broader lessons which could then be mainstreamed through policy and management in practise, improving forest governance so that multifunctional goods and services of regenerated forest cover are sustained.

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3.1 Context

The Changar region, situated in the district of Kangra in the northwestern state of Himachal Pradesh, forms the core of the Shiwaliks – the lower Himalayas. In local dialect "Changar" signifies remoteness, rugged hilly terrain and water scarcity. The region is conspicuous by its fissured hills, flood plains and steep slopes. It is a zone of naturally sensitive geological features and unstable soil profiles. Natural erosive processes have been accelerated by human use. Human impact is manifested through the increased human and livestock population and inappropriate forest management. This has led to excessive use of forests, causing steady but obvious resource degradation, particularly that of land and water resources. However, problems galling the practise of sustainable forest management are multidimensional and run across vertical and horizontal levels of forest governance. Centralization of forest governance with the state as the sole custodian of forests by imposing restrictive policies and legislation (National Forest Policy 1988 and Indian Forest Act 1927) has alienated forest users from planning, implementation and monitoring processes, leading to fundamental constraints for the practise of sustainable forest management. On the other hand, development initiatives of the state are not inclusive, so that pervasive poverty and inequitable benefit-sharing of forest yields are posing a great challenge for conservation of forest resources as disadvantaged user groups rely on these resources for their economic, energy and food security. "Earth Summit" (1992) was instrumental through the adopted Agenda 21 in endorsing the participatory role of forest user communities in local decision-making favouring sustainable forest management. Subsequently, a greater recognition of local-level rules and practises while planning for forest management was the result. Cumulative and credible evidence since Earth Summit globally, whether in sustainable forest management or in fulfilling the carbon-sequestering role of forests, endorses the view that participatory forest management (PFM) modalities have the potential to complement other viable mechanisms for poverty alleviation and social inclusion. To address the forest resource deficit of disadvantaged groups (i.e. the landless and poor) while rehabilitating degraded ecosystem, new forest-based economic avenues on an equitable basis provided a viable opportunity. The Changar Eco-Development Project between 1994 and 2006 (location in Fig. 3.1) tackled forest degradation by supporting, inter alia, PFM to contribute to the achievement of the project objectives (1) to empower people in the project area to manage their natural resources for the improvement of their livelihood options and (2) to enable, motivate and mobilize relevant institutions and line agencies working in the field of eco-development to adopt an integrated and people-oriented approach.

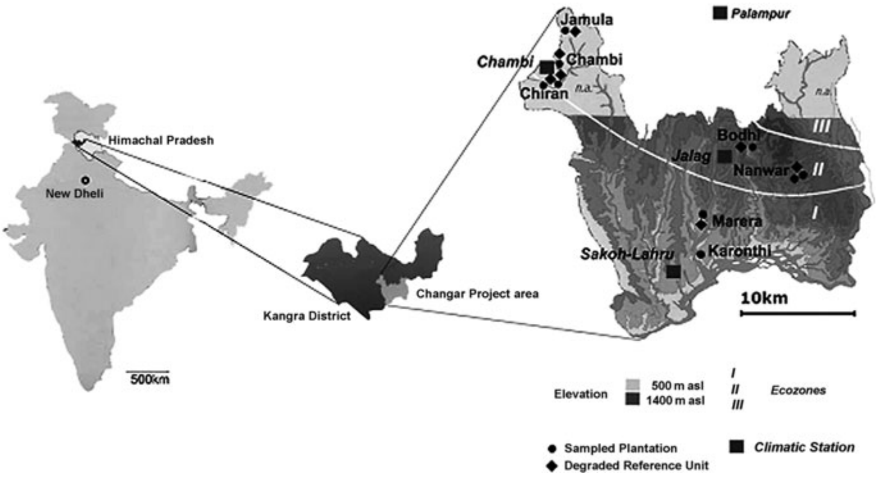

Fig. 3.1 Location of the Changar Eco-Development Project area, with plantation sample sites

3.2 Approach to Participatory Forest Management

In combination with local institutional arrangements, technical considerations are indispensable for the success of livelihood-promoting PFM. However, community mobilization and sustainable forest management cannot happen on their own if the capacity-building and empowerment of the communities in terms of local institutional strengthening and management of forest assets is not well conceived, effected and technically backed up. The people's interest in forestry is concomitant with early and intermediate benefits. The key objective of forestry in Changar, therefore, has been a long-term eco-development integrated with the regular flow of forest products needed by local communities and enriching local livelihoods in the process. The participatory process leading to the actual planning, implementation and management of a plantation emerged from the village-based integrated resource management plan (IRMP). This resulted from the amalgamation of people's knowledge and project expertise and forms the basis for PFM.

On the basis of the above-mentioned results, two types of plantations were adopted (1) closed plantations for firewood, leaf fodder and small timber production (up to 1,100 trees per hectare, mostly mixed stands of *Toona ciliata*, *Dalbergia sissoo*, *Acacia catechu* and *Albizia stipulata*); (2) agroforestry plantations for leaf fodder, grass and firewood production (250–900 trees per hectare, mostly *Acacia catechu*, *Albizia stipulata* and *Bauhinia variegata*). The groundwork for plantation as an activity was done through participatory resource appraisal (Kar and Sharma 1996) and a joint transect survey of the plantation site and the preparation of an approximate plan with the villagers through an issue-based workshop (prior to the finalization of the IRMP based on the updated rules and regulations of the Himachal Pradesh such as the Participatory Forest Management Rules or Regulations, 2001).

The plantation management guidelines (Table 3.1) developed with the local user community are part of a consolidated community forest management plan (CFMP) as part of the IRMP and consisting additionally of (1) operational rules, (2) collective decision-making regulations, (3) constitutional rules and (4) a memorandum of understanding with the forest department (MoU). In 2008, after 2 years of phasing out the project, the Appreciative Inquiry Commons. 2008 4 D cycle (for details of the process see http://appreciativeinquiry.case.edu/) was used to interact with local communities, with over 60% women as participants (20 cases out of which nine plantations were intensively surveyed) having opted for such plantations. In addition, technical management outcomes of some plantations were measured (e.g. basal area, biomass). The specific objectives were to (1) assess the physical condition of tree plantations and cumulated tree biomass (e.g. timber, firewood, fodder), (2) to assess the community's perception of forest management and (3) to reflect on current issues of forest governance for ensuring sustainability of achieved impacts. Case studies spread across the project area covered all the geographical diversities and plantation types, with clear forest management initiatives taken by local communities based on project CFMP guidelines (Table 3.1).

3.3 Reflections on the Sequel "Environment, Economy and Equity"

As established in an independent evaluation study (ABI 2006), the project impact assessed in accordance with OECD criteria clearly revealed that the local changes initiated in the natural resource management mark improvement of erstwhile degraded landscape. These changes occurred while addressing the issue of better access and benefit-sharing mechanisms leading to economic benefits on an equity basis. However, it is often questioned whether such benefits for the environment, economy and equity are sustainable once a project is complete. Hence, 2 years after the project ended was an opportune time to see the trends in maintaining the impacts effected by the project.

All the forest communities even in the post-project scenario are committed to protection of plantations despite variable quality of application of CFMP guidelines. However, local village institutions created by the project still exist, although the overall functions and performance expected from these are realized in accordance with the urgency of the need for interventions (e.g. allocation of grass harvesting patches to different households within the plantations each year or any restoration work that may be needed on the local perennial water source). Clearly, poor households regularly take firewood, grass and fodder from improved plantations, with women increasingly participating in the local decision-making (e.g. 50% women members in local village committees is still the norm). Women taking up the natural resource management issues within the village and with the forest department display proactive leadership. This was also encouraged by the project's

Table 3.1 Plantation management guidelines

Site classification	Proposed trees/plants	Management and use
1. Good deep soil, rare sprinklings of *Lantana camara*[a], good growth of grass, hardly any slope, no fire scars	*Leucaena leucocephala*, *Bauhinia variegata*, *Morus alba*, *Terminalia arjuna*, *Terminalia alata*, *Terminalia bellerica*, *Grewia optiva*, *Toona ciliata*, *Albizia stipulata*, *Albizia procera*, *Dendrocalamus hamiltonii*. Can be in close spacing (2 m × 2 m)	Production of grass, leaf fodder, fuel wood and small timber in short rotations. To be managed as a silvipastoral systems. Major yield starts from 2 to 3 years onwards
2. Good soil in patches but interspersed with small boulders and stones, rare *Lantana camara* and *Zizyphus indica* bushy growth. No fire scars	*Leucaena leucocephala*, *Bauhinia variegata*, *Morus alba*, *Terminalia alata*, *Terminalia bellerica*, *Grewia optiva*, *Toona ciliata*, *Albizia stipulate*, *Albizia procera*, *Ficus* species, spacing can be 2.5 m × 2.5 m. *Lantana camara* to be uprooted fully	Production of grass, leaf fodder, fuel wood and small timber in short rotations. No larger gaps in the canopy with short-term selective felling. Major fodder yield starts from the 2nd to 3rd year onwards
3. Dominated by stones and boulders with rare soil patches. Also bushy growth of *Lantana camara* and *Zizyphus indica*	*Acacia catechu*, *Dalbergia sissoo*, *Melia azedarach*, *Emblica officinalis* and *Leucaena leucocephala*, grafting on *Zizyphus indica*, spacing can be site-specific and when good soil is there. *Lantana camara* to be uprooted fully	Production of grass and leaf fodder with focus on soil and water conservation and selective felling to be done only after 10 years. Major yield starts from the 5th year onwards
4. Patches of good soil interspersed with stony and boulder patches. Cover of *Lantana camara* less than 50%	*Acacia catechu*, *Dalbergia sissoo* in strips and *Lantana camara* cleared patches *Leucaena leucocephala* and in depressions *Dendrocalamus*, Napier/ Seteria hybrid tufts. *Lantana camara* to be uprooted in dominant patches. Number of trees not to exceed 900 per hectare	Focus on grass production and leaf fodder initially shifting to economic use of *Acacia catechu*, *Dalbergia sissoo* after 15 years. *Leucaena leucocephala* to be used intensively. Major yield starts from the 5th year onwards
5. Patches of good soil and stones/boulders, with *Lantana camara* and *Zizyphus indica* patches covering 50–100% of the area	*Acacia catechu*, *Dalbergia sissoo*, in *Lantana camara* uprooted patches and strips plant *Leucaena*. Napier and Seteria hybrid tufts can be planted. *Lantana camara* to be uprooted in strips and in larger patches. Number of trees not to exceed 900 per hectare	Focus on grass production and leaf fodder initially shifting to economic use of *Acacia catechu*, *Dalbergia sissoo* after 15 years. *Leucaena leucocephala* to be used intensively. Major yield starts from the 5th year onwards

(*continued*)

Table 3.1 (continued)

Site classification	Proposed trees/plants	Management and use
6. Small rivulets/nalah in the making, grass growth and weeds, stones, boulders visible	To be planted with bamboo rhizomes, *Morus alba* and grass tufts of Napier/Seteria hybrids on the edges and with sporadic trees of *Dalbergia sissoo*	Focus on soil and water conservation primarily. Bamboo species and *Morus alba* to be used selectively as leaf fodder and for economic use. Major yield starts from the 7th years onwards

Sampling was carried out at regular points on a transect line running along the slope gradient, to equally represent both the upper and the lower areas of the slope. This sampling design is often applied in vegetation science (e.g. McElhinny 2006). A starting point on the upper part of the slope was selected randomly, located at least 5 m from the plantation boundary. From there, the transect followed the slope gradient downwards. The transect was then divided into four transect segments 20 m in length, with a buffer zone of 5 m around every structure plot. The central point of each transect segment represented the sampling location

[a]*Lantana camara* is an invasive weed which has established itself in all the degraded sites and does not let any other vegetation to come up

forest-product-based income generation initiatives providing short-term economic benefits to women and the poor. It is also reflected in the rehabilitated status of degraded planted sites as technical training of women regarding forest management has paid off. Hence, all technical interventions in plantations have in common that dead and decaying trees were harvested (e.g. social classes 4 and 5) regularly. Similarly, crown length was lopped up to 20% annually (e.g. for fodder and fuel wood). Partial grazing especially in thicket-like plantations was practised on a seasonal basis (i.e. non-growth season between November and March). In 10 years, some plantations registered substantial basal area despite regular use, with a protected plantation having a basal area of 21 m^2/ha (Table 3.2, protected plantation III). On the other hand, silvipasture plantations, which are mostly on private and communal land and are regularly used for grass, fodder and firewood harvesting, show a maximum basal area of 7.6 m^2/ha (Table 3.2, silvipastural plantation I). However, these plantations are not aimed at being fully tree dominated. A biomass study from 2006 showed that a degraded site can accumulate a considerable amount of 600 tonnes of green weight per hectare in 12 years, consisting of old-growth trees, planted and natural regeneration and massive growth of ground vegetation. Overall, 35 species (trees and shrubs) were recorded on this site. The return on investment works out at over 17% per annum, equivalent to a payback period of less than 5 years (final evaluation by ABI 2007), of which timber and non-timber use constituted a substantial part. In all 20 cases studied in 2006, focus-group discussion revealed that vegetation cover and the density of forests had increased. During that year, Himachal Pradesh also reported an increase in forest cover (forest cover inventory through satellite imagery by the National Remote Sensing Agency 2006).

Table 3.2 Total basal area (m^2/ha) and basal area per tree species in the sampled plantations (age between 10 and 12 years)

Tree species	Value	Silvipastural plantations					Protected plantations			
		I	II	III	IV	V	I	II	III	IV
Acacia catechu[a]	Mean	5.96	6.16	6.81	3.46	0.52	0.08	1.39	10.97	1.48
	SD	2.2	2.4	1.9	1.3	0.8	0.2	1.0	2.8	1.6
Toona ciliata	Mean					0.24		4.58	9.15	
	SD					0.3		1.9	2.5	
Albizia stipulata[a]	Mean	0.22				0.62	10.32	2.49	0.53	3.35
	SD	0.4				0.8	6.9	2.9	0.7	3.39
Dalbergia sissoo[a]	Mean					0.96	3.81		0.15	7.47
	SD					1.3	2.2		0.3	1.6
Bauhinia variegata	Mean					2.50		0.69	0.25	
	SD					1.6		1.3	0.5	
Miscellaneous	Mean	2.31	1.23	0.09	2.13	1.74	0.47	1.41	3.24	0.08
	SD	1.9	1.2	0.2	1.7	1.1	0.7	1.0	2.7	0.1
Total basal area	Mean	7.65	6.76	5.38	5.00	6.10	13.25	9.88	21.02	11.01
	SD	2.1	1.3	1.7	1.4	2.2	5.6	2.6	2.4	1.6
Nitrogen-fixing species (%)	Mean	74.9	80.7	98.8	63.8	31.5	93.8	31.0	47.4	99.2
	SD	21.0	20.8	2.3	25.5	16.4	10.7	26.4	8.5	1.5

SD standard deviation

[a]Nitrogen-fixing tree species. Their contribution to the total basal area is listed at the bottom of the table

3.4 Conclusions and the Way Forward

It is obvious that the project-influenced political participation of women and lower caste groups and leadership-capacity-building amongst women have led to social empowerment of the main target groups associated with forest management on the ground. Several village forest development committee (VFDC) members are elected to Panchayats (local governance bodies).

However, a general failure of the forest department in meeting the responsibilities set for it in “MoUs for PFM” has bred dissatisfaction amongst VFDCs, resulting in non-participation and inadequate management of several forest assets created by the project. Reframed policies, enacted laws and new regulations with reference to the Panchayati Raj Institutions (PRI) and for the Participatory Forest Management Regulations are redundant if they are not followed in practise by the key stakeholders. A radical mind-shift for better environmental governance can be achieved at the micro level, but its sustenance needs broader cooperation amongst practicing public investment actors in planning, implementation, monitoring and technical backstopping. Similarly, economic activities for women and other disadvantaged groups/the poor if designed on the basis of value addition of local surplus resources and integrated with the market promise local ownership and creativity for further innovations. Leadership-building amongst women and disadvantaged groups is the key for transforming local mindsets and leverage for demand-oriented support of the public service delivery system for promoting local development.

The PFM initiative in Changar can definitely benefit from proactive support of the forest department, creating the edifice for mainstreaming such a productive and protective concept fitting the sequel of environment, economy and equity. If decision-making on resource management is entrusted to local communities, their institutional capacities for management and benefit-sharing are built, and postintervention access to state and non-state service providers is guaranteed. Participation in all the local development work is established as an established principle across the erstwhile project region as several new public investment programmes have also adopted this mode of working and have reported success and greater ownership of local assets. Rehabilitation of degraded land and conservation on existing forest land is a huge task but as shown in Changar, it is manageable. For the sustainability of the impacts achieved, the following are indispensable (1) best practise lessons are disseminated; (2) lessons pertaining to a enabling framework need to be identified and mainstreamed at the policy and practise levels; (3) participatory approaches must be universally applied to get maximum and effective cooperation of a wide range of stakeholders and first and foremost the local communities and their institutions; (4) scientific forest management jargon has to be replaced with adaptive management, understandable and practical at local levels; (5) participatory monitoring systems must be effectively used to ensure that benefit-sharing is equitable, changes in the environment are registered and corrective measures are timely.

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Chapter 4
Operationalizing High-Conservation Values in Tropical Silviculture Through Access and Benefit Sharing

Konrad Uebelhör and Andreas Drews

Abstract The Convention on Biological Diversity (CBD) recognizes biodiversity as a global public good placed under the sovereignty of the provider country. The role of tropical forests for the conservation of biodiversity is well established. However, less attention is given to the contribution tropical silviculture can make in supporting the third objective of the CBD: "The fair and equitable sharing of the benefits arising out of the utilization of genetic resources" and related provisions regarding the use of traditional knowledge local and indigenous communities have about their resources. All tropical countries have ratified the CBD and are committed to implement this legally binding agreement. Outside protected areas efforts to implement the CBD in sustainably managed forests are still at the beginning. Considering the limited capacities in many tropical countries to manage their forests sustainably, only major challenges for tropical silviculture to implement the CBD and how provisions of the CBD can also further a modern silviculture are outlined.

4.1 The Convention on Biological Diversity: Important Issues for Tropical Silviculture

Over the last 16 years, since the Convention on Biological Diversity (CBD) entered into force, tropical silviculture and forest management in general did not take the CBD contents and processes much into consideration (Global Forest Coalition 2002). The role of the CBD is recognized for nature conservation but not for forest

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management, although the second and third objectives refer to sustainable use and related equity issues.

Only recently, there is growing recognition that the multilateral environmental governance framework on global public goods, such as biodiversity, is shaping national policy and legislative frameworks with corresponding impacts also on rather technical subjects such as tropical silviculture (TS). Because processes under the United Nations Forum on Forests (UNFF) yielded a voluntary framework only, the CBD provides the most important legally binding framework for biodiversity in forest ecosystems. Issues relevant for TS are among others addressed under the expanded Program of Work on Forest Biodiversity, the Addis Ababa Guidelines on sustainable use, the Akwé:kon Guidelines and in cross-cutting issues such as traditional knowledge, and the ecosystem approach (see Table 4.1).

While issues related to the second objective of the CBD ("sustainable use of biodiversity") are widely covered by present efforts toward sustainable forest management, agreed principles for it under the UNFF process and forest certification under the Forest Stewardship Council, the third objective of the CBD is hardly considered by forest managers. This third objective: "The fair and equitable sharing of the benefits arising out of the utilization of genetic resources," is presently receiving widespread attention due to the adoption of the Nagoya Protocol on "Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising out from their Utilization" (ABS Protocol).

In the following, therefore we will focus on and discuss the present state of ABS discussions and show where they could impact on tropical silviculture in two ways: informing management and contributing to the economics of sustainable forest management. We try to exemplify these issues for the case of *Prunus africana* and conclude with some remarks on the way ahead.

4.2 Regulatory Framework on ABS: Status Quo and Roadmap

The CBD reaffirms the sovereignty of countries over their biological resources (Preamble and Article 15) and that consequently the right to determine access to genetic resources rests with the national governments. The resulting challenge is to translate the lofty ABS vision into practicable mechanisms that will generate real benefits for countries that provide genetic resources.

A significant step was the adoption of the "Bonn Guidelines" on ABS at COP 6 in The Hague in March 2002. The "Bonn Guidelines" are intended to support the contracting parties and other relevant actors in shaping national policy, legislative and administrative frameworks on ABS, and/or negotiating bioprospecting projects in line with the principles of the CBD. Implementation of the guidelines is not binding; therefore, only few cases are known where the Bonn Guidelines were applied. Consequently, developing countries, representing the bulk of the remaining tropical forests, insisted on a legally binding international ABS regime. The impact of the ABS Protocol which will be legally binding for its Parties 90 days after 50

Table 4.1 Selected CBD programs and topics containing relevant aspects for tropical silviculture

CBD programs and topics	Relevant aspects for TS
Expanded program of work on forest biodiversity	Goal 1.1: Apply the ecosystem approach to the management of all types of forests Goal 1.2: Objective 1: Prevent introduction of invasive alien species Objective 4: Prevent and mitigate the adverse effects of forest fires and fire suppression Objective 5: Mimic natural disturbances such as fire, wind-throw, or floods to mitigate effects of loss of natural disturbance
Addis Ababa guidelines on sustainable use: principles	Practical principle 4: Adaptive management should be based on monitoring results with direct feedback on silvicultural decisions Practical principle 5: Sustainable use management goals and practices should avoid or minimize adverse impacts on ecosystem services, structure and functions as well as other components of ecosystems avoid clear cut in watersheds; use appropriate harvesting techniques; apply precautionary approach in management decisions Practical principle 7: Spatial and temporal scale of management should be compatible with the ecological and socioeconomic scales of the use and its impact Practical principle 9: An interdisciplinary, participatory approach should be applied at the appropriate levels of management and governance related to the use
Akwé:kon guidelines	Procedure and information requirements for the conduct of cultural, environmental and social impact assessments regarding developments proposed to take place on, or which are likely to impact on, sacred sites and on lands and waters traditionally occupied or used by indigenous and local communities
Ecosystem approach: 12 complementary and interlinked principles	Principle 3: Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems Principle 5: Conservation of ecosystem structure and functioning, to maintain ecosystem services, should be a priority target of the ecosystem approach Principle 6: Ecosystem must be managed within the limits of their functioning Principle 8: Recognizing the varying temporal scales and lag-effects that characterize ecosystem processes, objectives for ecosystem management should be set for the long-term
Traditional knowledge Program of work on Article 8(j) and related provisions	Respect, preserve and maintain the knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity, to promote their wider application with the approval and involvement of the holders of such knowledge, and encourage the equitable sharing of the benefits arising from the utilization of such knowledge

countries ratified the Protocol is almost impossible to assess as much will depend on the steps taken by Parties at national level for implementing the ABS Protocol.

Indigenous peoples and local communities often have a deep understanding of their environment and its ecology. This knowledge forms an important basis for the conservation of global biodiversity and for its sustainable use. Cultural and biological diversity are closely interlinked as expressed in the concept of biocultural diversity (Maffi 2008). Within the framework of the CBD, the contracting parties have committed themselves to respect and promote the use of traditional knowledge. Article 8(j) of the Convention states that "subject to its national legislation, [Parties will] respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity and promote their wider application..." The ABS Protocol specifies in Article 7 that "... each party shall take measures, as appropriate, with the aim of ensuring that traditioanl knowlegde associated with genetic resources that is held by indigenous and local communities is accessed with the prior and informed consent and involvement of the these indiegenous and local communites and that mutually agreed terms have been established" and further specifies in Article 12 that "... Parties shall ... take into consideration indigenous and local communities' customary laws, community protocols and procedures, as applicable, with respect to traditional knowledge associated with genetic resources". With these demands, access to traditional knowledge and related benefit sharing is closely interlinked with the corresponding obligations of Parties in Article 15 relating to genetic resources. ABS and prior informed consent (PIC) are thus key principles for the implementation of the CBD.

To promote these principles in the implementation process, a Working Group on Article 8(j) and Related Provisions of the CBD was established. A major outcome has been the elaboration of the "Akwé:Kon Guidelines" to ensure the active involvement of indigenous and local communities in the assessment of the cultural, environmental, and social impact of proposed developments on sacred sites and on lands and waters these communities have traditionally occupied. Moreover, guidance is provided on how to take into account traditional knowledge, innovations, and practices as part of the impact assessment processes. Although forest management plans are sometimes considered functionally equivalent to an environmental impact assessment, above issues were neglected in the past, have contributed to conflicts and are of increasing importance, considering that many of the remaining forest areas in the tropics are claimed by indigenous communities.

4.3 Potential Impacts of Traditional Knowledge and ABS on Silvicultural Practices

Objectives in tropical forest management in the past focused on timber. Starting in the 1990s under the pressure to demonstrate the economic feasibility of sustainable forest management, attention was drawn to other possible goods and services of tropical forests.

These other goods are categorized as nontimber forest products or nonwood forest products (see Chap. 10). Also, the concept of ecosystem services for human well-being as defined by the Millennium Ecosystem Assessment (2005) provides a more recent integrative framework for the multifunctionality of forest ecosystems. These goods and services must be reflected in the objectives of forest management. Forests for multipurpose objectives need a widened focus also of tropical silviculture.

Many of these goods are essential for the well-being not only of local and indigenous communities; furthermore, a high percentage of forests are located on land where native people have thrived for millennia (Mander 2008). Over the last 15–20 years, government involvement, also in the management of public forests, is receding. Decentralization, community-based management approaches all over the world, and recognition of indigenous, customary and community rights to forest lands and resources are increasing in number and scope (Sunderlin et al. 2008). These affirmations of local rights, also acknowledged for European forestry (Fabbio et al. 2003), create new scenarios to which tropical silviculture must adapt.

Local and indigenous knowledge impacts TS in various ways:

- Optimal solutions combine traditional ecological knowledge and scientific designs of sustainable management. Long histories of resource use in form of products derived directly from trees (medicine, edible fruits, seeds, poles) or non-woody plants (food, dyestuff, medicine) provide insights for utilization and cultivation techniques.
- Adaptive management, a prerequisite in times of climate change, can learn from local experience in dealing with environmental variability. Attentiveness to fluctuations and alterations in the natural milieu is an integral part of the life of local and indigenous communities, and remains of crucial cultural importance even in areas where lifestyles have been modified.
- Documentation of traditional knowledge contributes to safeguarding rights and secures benefits beyond local use.
- Recognition sustains cultural heritage and contributes to sustaining livelihoods.
- Facilitates comprehensive/holistic management because it involves not only technical practices, but also social institutions that organize technical practices (Wiersum 2000).

ABS impacts TS:

- Fair and equitable sharing of benefits introduces the concept of equity for deriving benefits from the use of genetic resources, which goes beyond fair trade.
- ABS contracts offer the possibility for additional benefits normally not connected with NTFP and remunerates the existence of a resource/derived product even if replaced by synthesized product.
- Valorizes the traditional knowledge related to a genetic resource.

An indicative lists of silvicultural aspects related to empirical knowledge of local and indigenous communities and due recognition to their rights is given in Table 4.2.

Table 4.2 Relation between silviculture and traditional knowledge (TK) and recognition of indigenous rights

Aspect of silviculture/management	Contribution of traditional knowledge/recognition of rights of indigenous peoples and local communities
Planning forest functions, selecting areas	Special use rights, forest areas for cultural and spiritual use (sacred forests), forests with special site conditions and rare biodiversity, identification of High Conservation Value Forests, especially types 5 and 6
Integrate biodiversity conservation needs	Identification of valuable tress, especially fruit trees, nontimber forest products associated with trees or special site conditions
Trees Nontimber forest products Medicinal plants, extracts Cultural, spiritual values	Prior informed consent in case of documenting TK related to medicinal plants, bioprospecting based on Bonn Guidelines/future international ABS regime, experience of sustainable harvesting volumes, also under changing conditions values connected to plant species, requiring special treatment or harvesting regulations
Requirements for regeneration	Empirical knowledge available from long-term observation (including plant–animal interactions)
Research and monitoring, especially silviculture in community based forestry	Collaborative research, indigenous and local practices to be combined with scientific insights, taking active part in monitoring facilitates adaptive management

There is no doubt that tropical silviculture incorporating the empirical knowledge of local people using existing indigenous forest and agroforestry management systems, on the one hand, and taking into consideration international frameworks as given by the CBD will be more complex. On the other hand, integrating indigenous silvicultural practices to solve forest management problems as perceived by local people will increase implementation. An example incorporating several of the above mentioned aspects is presented in the next paragraph.

4.4 The Case of *Prunus africana* an Example

The Afromontane hardwood tree *Prunus africana* (Hook.f) Kalkman (Rosaceae; African Cherry, Pygeum, Red Stinkwood) is a multiple-use tree species with local and international economic and medicinal value. It is an Afromontane forest tree measuring 30 m or more and is widely distributed in mainland Africa, Madagascar and the islands of Grand Comore, Sao-Tome and Fernando Po. Pygeum is one of 13 keystone species in high altitude montane mixed forest and has been important as an extremely valuable commercial hardwood commodity, as well as an important factor in traditional African medicine, where it was used as a remedy for urinary and bladder ailments, malaria, chest pain and fevers. In the 1960s, a liposoluble complex was discovered in the bark that was proven to be effective in treating prostate gland enlargement. Pygeum's principal biological activity is traced to a "phytosterol" compound known as beta-sitosterol. Pygeum became an important export to pharmaceutical companies worldwide, most notably to France, where it is sold under the brand name, *Tadenan*, and to Italy, where it is patented under the name of *Pygenil* for the treatment of benign prostatic hyperplasia (BPH). *Prunus* is traded in the form of dried bark and as bark extract.

Since the 1980s increasing commercial exploitation, habitat loss and unsustainable harvesting have led to a decline in *Prunus africana*, threatening conservation of its genetic diversity. For example in Cameroon sustainable harvesting schemes (the bark was to be removed from opposing quarters of the trunk – to avoid killing the trees through girdling – followed by periods of rest of 4–5 years) in place since the early 1970s, with only one company operating at Mount Oku, collapsed after 1985 when the Government issued 50 new permits for bark collection. Complete girdling became the norm or even complete felling of the trees, so that they could be easily stripped of their bark. Furthermore, with the new permits traditional authorities, which had developed a sophisticated system of taboos for managing watersheds and forest ecosystems, had no control over the outsiders who violated the local norm with impunity. Thus, the unsustainable harvest of *Prunus africana* contributed to the erosion of the resource conservation ethic that continues until today (Steward 2003).

As mentioned above, *Prunus* is widely distributed in montane Africa, but because its populations are isolated from each other they are genetically distinct. Since 1995, international trade in *Prunus africana* is regulated by the Convention on Trade in Endangered Species (CITES). The species is included in Appendix II, which stipulates that exports and imports have to be declared, with the exporting countries being required to demonstrate that their quotas have been set at levels that do not adversely affect the species. Despite the quota-based regulatory framework in place and over two decades of research, developing sustainable harvesting techniques and regeneration planting, the species faces major problems of over exploitation, illegal harvesting and degradation of its montane forest habitats. There is no current knowledge of the natural or planted stock of Pygeum, no monitoring system and no long-term management plans (details in Cunningham and Mbenkum 1993; Cunningham et al. 2002; Ingram and Nsawir 2007).

It is obvious that traditional African medicinal knowledge provided the lead for the modern medical use of Pygeum and its active ingredients, which as extracts and isolated compounds in modern formulations cure essentially the same ailments as African traditional healers do. Therefore, the obligations of CBD Articles 8(j) and 15 apply – at least for ongoing research on Pygeum. One could even argue that the continued "access" to the *Prunus africana* bark for producing the Pygeum extract constitutes an act of continued utilization of the traditional knowledge and the genetic resource, thus triggering continued benefit-sharing obligations beyond fair trade and sustainable harvesting regimes toward the resource providers and the traditional knowledge holders.

The Ugandan Government is using the ABS regime of the CBD based on the National Environment (Access to Genetic Resources and Benefit Sharing) Regulations of 2005 to create equity between the different stakeholders involved in Pygeum research: A material transfer agreement (MTA) between the National Forestry Resources Research Institute and Austrian research institutions specifies allowed uses, benefits to be shared and dispute settlement mechanisms. The MTA seeks to develop through research *Prunus africana* varieties suitable for commercial growing and harvesting by local communities in Uganda.

Whereas harvesting and selling of *Prunus africana* bark is still considered by most African countries as (bio-)trade with a commodity that needs not to take into account the ABS provisions of the CBD, the lack of certification schemes for sustainably harvested bark – as required by CITES – leads to a complete collapse of legal Pygeum trade and to a complete loss of income for the local harvesters as is the case in Cameroon. The Hoodia case is illustrating how such a situation can be solved: Hoodia export from South Africa requires a CITES permit, which is only issued, if the material has been cultivated by a farmer who is member of the Hoodia Growers Association, which guarantees that a 5% share of the market price of Hoodia flows to a Hoodia Trust Fund of the acknowledged regional indigenous peoples organization, whose traditional knowledge is the basis for the demand for Hoodia by consumers in industrialized countries.

4.5 Bridging the Gap: Conserving Global Biodiversity Interests, Satisfying Local Use Rights

Since the adoption of the CBD, there is an increased recognition of indigenous and local peoples in tropical forest management and an explicit support of their interests and rights is backed by the recently signed UN Declaration on the Rights of Indigenous Peoples (United Nations General Assembly 2007). Sustainable forest management can make a major contribution to achieving the goals of the CBD and to maintaining the biodiversity values that are of such great importance to the people who live in and around the forests.

Global interests to conserve biodiversity must avoid creating opportunity costs at the national and subnational level, especially in countries where due to poverty health, education and economic development are political priorities. Tropical silviculture in the twenty-first century has to incorporate processes and frameworks, which allow countries to defend their interests in a globalized world. On the other hand, the globalized economy and the opportunities arising from the need to find solutions for global problems offers also to tropical silviculture new possibilities. One of these possibilities is to considering the potential benefits arising from the utilization of genetic resources. Although the negotiations for an international regime on ABS will only be concluded in 2010, tropical silviculturists will be well advised to incorporate this issue into their management strategies.

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Chapter 5
Forest Concessions in Peru

Christian Großheim

Abstract Early this century, Peru changed its forest policy moving toward sustainable forest management. The new forest system is based on forest concessions focusing on the promotion of small- and medium-sized timber extractors within a competitive market. The concession design as far as legal and technical aspects are concerned can be considered as solid ground to work on. Regardless, implementation is still not concluded and most of the forest concessionaires face serious financial problems. The role of the Peruvian State as a promoter for sustainable forest management and facilitator to build up a competitive forest sector has not prospered by now. Consequently, the most important question to be answered is whether the changes to forest law and the subsequent introduction of forest concessions in Peru were an appropriate measure to accomplish the unfulfilled challenge to lay groundwork for a sustainable rural development in the Peruvian Amazon Basin.

5.1 Introduction

Peru, which has the second largest forested area in the Amazon Basin, significantly changed its forest policy early this century, and, despite major difficulties, begun moving toward sustainable forest management. The new forest system established (a) areas for permanent timber production granted by public bidding, (b) requirements for management plans for all forms of access to the resource, (c) the promotion of the integral use of forest resources, and (d) the creation of a fund to support the development of the forest sector (Melgarejo et al. 2006).

The current forest law (Act No. 27308) therefore favors a strategy oriented toward a competitive forest sector, incorporating principles of sustainability in timber extraction. Thus, the new way of access to forests, by concessions, places

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emphasis on the promotion of small- and medium-sized timber extractors within a competitive market (Galarza and La Serna 2005).

However, the enacted changes in forest law have not yet produced significant changes in the way the forest is being managed. Moreover, conflicts still remain between local stakeholders such as indigenous communities or other land owners and forest concessionaires as can be seen in Fig. 5.1.

The map exemplifies the area of conflict as a result of the spatial arrangement in the first place between forest concessions and indigenous communities. The potential of conflict is based on land tenure problems. Therefore, it occurs mainly where indigenous territories and forest concession are close to one another. This is mainly the case in the center parts and bottom of the map. In the second place, there is a conflict due to illegal logging, mainly Mahogany (*Swietenia macrophylla*) and Spanish Cedar (*Cedrela odorata*) but recently also Cumaru (*Coumarouna odorata*) from protected areas, which are poorly controlled by the Peruvian State and easily accessible by river. Furthermore, there are conflicts as a result of uncertain competences and lack of coordination between national authorities pertaining to the different forms of possible land use such as mining, oil drilling, or forest management, which still have not been resolved.

Consequently, the principal issue will be to figure out whether the changes to forest law and the subsequent introduction of forest concessions in Peru were an appropriate measure to accomplish the unfulfilled challenge to lay groundwork for a sustainable rural development in the Peruvian Amazon Basin.

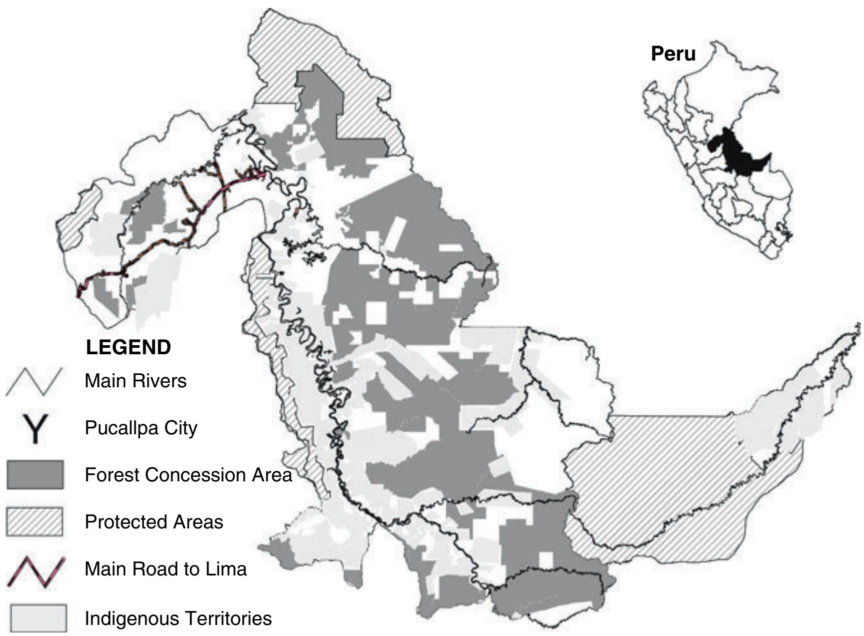

Fig. 5.1 Area of conflict due to land tenure and use in the department of Ucayali, Peru

5.2 The Peruvian Concession Design

The Constitution of Peru establishes, as a fundamental principal, natural resources and consequently forest resources as national patrimony. Therefore, the contractual arrangement regarding forest concessions of all types always includes the National Forest Authority as the public stakeholder; currently, the National Forest Authority is the Main Department of Forests and Wild Fauna within the Ministry of Agriculture. For supervision and evaluation purposes, Peru implemented the Government Agency for Wood Resources in Forests (OSINFOR Spanish abbreviation), and to assure its independence from the National Forest Authority the political responsibility was given to the Presidency of the Council of Ministers.

5.2.1 Legal Framework of the Concession Design

The legal basis for the forest concession granting process can be found in the current forest law. In general, two different forms of forest concessions exist (a) Forest concessions with a focus on timber extraction and (b) Forest concessions with a focus on nonwood forest products. The first group is composed of forests classified as suitable for permanent production and therefore considered as appropriate for timber production. The second group of forest concessions is composed of those focusing on nonwood forest products, eco-tourism or nature conservation, including environmental services. In all types of forest concessions, the duration of the contract is 40 years with the option of renewal. However, periodic revisions are conducted by OSINFOR to assure the fulfillment of the contract. Finally, the royalties to be paid for the forest concession depend on the offer delivered during the bidding process and have to be paid yearly. Up to the present, the National Forest Authority has organized two auction processes, one in the year 2002 and the other in 2003. In these occasions, 588 forest concessions were granted with a total size of more than 7 million ha. Concession size varies from 5,000 to 50,000 ha for one legal entity, but consortiums can be formed among forest concessionaires to gain access to bigger areas. That means, almost 10% of all Peruvian forests are conceded. By the end of the year 2007, 507 forest concessions were still valid, but not necessarily operative, the remaining 81 were in process to be resolved or have already been resolved (INRENA 2008), mainly due to illegal logging.

During the same time, the National Forest Authority granted 25 concessions for ecotouristic purposes, 15 with focus on nature conservation, and 934 for Brazil nut (*Bertholletia excelsa*) cultivation.

5.2.2 Technical Framework of the Concession Design

The technical aspect of the concession design is based on the elaboration and approval of relevant documents for short-term and long-term planning.

The long-term planning is composed of a General Forest Management Plan in which the concessionaire presents a general strategic planning framework for the use of the concession as well as a projection of the enterprise. Above all, the General Forest Management Plan should provide information about the current state of the concession site, its present productivity, and future potential with regards to a sustainable timber production.

The short-term planning refers to the Annual Operative Plan (POA, Spanish abbreviation). The POA gives information about the conducted inventory prior to timber extraction. In this inventory, the concessionaire determines, among others, trees to be harvested, trees for future harvests and seed trees. To differentiate between trees to be harvested and trees for future harvest, a "species-specific" target diameter is determined by the National Forest Authority. The area to be harvested is determined by area regulation and takes into account a minimum rotation time of 20 years. About 3 months before finishing a POA, the concessionaire has to present the plan for the next operative year and, most important, has to pay its royalties for the passed year. Without paying the set royalties, the new POA can be approved by Regional Forest Authority but timber transportation is restricted until the royalties are paid.

5.3 Limits to Sustainable Forest Management

Considering that Peru is a country with a high per-capita forest area but a relatively low per-capita income, its forest resources should be used primarily to guarantee a sustainable economic development in the less developed areas of the country and then for environmental protection. However, Peruvian forest concessionaires have serious problems to make their concessions profitable and therefore cannot use its potential to contribute to the economy, increase state revenues and social welfare. The forest sector's contribution to the gross domestic product (GDP) is only about 1%, which is relatively low compared to other Latin American countries such as Brazil (4.5%), Chile (3.6%), Paraguay (2.9%), and Ecuador (1.9%) (FAO 2004, 2006).

The Peruvian forest sector is composed, almost exclusively, of small and medium-sized forest enterprises (SMFE) as defined by Mayers (2006)[1] which, in the majority, are not competitive due to financial constraints as far as working capital and access to credits for investment are concerned. In addition, the forest sector suffers a lack of qualified personal along the whole chain of production. As a result, most of the timber extracted from forest concessions will be wasted during

[1]An SMFE is a business operation aimed at making a profit from forest-linked activities, employing 10–100 full-time employees, or with an annual turnover of US$ 10,000–US$ 30 million, or with an annual roundwood consumption of 3,000–20,000 m^3.

the transformation process or be exported as sawn wood to emerging markets in Mexico and China without major value added.

The promotional mechanisms from the Peruvian State to change the current situation are insufficient and therefore the dependence on foreign direct investment is very high. Moreover, the foreign direct investment is only available under two main conditions (a) a high rate of return on invested capital and (b) the risks (tenure, political stability among others) being controllable. It is the first condition, which is hardly compatible with the principles of sustainable forest management.

All the same, with the Forest and Wildlife Law of the year 2000 Peru laid theoretical groundwork to achieve sustainable forest management. In fact, before the year 2000, the timber extraction has been much more traditional and selective (Colán et al. 2006). Smith et al. (2003) illustrate it as follows: the most accessible timber is extracted from the authorized area, without making majors efforts to maintain the productivity of the forest. Even though this is a description of the situation before the year 2000, it also applies to the current situation in most forest concessions where the majority of forest concessionaires maintain a vision of extracting timber rather than managing forests. The National Forest Authority, so far, has failed to implement a forest policy reflecting the radical changes in the legal framework, by not paying attention to the long-term aspects of sustainable forest management and the forest concessionaires' real needs such as technical assistance and promotional credits. Although the Fund for the Promotion of Forest Development (FONDEBOSQUE, Spanish abbreviation) was created by the National Ministry of Agriculture with the purpose to contribute and facilitate the development and the financing of plans, projects, and activities geared toward sustainable forest development, a coherent concept to develop a competitive forest sector is not in sight. Besides institutional and financial limitations, there are also technical constraints to the sustainable forest management implementation. The most important one is that as a result of continuous timber extraction, rather than management, the Peruvian lowland rainforests are depleted. Accordingly, the yield in most forest operations does not exceed 10 m^3/ha from the annual plot. Transferred to the whole concession, applying a 20-year rotation, the extraction does not exceed 0.5 m^3/ha/year; in most concessions, it is far less. The abundance of commercial tree species such as Lupuna (*Ceiba pentandra*), mainly used in plywood production is limited and its long-term availability is at stake, also caused by an inappropriate target diameter of 64 cm. The same applies for Cumaru (*C. odorata*) used in parquet and floors, which recently is extracted at an accelerated rate that makes it unlikely to believe that its minimum exploitable diameter of 51 cm will help to assure its long-term availability. An ITTO-Project conducted by the National Agrarian University of La Molina (Lima, Peru) points out that 40% of mahogany (*S. macrophylla*) population is below the minimum exploitable diameter of 75 cm and 60% above. A situation which is not sustainable in the long run, thus increasing the minimum exploitable diameter should be considered (Lombardi I and Huerta 2007). All the same adjusting the minimum exploitable diameter for the different tree species is not the key to assure their long-term availability. In fact, it only slows down their rate of disappearance but cannot impulse their regeneration. Actually, the application of species-specific regeneration methods would be necessary to guarantee real sustainability for most of the highly valued tree species; however, the enrichment and subsequent management of logged-over stands with natural, highly valued species is, due to short-term perspectives and costs, not an option for the majority of forest concessionaires since forest law does not obligate them to do so. In contrast, the concessionaires rely on natural regeneration and the concept of seed trees. The result obtained from this *laissez faire* management is that the focus is even more on selective logging with all its collateral effects of high fix costs and extreme impacts on the residual stand. The weak willingness to adopt sustainable forest management is also reflected by the incipient level of certified forest management compared to Brazil and Bolivia. At the present exist only seven certificates with regards to forest management with a total area of 623,223 ha (2.5% of total area suitable for permanent timber production) and one more certificate for non-wood forest products with a total 30,386 ha (Source: http://www.fsc-info.org).

5.4 Conclusions for the Enforcement of Sustainable Forest Management

Considering that one objective for the forest concession's implementation, among others, had been to impulse the rural development, especially in the Peruvian Amazon Basin the evaluation of its success has to take into account the improvements aroused by the measures.

The Human Development Index is a widely recognized statistical measure to evaluate the state of development. The results can be observed in Table 5.1.

Generally spoken, there is a change for the better as far as the human development is concerned at least in the Peruvian Amazon regions; however, the economic situation has not improved contrasting the national development.

During the same time, the Peruvian timber production has increased year by year, which, in fact is a result of the timber extraction in the lowland rainforests of the Amazon. Table 5.2 exemplifies the amounts of national timber production, as well as timber exports and imports.

It may appear a contradiction that timber production increases from already depleted forests. However, due to the concession process, the total area in which timber extraction takes place has increased; consequently, the national timber production is rising.

Table 5.1 Information about human development (PNUD 2006)

Area	HDI 2000	HDI 2005	Per capita family income 2000 (US$[a])	Per capita family income 2005 (US$)[b]
Peru	0.620	0.598	100.84	126.70
Amazon	0.562	0.572	71.60	67.06

[a]The conversion rate of the Peruvian Nuevo Sol for 1 US Dollar for the year 2000 is 3.50
[b]The conversion rate of the Peruvian Nuevo Sol for 1 US Dollar for the year 2005 is 3.30

Table 5.2 Peruvian timber production, exports, and imports (INEI 2007)

Timber production (m^3)	2002	2004	2006
National	730,100	854,500	1,007,300
Exports	159,120	218,228	324,594
Imports	136,336	225,672	265,989
Self-supply (%)	80.72	73.82	71.96

As a final conclusion, one has to admit that the forest concession system, adopted by the Peruvian government at the beginning of the century, has not yet achieved its purpose since it has not contributed significantly to the Amazon Region's rural development. However, technically spoken the concession system is solid ground to improve sustainable forest management, and even more so if one considers the unsustainable forest use before the year 2000. Certainly, it also appears to be important to adjust the concession design in almost all its dimensions. Nearly all the recommendations in the literature to encourage sustainability in Peruvian forest concessions are very similar (Colán et al. 2006; FAO 2005; Galarza and La Serna 2005; Sabogal et al. 2006) and come to the following conclusion:

It is important to strengthen forest administration and the institutions responsible for investigation issues and forest extension to achieve sound forest management based on solid investigation. It also includes investigation in methods to apply species-specific assisted regeneration to improve forest stand's quality and long-term productivity.

Furthermore, it is indispensable to create specific incentives geared toward the promotion of technical assistance and a flexible credit design to encourage investment in modern technologies and a long-term vision of forest management. All these incentives must be incorporated in the National Forest Strategy to become part of a State's policy.

Last, but not least, it will be necessary to simplify the forest legislation to make it clearer and more coherent and, as a result, much more applicable for forest concessionaires.

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Part III
New Aspects in Tropical Silviculture

Chapter 6
Review
New Aspects in Tropical Silviculture

Michael Weber

Abstract During recent decades many environmental, social and legal conditions for sustainable management of forest resources have changed. Against this background, traditional and well-established silvicultural practices have to be reevaluated concerning their impact on conservation, biodiversity and environmental services and functions but also concerning their capability to ensure the ability of forest ecosystems to adapt to global environmental changes and to deliver the products and services that are requested by society. This chapter reviews exemplarily some of these new aspects, such as climate change, increased importance of goods and services other than timber and increased requirements for inventory, monitoring and planning as well as the consequences for silvicultural science and practice.

Keywords Climate mitigation · Adaptation · Biodiversity · Conservation · Nonwood forest products · Ecosystem services · Inventory

6.1 Introduction

For many decades the provision of pragmatic procedures to sustain a steady yield of wood from a few economically attractive species has been considered the main objective of tropical silviculture (Bertault et al. 1995). Accordingly, earlier books about silviculture in the tropics dealt mainly with traditional silvicultural systems in different regions and types of forests (e.g., Bruenig 1996; Dawkins and Philips 1998; Lamprecht 1986). In recent years, forest management and silviculture have faced substantial shifts in the conditions and requirements for sustainable use of

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forest resources, mainly owing to changes in societal and environmental circumstances on global and regional levels. A few of them should be mentioned here:

- Since the world population reached six billion in 1999, it has continued to grow by 83 million per year and will reach seven billion in 2011 (PRB 2009).
- The global forest area experienced a net loss of 5.2 million hectares per year in the period from 2000 to 2010, primary forests have been reduced by 40 million hectares since 2000 and the area designated primarily for productive purposes has decreased by 50 million hectares since 1990 (FAO 2010).
- Between 1995–1997 and 2004–2006 the number of undernourished people increased in all regions except Latin America and the Caribbean and is estimated to be 1.02 billion today (FAO 2009).
- The global atmospheric CO_2 concentration increased from the preindustrial level of 280 to 379 ppm in 2005. As a consequence of the accumulation of anthropogenic greenhouse gases (GHGs) in the atmosphere, the global mean surface temperature increased by 0.76°C from 1850–1899 to 2001–2005 (IPCC 2007a).
- Biodiversity, which provides an existential contribution to human welfare and livelihood, is decreasing at alarming rates (Millennium Ecosystem Assessment 2005). Between 10% and 50% of the higher taxonomic groups (mammals, birds, amphibians, conifers and cycads) are currently threatened with extinction. The main causes are habitat changes owing to land-use change, climate change, invasive alien species, overexploitation and pollution.

Human societies are concerned about these changes and their possible effects on the future availability of natural resources such as food, timber and freshwater. These concerns are reflected in several international agreements and regulations addressing better management of natural resources and improvement of the framework conditions. The most important political milestone toward international agreements on the sustainable management of forests resources was the United Nations Conference on Environment and Development (UNCED) in 1992 in Rio (Earth Summit), where several conventions and principles were adopted. The definitions, principles and standards formulated in these documents have many direct and indirect consequences for forest management and development of silvicultural concepts. Four of the most influential documents with direct influence on silviculture should be mentioned here exemplarily:

- The Convention on Biological Diversity (CBD 1992) commits the 168 signature states to support the three main objectives of the convention which are (1) the conservation of biological diversity, (2) the sustainable use of its components and (3) the fair and equitable sharing of the benefits arising from the utilization of genetic resources. As one concrete measure toward these objectives, Article 8 of the convention requires, for instance, that the countries prevent the introduction of alien species which threaten ecosystems, habitats or species and endeavor to provide the conditions needed for compatibility between present uses and the conservation of biological diversity and the sustainable use of its components. In 2000, the Convention on Biological Diversity adopted 12 principles, which also consider local developmental needs and stress the importance of landscape-scale issues in managing natural systems (COP5 decisions). In 2004, the Addis Ababa Principles and Guidelines for the Sustainable Use of Biodiversity were adopted, which also address a number of issues related to biodiversity in managed systems (CBD 2004).

- The Framework Convention on Climate Change (UNFCCC 1992) with its follow-up document, the Kyoto protocol, emphasizes the role of forests as important reservoirs for carbon and their function as sinks for or sources of GHGs, especially CO_2, as influenced by forest management. The Clean Development Mechanism (CDM) of the Kyoto protocol and the Reducing Emissions from Deforestation and Forest Degradation (REDD) mechanism, which is actually under discussion as a new mechanism in the post-Kyoto agreement, offer new options for tropical forest management which may allow several barriers to be overcome that are currently restricting sustainable use of tropical forests in many cases.
- Agenda 21 (UN 1992a), which defines the objectives to sustain the multiple roles and functions of all types of forests, forest lands and woodlands as well as to enhance the protection, sustainable management and conservation of all forests, and the greening of degraded areas, through forest rehabilitation, afforestation, reforestation and other rehabilitative means. Another aim is to ensure the ecological, economic, social and cultural role of forests. It also endorses the participatory role of local communities in decision-making in sustainable forest management.
- The "Non-legally Binding Authoritative Statement of Principles for a Global Consensus on the Management, Conservation and Sustainable Development of all Types of Forests" (Forest Principles) (UN 1992b), with its 17 points, reflects the global consensus on forests and applies to all types of forest, natural and planted, in all geographical regions and climatic zones. The principles encompass that states have the sovereign right to utilize, manage and develop their forests in accordance with their own development needs. Forest resources and forest lands should be sustainably managed to meet the social, economic, ecological, cultural and spiritual needs of present and future generations. Furthermore, the potential contribution of plantations of both indigenous and introduced species for the provision of fuelwood for household and industrial wood should be recognized as should the role of planted forests and permanent agricultural crops as sustainable and environmentally sound sources of renewable energy and industrial raw material, which shall be enhanced and promoted.

Although in the past environmental policies have been poorly considered and integrated in national policies, this has changed. Since UNCED in 1992, the countries have developed regional and international criteria and indicators that allow them to measure and monitor successes or failures in achieving sustainable forest management (Siry et al. 2005). Furthermore, a multitude of governmental and nongovernmental organizations with distinct but distinguished objectives is critically observing silvicultural activities in tropical forests so that failures or ecological sins will be most probably identified and made public, which – owing to public and economic pressure – can put forest enterprises at risk. This "watchdog" function of nongovernmental organizations is supported by the excellent new technology of high-resolution remote sensing.

In addition to the legal and societal framework conditions for forest management environmental conditions are also changing with different scales of intensity and time, resulting in direct or indirect effects on ecological processes and thus requirements for silvicultural treatment. Examples for such changes are the increasing CO_2 concentration in the atmosphere with the resulting global climate change (IPCC 2007a), loss of genetic resources owing to continuing loss of species and biodiversity (CBD 2004) and anthropogenic imissions into the ecosystems (Boy and Wilcke 2008; Boy et al. 2008; Fabian et al. 2005).

Consequently, silvicultural concepts and techniques cannot only be directed toward the objectives of the forest owners but must also consider the manifold societal demands as well as the ecological requirements arising from environmental changes. Therefore, they must be critically reviewed against the background of the new aspects and evaluated concerning their ability to meet the expected requirements. For instance, protectionist actors often accuse silviculturists of not having the technical knowledge required to satisfy the new and multiple demands without endangering the resource. Many forest managers, on the other hand, are convinced that the technical solutions to the challenges of sustainable forest management already exist and that it is only a matter of applying them appropriately (Bruenig and Poker 1989). However, often either the empirical evidence for the latter is lacking or the practices demonstrated by research are not applied by the timber companies, even if they are incorporated into forest management regulations (Embrapa/CIFOR 2000 cited in Olegário et al. 2008).

As a consequence of the above-mentioned aspects many realities for management of tropical forests have changed and it is important that silviculturists are aware of the changes and the resulting demands. The following sections will review in more detail some of the new aspects with paramount impact on forest management.

6.2 New Perspectives on Biodiversity and Conservation Management

The importance of biodiversity has been emphasized at different political levels through many international conventions and agreements promoting sustainable forest management (e.g., Montreal Process, Pan-European Process) as well as on commercial levels as part of certification schemes. For a long time the role of biodiversity has been recognized only in nature conservation but not in forest management (Chap. 4). Only recently, its recognition regarding rather technical aspects such as tropical silviculture has been growing. Similarly, for a long period of time the establishment of protected areas and parks has been considered the most promising option to ensure the continued existence of natural landscapes and ecosystems as well as of the genetic bases of tropical forests. Meanwhile, 12% of the world's forests (460 million hectares) are designated for conservation of biological diversity (FAO 2010). However, there is evidence that thousands of species are likely to disappear when biodiversity outside protected areas is neglected (Putz et al. 2000). Furthermore, there is also increasing opposition from forest dwellers to forest reserves that restrict the traditional access to and use of the natural forest resources. The disregarding of these rights was one reason to exclude "avoided deforestation" projects from the Kyoto protocol's CDM in the first commitment period. Hence, conservation through careful use, e.g., biodiversity-sensitive silviculture in managed forests, is increasingly accepted as a valuable option in successful conservation strategies (Putz et al. 2000; ITTO, IUCN 2009).

Against this background, many traditional and well-established silvicultural practices have to be reevaluated concerning their impact on biodiversity and conservation as well as on environmental services and functions. Because conservation of biodiversity in timber production forests depends highly on the way in which they are managed (Campos et al. 2001), much more attention has to be given to techniques that have been widely disregarded so far, e.g., proactive prevention measures (e.g., against fire) and postharvesting activities (Chap. 12). Several management options that contribute to the maintenance of biodiversity or to limiting negative effects of silvicultural interventions have also been presented by Putz et al. (2001): seed tree retention, modifying the seed beds for germination, mechanical scarification, herbicide treatment, enrichment planting, liberation thinning, vine cutting and mimicking natural disturbances. For many forest managers, except in dry forests, reduced-impact logging (RIL) has already become a standard as it does not only reduce incidental environmental damage and conserve biodiversity, it also contributes to improved sustained yield and carbon offset (Chap. 16). RIL largely contributed to an increase in the area of natural forests under sustainable management from one million hectares in 1988 (Poore et al. 1989) to about 36 million hectares by 2005 (ITTO 2006). Bawa and Seidler (1998) concluded in their review of natural forest management that the extent to which it can be expected to conserve biodiversity depends on several factors, including the initial structure of the forest, the scale and intensity of operations in space and time and the geographical configuration of managed forest areas within the matrix of undisturbed primary forest. They favor increased support for management of secondary forests, restoration of degraded lands, and plantation forestry as silvicultural means of retaining the diversity of tropical forest communities.

Carrying out silvicultural treatments in conjunction with logging not only reduces costs, it also reinforces the idea that logging can be silviculturally useful. As soil effects and damages to the remaining stand increase with logging intensities, silvicultural interventions should be moderate and concentrated in small areas, which will also help to reduce costs. Owing to progress in forest planning and monitoring (satellite images, digital landscape models) as well as in logging technology (e.g., cable or helicopter yarding) which allow, for example, better design, construction and maintenance of road networks, new means to limit damaging effects of silvicultural interventions are also available at the management level. As monitoring and controlling are more and more becoming integral parts of management, silviculturists will be enabled to steadily learn from successes and failures. For example, for biodiversity conservation in a comprehensive *Acacia mangium* plantation in Sarawak, a geographic information system was used to plan, implement, monitor and control activities in all planted compartments as part of an integrated plantation management system (ITTO, IUCN 2009). A landscape-scale map shows the mosaic pattern of natural and planted forests, and large and small conservation set-asides.

With the criteria for modern approaches to forest management considering conservation and biodiversity aspects as formulated in the certification schemes, for instance by the Forest Stewardship Council's Principles and Criteria for Forest Stewardship, new standards are set that cannot be disregarded by silviculturists even under noncertified conditions because markets will force them to be sensitive to the growing environmental concerns of global consumers and shareholders (Laurance 2008).

During the last two decades forest science has substantially improved the understanding of the effects of silvicultural activities in many forest ecosystems around the world (Bawa and Seidler 1998; de Graaf et al. 1999; Finkeldey and Ziehe 2004; Günter et al. 2008; Johns 1992; Kobayashi 1994; Lambert 1992; Pariona et al. 2003; Weber et al. 2008; Wilcke et al. 2009). Putz (Chap. 7) stresses also the need for clear differentiation among biodiversity aspects on different scales (landscape, ecosystems, community, species and genetic levels) as this has a direct influence on the evaluation of specific measures under different economic conditions and spatial levels. Because of better training and easier access to information, forest managers will be able to apply more sophisticated silvicultural concepts in the future.

6.3 Climate Change

Forest management can increase or decrease carbon flows between forests and the atmosphere and thus contribute to both acceleration and mitigation of atmospheric CO_2 accumulation, which is a dominant cause of climate change. Tropical forests account for about 40% of the carbon in the terrestrial biomass and 30–50% of terrestrial productivity (Dixon et al. 1994; Phillips et al. 1998; Watson et al. 2000). Forest destruction and degradation, predominantly in the tropics, accounts for about 17% of the total anthropogenic GHG emissions (IPCC 2007a). Because the role of forests as a sink for or source of carbon is a function of storage, accumulation or loss of biomass, any activity or management practice that changes the biomass in an area has direct effects on the carbon budget (Moura-Costa 1996). Putz et al. (2008) state that the potential for emission reductions through improved forest management is at least 10% of that obtainable by curbing tropical deforestation.

Consequently, management of tropical forests is high on the political agenda to mitigate climate change.

Furthermore, climate change is likely to have enormous impacts on tropical forests and their biodiversity (ITTO, IUCN 2009), which will also influence their capability to sequester carbon. All silvicultural activities must therefore also contribute to ensuring the potential of forests to adapt to the expected changes. Thus, climate change is a challenging new aspect for silviculture in tropical forests as it requires sustainable forest management to be reconciled with mitigation and adaptation management. To enable forest managers to cope with this situation, they need a clear decision on whether the management objective is to maintain carbon storage, to prevent carbon losses or emissions, or to actively remove CO_2 from the atmosphere. If a mixture of objectives is intended, a clear hierarchy of the different management targets is required.

In the following sections the main topics linked with the different management objectives are presented.

6.3.1 Management for Mitigation

The central means in managing forests for mitigation are maintaining high carbon stocks in natural forests, increasing the amount of carbon held within managed forests and reducing carbon losses due to management interventions.

The most effective measure to maintain high carbon stocks is to conserve standing forests, especially old-growth forests, which are high in carbon. However, this option disregards the multiple demands of the people living in or from the forests, or both. Thus, it is important to determine if a forest is going to be used for a mitigation purpose only or also for provision of products, income or services other than carbon. In the first case, the cost for the abandonment of utilization must be compensated for by the sale of the corresponding carbon credits. In the latter case, carbon credits could be a supplemental source of income if "additional" measures are applied or omitted that result in reduced emissions compared with "business as usual." In any case, sound information on trade-offs between management for products (timber, nonwood forest products, etc.) and for carbon is needed: first to decide what type and intensity of silvicultural treatment should be chosen, and second for the proper accounting of the corresponding carbon effects. Ashton et al. (2001a) claim that a unique set of silvicultural treatments should be tailored for the biophysical and social characteristics of each site to be able to effectively manage forests for carbon. However, it should be kept in mind that any extraction of timber from a forest will result in a reduction of its carbon stock, at least for a certain period of time.

There are various options for and intensities of silvicultural treatments that can be applied both before and after logging operations to promote increased carbon storage, or to improve the overall health and productivity of a forest (Table 6.1). However, all of them require sufficient knowledge of the current status of a stand

Table 6.1 Forest management options for mitigation greenhouse gas emissions and increased carbon storage

Management option		Potential other (noncarbon) effects
Objective: Increasing biomass in existing forests		
Increasing rotation length or intervention intervals	+	Increased ecological values
	–	Transient renunciation of income and products; increasing risk of catastrophic breakdown
Increasing stocking density (e.g., by reduced thinning or underplanting)	+	Higher biomass, better economic yields in the future
	–	Reduced stability, transient renunciation of income and products
Increasing amount of deadwood volume	+	Increased habitat values
	–	Transient renunciation of income and products
All aged forests with selective logging	+	Increased stability and habitat values; more continuous income
	–	Higher management skills required
Increasing forest growth (e.g., by fertilization or change of species composition)	+	Higher productivity, more and earlier income
	–	Loss of naturalness, eventually N_2O emissions
Postharvesting measures (e.g., weed control, planting)	+	Faster regeneration, better growth conditions
	–	Increased management costs
Enrichment planting in secondary forests	+	Higher proportion of valuable tree species, higher income
	–	High complexity, intensive management and high level of skills required
Objective: Preventing carbon losses		
Reduced impact logging (e.g., vine cutting, adapted machinery, careful planning and timber extraction)	+	Less erosion and compaction of soil, less damage to the remaining stand
	–	Higher costs and level of management skills required
Improving forest health and stability (e.g., appropriate species choice, mixed stands)	+	Higher stability, reduced losses due to pests and diseases, better adaptation to climate change; species with high wood density and quality which allow assembly of long-lived wood products
	–	Increased management costs
Prevention of forest fires	+	Increased safety and income
	–	Higher management costs, aggravated establishment of seedlings
Continuous cover forests	+	Increased stability and ecological value, flexible income
	–	Higher level of management skills required
Reduced ground preparation (fire, ploughing)	+	Reduced soil respiration, erosion and emission of greenhouse gases (N_2O, NO_x, NH_3, N_2)
	–	Aggravated establishment of seedlings
Correct timing and intensity of fertilization	+	Avoided emissions of N_2O, NO_x and NH_3, and eutrophication of watercourses
	–	Reduced growth
Adequate thinning	+	Reduced windthrow and anaerobic conditions owing to excessive thinning; reduced mortality owing to insufficient thinning
	–	Initial application of the thinning will result in a loss of standing aboveground carbon because of the reduction in the site's gross carbon volume
Improving pest control	+	Better productivity and reduced losses
	–	Pollution of soils or water

+ positive effects; – negative effects (according to IPCC 2003, changed) and the impacts of silvicultural interventions on carbon in biomass and soil as well as on other GHGs, e.g., CH_4 and N_2O, which might be released as a consequence of a measure applied. Cid-Liccardi and Kramer (2009) state that efforts to maximize carbon uptake and to reduce carbon losses need to be based on site dynamics and the application of silvicultural practices that are based on forest type and site characteristics. Furthermore, it is recommended that silviculture should consider landscape-scale effects as well because the maximum amount of carbon can be sequestered without compromising the long-term sustainability of the carbon storage when the stands are managed within a functional landscape matrix. This is also necessary to identify and consider zones of high carbon or other ecological values, e.g., for biodiversity or watersheds (Ashton 2003). Sist et al. (2003) stress the need to look also for harvesting impacts and trade-offs across larger forest landscapes, to expand RIL beyond silvicultural concepts and to ensure the maintenance of other forest goods and services. According to Ashton (2003), the landscape-scale template should reflect an integrated network of stands allocated to production and protection, with the focus on maximizing carbon storage within the landscape.

Table 6.1 provides an overview of management means for maintaining or increasing carbon stocks in existing forests or reducing management-induced losses.

Moura-Costa (1996) presented a table with estimated carbon offsets that can be achieved by several activities in the tropics over different time periods. The values range from 1–2 tons ha^{-1} (time frame 1 year) for soil improvement, to 20–30 tons (1 year) for soil protection, 250–350 tons for fire protection (undefined time frame), 35–150 tons (2 or 10 years) for RIL, 90–150 tons (30 years) for silvicultural treatments, 90–150 tons (20 years) for agroforestry, 150–280 tons (50–70 years) for enrichment planting with hardwoods and 100–200 tons (10–20 years) for plantations of fast-growing species. The offsets are calculated as the difference between the carbon accumulated in planted vs. untreated forest, or between conventional logging vs. RIL as the latter results in less damage to the residual stand, and more healthy stands, especially when aerial yarding by cable or helicopter is applied. Nabuurs and Mohren (1993) compared carbon-fixation rates of 16 global forest types under different management and found that management of selectively logged tropical rain forests is one of the most effective means. Ideal candidates for rehabilitation through enrichment planting with high wood density, long-lived canopy trees for carbon sequestration are, for instance, many logged-over and second-growth forests (Cid-Liccardi and Kramer 2009). Monospecific plantations with fast-growing species can sequester carbon in a very efficient way, but bear high risk of losses due to fires, pests or diseases.

When tropical forests are to be managed for carbon, treatments that may affect, expose or reduce soil carbon should be minimized and treatments that free growing space for desired species should be encouraged (Cid-Liccardi and Kramer 2009). Because the highest losses of carbon are caused by damage linked with badly planned or executed logging operations (Putz and Pinard 1993; Putz et al. 2008), reduction of avoidable logging damage to residual forest, soils and critical ecosystem processes through application of RIL must become a self-evident element in silvicultural planning (Pinard and Putz 1996). The means employed under RIL cover a multitude of activities ranging from identification of sensitive areas to careful planning of roads and skid trails, use of selective felling instead of clear felling, erosion control measures, preharvest liana cutting and training of loggers for directional felling or fire prevention and control. Comprehensive and detailed overviews of RIL and its effects and costs have been provided by Enters et al. (2002) and Dykstra and Heinrich (1996).

Despite the many undisputable benefits of RIL, with regard to CO_2 mitigation, its application requires great knowledge on the impact and timescales of each measure. For instance, liana removal affects carbon storage because it increases the light available to trees and reduces competition, allowing growth rates and carbon to increase in the stand (Wadsworth and Zweede 2006; Keller et al. 2007; Zarin et al. 2007). However, the positive benefits of liana removal persist only for a few years, and repeated treatments are required over a cutting cycle (Peña-Claros et al. 2008a, b). Other measures, such as prescribed burning and soil scarification for exposure of mineral soil to encourage regeneration, can even have negative effects on carbon storage if done inappropriately because they could reduce soil organic matter. On the other hand, carbon emissions linked with site preparation may be compensated for by a higher increment of planted trees compared with untreated trees (Moura-Costa 1996). Sist et al. (2003) concluded from their studies in mixed dipterocarp forests in Borneo that RIL techniques cannot guarantee silvicultural sustainability when solely based on a minimum-diameter cutting limit. They suggest three silvicultural rules to ensure sustainable management (1) to keep a minimum distance between stumps of about 40 m, (2) to ensure there are only single tree gaps using directional felling and (3) to harvest only stems with 60–100 cm diameter at breast height.

However, besides the mitigation potential of forest management described, it should be mentioned that the key opportunity in tropical regions is the reduction of carbon emissions from deforestation and degradation.

6.3.2 Management for Adaptation of Forests to Climate Change

When managing tropical forests for mitigation, one has to consider that the effects of climate change, e.g., regional drying and warming (Salati and Nobre 1991), increasing frequency and intensity of extreme weather events (IPCC 2007a), possible intensification of El Niño phenomena (Sun et al. 2004), changes in phenological relations, or loss of biodiversity (IPCC 2007b), may limit or even reverse the sink effects of the forests. Especially the huge carbon stocks in mature tropical forests may be vulnerable to disturbances induced by climate change. To reduce the risk of such carbon losses, silvicultural concepts and strategies must strive to maintain the ability of forests to adapt to climate change. This may require alterations of established management regimes. Forest management will need to be highly adaptive, which will require good and up-to-date information about what is happening in the forest (ITTO, IUCN 2009). However, as climate change in the tropics is still associated with many uncertainties and no empirical evidence is available, it must be assumed that silvicultural adaptation strategies will not be substantially different from other risk-reducing strategies, and thus should strive to reduce vulnerability to pests, diseases and abiotic damage by ensuring forest health and ecosystem diversity as a prerequisite for the ability (resilience) of the forest to recover after disturbances (Adger et al. 2005; Van Bodegom et al. 2009). Cid-Liccardi and Kramer (2009), for instance, state that in the long term a species-rich forests with a diverse vertical structure will be more resilient to catastrophic disturbances, and therefore to carbon loss. On the basis of Smith et al. (1997) and Ashton et al. (2001a, b), they conclude that these objectives can be most likely achieved by regeneration via a shelterwood system and its variations around structural retention and age class. An excellent overview of silvicultural practices to maintain and enhance the adaptive capacity of natural and planted tropical forests was provided by Guariguata et al. (2008).

It has to be mentioned as well that some adaptation measures, e.g., regenerating old, mature stands or understory removal for fire prevention, may conflict with the objective of maximizing carbon storage as the highest stocks are achieved in old forests with high biomass, which are considered to be more vulnerable to disturbances. Management for adaptation must also consider the landscape scale, for instance by making an effort to maintain and improve the connectivity between forest fragments or habitats to ensure minimum viable populations. Furthermore, it requires intensive monitoring to quickly detect and tackle outbreaks of pests and diseases, and changes in the dynamics or composition of the forests (Van Bodegom et al. 2009).

Consequently, Phillips et al. (1998) called for a dedicated large network of permanent biomass plots to obtain insight into the future role of tropical forests in the carbon cycle. The incorporation of multiple variables into ecosystem and forest models for tropical forests can also contribute to a better understanding and consequently better management of the carbon fluxes and climate change effects (Nightingale et al. 2004). For this purpose, intensive research on the positive and negative feedbacks of possible impacts of climate change, such as droughts, windstorms, biotic pathogens, and fires, on forest dynamics and carbon stocks is still required (Meister and Ashton 2009).

6.4 Increasing Importance of Goods and Services Other Than Timber

6.4.1 Nonwood Forest Products

Gathering forest resources other than wood, such as berries, mushrooms, fruits, herbs, and bush meat, to provide food, energy, or construction material has been practiced by humans since historical times. However, since the 1980s NWFPs have achieved renewed and increased interest on a global scale as an additional income-generating option (García-Fernández et al. 2008; Chap. 10). Although information from many countries is still missing and the true value of subsistence use is rarely captured, the value of NWFP removals in 2005 is reported to amount to about US $18.5 billion (FAO 2010). Consequently, integration of timber and nontimber forest uses is considered to offer new opportunities for subsistence and market economies of rural communities to enhance their well-being and to reduce the risk of losses due to a more diverse portfolio of assets. NWFPs can be a significant complementary asset to timber production, particularly in low-wage, rural economies. Several studies have shown that in combination, net present values can double or even triple, with NWFPs often providing most of the income (Godoy et al. 1993; Boot and Gullison 1995; Ashton et al. 2001b). Peters et al. (1989) stated that the potential long-term economic returns from forests managed for NWFPs are greater than the returns from timber or forest conversion to agriculture. Thus, NWFP management has also caught the attention of conservationists as a means of ensuring forest conservation and as alternative to conversion (Hiremath 2004).

Including NWFPs in diversified forest management plans is increasingly used in sustainable forest management to offset the costs of RIL. However, skeptics question the extent to which the economic returns from NWFPs are sufficient to compensate for the costs of applying RIL (Barreto et al. 1998; Pears et al. 2003) and corresponding silvicultural practices, e.g., enrichment planting or liberation thinning, needed for sustaining timber production over the long term (Schulze 2008; Wadsworth and Zweede 2006).

For silviculture, managing different types of products (timber and NWFPs) is a new challenge as it requires different knowledge and skill sets which are still segregated among different forest users. To be able to cope with this challenge, managers must extend their silvicultural expertise from forest management to agroforestry and farming practices. Consequently, such aspects should be included in modern tropical silvicultural education and training. However, NWFPs are still predominantly treated in relative isolation (Lawrence 2003 cited in Guariguata et al. 2010). Although Whitmore (1990) stated that incorporation of NWFPs with timber extraction was common until the middle of the last century, data on the production and reproduction of NWFPs within timber management as well as integration of silvicultural interventions for NWFP species in overall forest management are rare or nonexistent (Panayotou and Ashton 1993; Chap. 10).

To be able to sustainably integrate nonwood forest species management in silvicultural concepts, their ecological and productive characteristics must be explored and tested. Only little is known about NWFP harvesting impacts and the available information seems not consistent. Ticktin (2004) reviewed 70 studies that quantified the ecological effects of harvesting NWFPs from plants and concluded that the tolerance of NWFP species to harvesting varies according to life history, the part of plant that is harvested, environmental conditions and the management practices used. Moreover, the impact can vary from the level of genes to individuals, populations, communities and ecosystems (Hall and Bawa 1993; Ticktin 2004). Although a close relation to logging intensity has been established, further assessments of the diverse harvesting impacts are needed. Furthermore, specific silvicultural systems for NWFPs are essential for sustainable management (Hall and Bawa 1993). For instance, to promote population persistence of specific NWFPs, sparing of individuals, size restrictions, overstory light management, thinning, transplantation, coppicing and replanting of plant parts can be adequate management practices (Ticktin 2004).

Guariguata et al. (2010) emphasized six topics to be considered as key components of sustainable forest management integrating NWFPs (1) integration of NWFPs in inventories (yields, mapping), (2) ecology and silviculture for timber and NWFPs (effects of logging intensity and corresponding changes in forest structure, radiation, soil), (3) conflicts in the use of multipurpose trees, (4) wildlife conservation and use (habitat changes, increased access), (5) tenure and access rights (types of rights, multiple right holders) and (6) product certification (ecological and social constraints). They concluded that compatible management has to be inherently multifactorial and context-dependent.

Sustainable extraction and management of timber and NWFPs is influenced not only by silviculture but also by many other factors. Guariguata et al. (2010) published an indicative list of factors, with, e.g., habitat overlap, length of rotation cycles, property rights, local governance, and market chains. Consequently, barriers for successful adoption of silvicultural practices are rarely just technical in nature but depend on diverse perspectives of individuals (Walters et al. 2005). As some user demands may conflict with others, participation of the main stakeholders in forest planning is required to balance all expectations. However, this will make planning even more complex. To make integrated management of timber and NWFPs more feasible, attractive and competitive for other land users and thus to avoid forest conversion, it may also be necessary to increase the number of species and products utilized from tropical forests. Actually, only about 150 NWFPs are of major significance in international trade although approximately some 4,000 botanicals species enter international markets (Chap. 10). Especially for small-scale operations, integration of a wide array of goods and services would be beneficial (Campos et al. 2001). As a high proportion of NWFP goods are still collected from the wild, domestication measures will have to be applied for many species to increase the efficiency of their management (Bhattacharya et al. 2008). However, one should proceed with caution because highly valued NWFPs may increase the risk of gradual conversion of forests into tree-gardens or agroforestry systems (Michon et al. 2007) or the setting up of plantations. For more details on the silvicultural options of NWFP management see Chap. 10.

Several promoters of multiple-use forest management emphasize that by incorporating many forest goods and services and by considering the interests of multiple stakeholders, a social and financial advantage can be gained over timber-dominated models (Ashton et al. 2001a; Campos et al. 2001; Hiremath 2004; Wang and Wilson 2007). According to the FAO (2010), the area designated primarily for productive purposes has already decreased by more than 50 million hectares since 1990, whereas that for multiple uses has increased by 10 million hectares in the same period.

6.4.2 Biofuels

In many tropical countries fuelwoods traditionally play an important role. For instance, in Africa fuelwood is the dominant source of heat energy in rural households, where it is used for cooking, heating and steam raising (Chap. 24). However, since the use of fossil fuel is causing 57% of the total GHG emissions (IPCC 2007a), switching from fossil fuels to biomass fuels will play an increasing role in strategies to reduce GHG emissions. In this context the tropics receive special attention because of the high production rates of up to 40 tons ha^{-1} year^{-1} which can be achieved in this region (Chap. 9). Although a large proportion of "modern" fuelwood comes from agricultural areas or from nonwoody biomass, the situation also provides new opportunities and challenges for silviculture.

On land that has not been available for forestry so far (e.g., grassland, fallow land, marginal and abandoned land) new forests could be established with the sole objective of biomass production. Grainger (1990) considered fuelwood production in a fully sustainable cyclic system as the most promising option for carbon sequestration in the long run. Consequently, silviculture must develop concepts to produce biomass for energy purposes under different conditions in an ecologically, economically and socially sustainable way.

One promising option is biomass production in short-rotation forestry. Although short-rotation forestry has a long history, e.g., coppice as an ancient silvicultural system to grow small wood for fuel, charcoal or fencing, there is a need to review the concepts against the background of the new practices and technologies in production and processing. Essential silvicultural features in this context are, e.g., adequate stocking densities to achieve fast site occupancy and high mean annual increments, tree breeding programs, use of fertilizers, integrated pest and disease management, and mechanized harvesting (Mead 2005).

Another important aspect is the identification of suitable species for different types of land and environmental conditions, which are adapted to the sites and are easy to establish, show good growth rates and coppicing ability and produce high-calorific wood (NAS 1980; Nair 1993; Mead 2001, 2005). Moreover, good understanding and monitoring of the nutrient cycles, especially if whole tree harvesting will be applied, is required. Silviculture has to explore and develop ways in which biofuels can be produced and harvested in the context of larger landscapes and all forest resources without violating the ecology or the demands of local societies, especially for food production.

Biofuels can also offer new opportunities for traditional timber management, as residues from harvesting that would otherwise be left to decay could be considered as valuable by-products to increase the environmental and financial benefits of bioenergy (Hwan Ok Ma 2007). If biomass production is additional and sustainable, it can also generate supplemental income by generating carbon credits. However, it has to be considered that harvesting of logging residues has silvicultural implications, apart from soil fertility concerns (Asikainen et al. 2002).

More detailed information on silvicultural aspects of bioenergy production is provided by Onyekwelu and Fuwape (Chaps. 9 and 24).

6.4.3 Ecosystem Services

Apart from the delivery of marketable goods such as timber, NWFPs and biofuels, natural and managed forest ecosystems also provide many services that are in demand from different interest groups but can hardly be merchandized directly by the forest owners on local markets. The Millennium Ecosystem Assessment (2005) distinguished among regulating (air quality, climate, water), cultural (recreation, spiritual enrichment), and supporting (nutrient or water cycling, photosynthesis) services. Some of these services are scale-dependent; for instance CO_2 sequestration is more effective at a global scale, whereas maintenance of water quality or erosion protection is more locally or regionally relevant. Some services can be ensured by the pure existence of a forest (e.g., photosynthesis), whereas others rely on or can be improved by silvicultural activities (e.g., recreation).

During recent decades a shift from monostructured silviculture focussed on procedures to achieve high wood yields for a few economically attractive species toward multiple-use forest management considering timber, NWFPs, and environmental services has been observed (Bertault et al. 1995). However, owing to the persistent deforestation and degradation together with the steadily rising demand on natural resources, it is becoming critical to embrace all requirements in management of tropical forests. Thus, there is a growing need to promote concepts for sustainable extractive uses alongside the persistence of ecosystem services, especially biodiversity (Millennium Ecosystem Assessment 2005).

For a long time it was assumed that provision of forest services follows in the wake of timber production (Glück 1987). This meant that the production of timber also encompasses other objectives, such as sustaining the function and dynamics of ecosystems, maintaining ecosystem diversity and resilience, and provision of various ecosystem services of value to mankind (Coates and Burton 1997). However, several forest management operations, especially in the tropics, affect ecosystem services (Olegário et al. 2008) or are not compatible with specific services. Because many environmental services are considered as public goods, they were taken for granted. The direct costs of the forest owners related to the provision of environmental services, e.g., special management interventions required or higher costs of forest operations due to management restrictions, were a major barrier for better consideration in silvicultural planning and practice.

With increased awareness about the environmental services provided by forests and the perceived scarcity, the appreciation of beneficial forestry activities and the readiness to pay for them have improved. Thus, payments for environmental services (PES) emerged as an innovative means to compensate service providers for their expenses to maintain or achieve a requested level of environmental service provision. The most dynamic and advanced PES types so far are those for carbon sequestration, biodiversity conservation, watershed protection, and landscape beauty (García-Fernández et al. 2008). PES offer a great chance to facilitate sustainable forest management, or to make it competitive with other land uses and thus to avoid further forest destruction and loss of the forest environmental services. A study on how this can be achieved practically was presented by Knoke et al. (2009) using an example of Andean Ecuador. Despite progress in recent years, there are still many methodological problems, especially regarding the allocation of concrete economic values to specific services (Pirard and Karsenty 2009). To ensure that the benefits of managing a forest area in a sustainable manner including environmental services will really be captured by the forest owners, it is necessary to fill in the existing gaps of ecological knowledge about the effects of specific silvicultural techniques on the provision of environmental services. Thus, there is an urgent need for corresponding research that also involves monetary evaluation of silvicultural aspects. For example, Smith and Applegate (2004) stated that both opportunity costs of shifting from conventional logging to improved practices and the long-term carbon and biodiversity benefits of improved forest management have been underestimated.

For silvicultural planning it is essential to know which special environmental services are in demand and if they should be provided by integrated management of a total given forest area or if segregation into different districts is possible. This implies that the expected environmental services as well as the different stakeholders must be clearly identified and included in management planning. Last but not least, it has to be clarified if the compensation will be paid on an individual basis based on the application or omission of specific procedures or as a flat rate which is independent of the particular costs. Anyway, production of more diverse forest values requires the consideration of the fine-scale variability within forest stands and a better understanding of the spatial and temporal responses of forest ecosystems to manipulation (Coates and Burton 1997). Silviculture has to provide procedures that meet the objectives of timber production without compromising environmental services.

Depending on the spatial scale, silvicultural planning must involve the identification of sensitive areas or structures, e.g., to avoid downstream effects on soil and water.

Box 1: Gap-Based Models as a Tool to Understand Fine-Scale Ecosystem Responses to Silvicultural Manipulations

Coates and Burton (1997) proposed employing gap-based approaches to study stand responses to silvicultural manipulations that “(1) aids development of cutting prescriptions that maintain functional mature or old-growth conditions; (2) refines and extends our understanding of how biological structures, organisms and ecosystem processes are affected by fine-scale variation within stands…” Such studies could help to identify optimal gap sizes, distribution and frequencies that are adapted to different stand structures and site conditions. Table 6.2 provides an overview of the characteristics of tropical and subtropical ecosystems that are related to different gap attributes.

(continued)

Table 6.2 Examples of characteristics of tropical and subtropical ecosystems as affected by gap attributes

Ecosystem characteristic	Gap attribute					References
	Presence	Size	Position	Age	Substrate	
Tree species establishment/density/composition						
Tropical moist forest, Panama	–	+	–	–	+	Putz (1983)
Tropical moist forest, Panama	–	+	–	+	–	Brokaw (1985, 1987)
Tropical cloud forest, Costa Rica	+	0	–	0	–	Lawton and Putz (1988)
Amazonian forests, Venezuela	0	0	0	–	–	Uhl et al. (1988)
Subtropical broadleaved, India	+	+	–	–	–	Barik et al. (1992)
Tropical cloud forest, Ecuador	+	+	–	–	+	Kuptz et al. (2010)
Birds						
Richness/composition	+	–	–	–	–	Schemske and Brokaw (1981)
Guilds/species	+	+	–	–	–	Levey (1988)
Small mammals						
Bat species	+	–	–	0	–	Crome and Richards (1988)
Insects						
Level of attack	+	–	–	–	–	Harrison (1987)
Abundance	+	–	–	–	–	Shelly (1988)
Population structure of moths	+	+	–	–	–	Günter et al. (2008)
Epiphytes	+	–	–	–	–	Günter et al. (2008)
Irradiance						
Tropical forest, Costa Rica	+	–	+	–	–	Denslow et al. (1990)
Tropical cloud forest, Costa Rica	–	+	–	–	–	Lawton and Putz (1988)
Subtropical broadleaved, India	+	+	–	–	–	Barik et al. (1992)
Tropical evergreen rain forest, Mexico	–	+	–	+	–	Dirzo et al. (1992)
Tropical cloud forest, Ecuador	+	+	–	–	–	Kuptz et al. (2010)
Climate parameters						
Air temperature (mean/min./max.)	0	0	–	–	–	Barik et al. (1992)
Humidity	+	0	–	–	–	Barik et al. (1992)
Soil parameters						
Surface soil moisture	+	+	–	–	–	Barik et al. (1992)
Soil temperature (mean/min./max.)	0	0	–	–	–	Barik et al. (1992)
Soil respiration	+	–	–	–	–	Günter et al. (2008)
Soil nutrient availability	0	0	0	–	–	Uhl et al. (1988)
Nutrient levels	+	–	–	–	–	Wilcke et al. (2009)
Phosphorus concentration canopy zone	0	–	–	–	–	Vitousek and Denslow (1986)
Phosphorus concentration root-throw zone	+	–	+	–	–	
Litter thickness/decomposition	+	+	–	–	–	Barik et al. (1992)
Fine root biomass	+	–	0	–	–	Sanford (1989)
	+	+	–	–	–	Sanford (1990)

Based on Coates and Burton (1997), extended
+ significant effect; 0 no effect; – not tested

Adapted cutting cycles and directional felling should also be minimum requirements. For the acquisition of PES, it will be necessary to clearly identify and assess the supplementary direct and indirect costs of activities that go beyond the "normal" management for timber production, for instance the application of RIL techniques or postharvesting practices. Because biodiversity conservation is one of the environmental services most demanded, for PES forest managers must be capable of monitoring changes in both biodiversity and society's requirements for biodiversity and be capable of adapting their management accordingly (ITTO, IUCN 2009). Thus, there is high need for consideration of new subjects in silvicultural research, education and technical training, e.g., comprehensive land-use planning, efficient inventory and monitoring methods, participation of the local population, better integration of silviculture and harvesting, better consideration of low-abundance/low-value species, renouncement of the use of pesticides and herbicides, habitat management, and conservation (de Haan 2008). All these aspects should be linked with economic considerations related to the marketing of the resulting effects as ecosystem services.

6.4.4 Ecotourism

Many tropical forests play an important role as a resource of biodiversity and refuge for endangered species as well as for indigenous communities. This did not only increase their importance for conservation and recreation of local people but has also attracted the interest of ecotourism. This implies that as far as managed forests are concerned all silvicultural interventions have to be applied in a very sensitive way, which means that the general character of the stands must be maintained as naturally, biodiverse and attractively as possible, whereas obvious logging damage will not be compatible with ecotourism. In the context of ecotourism, much more attention must also be paid to landscape effects of silviculture and to the compatibility with the presence and abundance of attractive animals. Forests have to be considered as formative elements embedded in human-populated landscapes. This requires local people becoming involved in planning in participatory procedures. Adequate compensation would also allow specific activities to be applied, such as managing ornamental species, creation of apertures and outlooks, the creation of scenic roads or hiking corridors, and increasing the rotation length, that increase the value of a forest for recreation and ecotourism. In managed forests the income that can be generated by ecotourism has to be balanced against possible losses or increased costs due to, e.g., reduced removal volumes or growth rates, maintenance of infrastructure, or security aspects. Nevertheless several studies have shown that ecotourism can provide significant increases in the livelihood and purchasing power in rural communities (Wunder 1999; Tobias and Mendelsohn 1991).

6.5 Increased Requirements for Inventory, Monitoring and Planning

Adequate consideration of the specified aspects in silvicultural decision-making and controlling entails an increasing demand for detailed information. Today forest managers are expected to provide empirical evidence that their forests are well managed or that a certain intervention does not harm ecological processes or ecosystem services. Thus, enhanced sustainable forest management requires credible verification and reporting (Siry et al. 2005). For many products and services a continuous chain of custody of ecological and social management standards is required today for successful marketing or approval under certain programs. As a consequence, the global forest area certified by one of the major certification schemes increased from 121 million hectares in 2002 to approximately 270 million hectares in 2006 (FAO/UNECE 2006).

To be able to fulfill the multiple demands and to provide the information requested by legal institutions, stakeholders, international markets or certifiers, efficient tools for inventory and monitoring of ecological, economic and social conditions must be developed. For instance, changes induced by specific silvicultural interventions have to be analyzed and documented at different temporal (from short term to long term) and spatial (from stand to landscape) scales. Under sustainable multifunctional forest management it is therefore necessary to include explicit spatial structures and objectives in planning and monitoring (Baskent and Keles 2005). Many forest functions are directly related to the characteristics of a forest, e.g., size, structure and shape. Furthermore, treatment in one forest unit may also influence adjacent units. A clear-cut, for instance, can increase the risk of wind damage, sunburn, or erosion in the neighborhood (Öhman 2001). Disregard of spatial characteristics can therefore result in lower yields, reduced water quality or habitat disruption.

The specified requirements are reflected in new methods and technologies, especially in the field of remote sensing, GPS and georeferencing. These new technologies enable forest managers to better keep track of the described development. Owing to progress in remote sensing, high-resolution images with comprehensive information on stand composition and structures, terrain, environmental conditions and soil are available. Geographic information systems allow direct interlinking of this information and characterization and analysis of spatial relationships among explicit management units at different scales (e.g., susceptibility for disturbances, or connectivity). Furthermore, growth and yield models (Vanclay 1994, 1995; Clark and Clark 1999; Huth and Ditzer 2000; Peng 2000; Ong and Kleine 1996), process-based ecosystem models (Mäkelä et al. 2000; Miehle et al. 2010), new bioeconomic models (Knoke et al. 2009) and management-oriented models to evaluate the effects of different management strategies (Huth and Ditzer 2001; Kammesheidt et al. 2002; Schelhaas et al. 2004) can be used to manage all information in an integrated way. Many decision-support models have been developed as well (Battaglia et al. 2004; FORSYS 2010; Iliadis 2005) to help forest managers to better structure and analyze the complex and sometimes chaotic conditions and to make decisions based on sound facts and the most recent scientific knowledge in a reproducible way. However, managers are still hesitating to really use them (Whyte 1996). Last but not least, negotiation-support models are available that assist the communication of the complex and sometimes conflicting aspects with the many actors and stakeholders and thus avoid conflicts, identify common demands and views or rank multiple demands according to priorities (Van Noordwijk et al. 2001).

All the instruments described offer new opportunities to include the improved knowledge of ecosystem functioning in silvicultural planning and to make it more spatially explicit toward a kind of "precision forestry" (Farnham 2001). This makes it, for instance, feasible to give much better consideration to functional characteristics of tree species, keystone species, efforts to connect habitats and landscapes, energy and material flows induced by interventions, or interactions (e.g., complementarity) among different ecosystem components, which may lead to higher productivity or less risk. High-resolution monitoring of landscapes enables silviculturists to quickly detect and tackle outbreaks of pests and diseases, and to improve effective fire management. Nevertheless, to be able to assess the temporal stability of ecosystems under different silvicultural treatments and to identify feedbacks, many more long-term silvicultural experiments are needed.

6.6 Conclusions

Although timber production has been the dominant function of forests in the past, in recent years a more multifunctional and balanced view of forest management has become widely accepted. Recreation, health and well-being, biological diversity, mitigation of climate change and adaptation to it have been increasingly recognized as integral components of forest management. As most silvicultural practices have been developed not for these purposes but for improving the timber production of commercial species (Feldpausch et al. 2005), there is an urgent need to evaluate traditional concepts against the background of the new aspects and needs. Sustainable management of tropical forests requires the integration of ecological knowledge with social, economic and political-institutional constraints and options (Hooper et al. 2005). Consequently, silvicultural concepts must encompass a broad range of objectives as well as the full set of options to manage trees for special purposes, such as natural forests, plantations, agroforestry and trees in landscapes (Van Bodegom et al. 2009). Unfortunately, corresponding silvicultural research on timber species, NWFPs and ecosystem services is sparse or lacking, and the existing information is concentrated on only a few species and in particular regions.

Nevertheless, in the last two decades significant, substantial progress toward sustainable multifunctional management of tropical forests has been made, particularly in terms of designation of permanent forest estates, formulation of policies to guide forest management and establishment of management plans. Furthermore, a range of initiatives to accelerate the implementation of sustainable forest management have been adopted: certification schemes, voluntary partnerships, forest law enforcement and government efforts. Initiatives such as the voluntary codes of practice developed by the FAO and the International Tropical Timber Organization (ITTO) provide benchmark standards for managers (ITTO 2006).

In this context, silvicultural concepts for planning and application are needed that can be applied under spatially explicit conditions and that consider different levels of scale ranging from landscape to species and genes (see also Chap. 7).

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Chapter 7
Biodiversity Conservation in Tropical Forests Managed for Timber

Francis E. Putz

Abstract All silvicultural interventions have biodiversity impacts, often by design (e.g., freeing future crop trees from species perceived of as weeds). In the case of timber stand management, the magnitude of the tradeoff between production of merchantable wood and retention of pre-intervention biodiversity varies with the intensity of the silvicultural interventions and the care with which they are applied. Given that the most severe silvicultural impacts on tropical forests result from selective logging, substantial biodiversity benefits can be realized by the adoption of environmentally sound timber harvesting practices.

7.1 Conservation Through Careful Use

Rampant deforestation and forest degradation in the tropics will most effectively be reduced with a portfolio of approaches to conservation. The mainstay of tropical forest conservation campaign will undoubtedly remain strict protected areas, but there are limits on the area that can and will be set aside entirely for conservation (Romero and Andrade 2004). Fortunately, where good management practices are used, forests from which timber is harvested also retain most of their native species, continue to store carbon and yield clean water, and generally provide the ecosystem services and other values that society demands (Putz et al. 2001). In this chapter the factors affecting biodiversity conservation in managed forests are evaluated, ways to mitigate the deleterious impacts are presented, and a global climate change motivated source of financial incentives for improved forest management is discussed.

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7.2 Diverse Forests, Equally Diverse Reactions to Silvicultural Interventions

Forests differ in their biodiversity, in their resilience when subjected to different intensities and types of silvicultural treatments, in pressures for conversion to nonforest use, and in the abilities of the relevant institutions to regulate their management (Putz et al. 2001). Although it is important to keep these differences in mind, it is also important to recognize that all forest interventions, no matter how purportedly "sustainable," have impacts on biodiversity. Just how much change in forest composition, structure, and function is acceptable is a societal decision but one that needs to be informed by forest scientists (Sayer et al. 1995; Luckert and Williamson 2005).

Assessing the biodiversity impacts of tropical forest management activities is made challenging by the wide variety of conditions under which forestry is practiced and the incredible diversity of organisms at stake. Even in the tropics some forests are naturally low in diversity and are exceedingly resilient owing to a history of major natural environmental disturbances (e.g., fire-maintained forests and typhoon forests). But many tropical forests are hyperdiverse, and even the identities of large numbers of species are unknown, let alone their likely responses to different silvicultural interventions. To put this reality in perspective, even in relatively low diversity forests in the temperate zone, rare and little-known taxa are generally in the majority (Rafael and Molina 2007).

Another factor to be considered when evaluating the biodiversity impacts of tropical forestry is the wide range of types and intensities of silvicultural interventions. Commercial logging intensities, for example, span nearly three orders of magnitude, from 5 to 150 $m^3\ ha^{-1}$ (Putz et al. 2001). Further complicating the picture is that logs can be extracted using a wide range of technologies, helicopters to draft animals, each with its own sort of environmental impact (Dykstra and Heinrich 1996). Both harvesting intensities and the log-yarding methods employed substantially influence biodiversity impacts, but equally important is whether logs are extracted from forests in unnecessarily destructive manners or with planning and care.

Many of the environmental concerns about tropical forestry are focused on logging, which is usually the most common, the most financially lucrative, and the most damaging of forest interventions. Although this focus is justified, substantial confusion has resulted from not distinguishing between the direct, or primary, effects of tree felling, yarding, and hauling, and the secondary effects, which derive mostly from the improved access provided by logging roads. The primary impacts of logging vary with the quality of management planning, with the type and intensity of felling, depending on whether "reduced-impact logging" (RIL) techniques are used (Putz et al. 2008a), and depending on which, if any, prelogging and postlogging silvicultural treatments are applied (e.g., soil scarification and liberation of future crop trees), but also depending on the particular species or other factors being considered. Secondary impacts result from human colonization and forest conversion radiating out from logging roads, wildlife poaching, vulnerability to fire, and overexploitation of forest resources by colonizers, company workers, and outsiders. These secondary impacts are often much more severe and enduring than the primary impacts of timber harvesting, but the emphasis in this review is nonetheless on the latter.

Nontimber forest product (NTFP) harvesting, which is a critical contributor to the livelihoods of millions of mostly poor people in the tropics and is the basis of a huge international industry, also has both primary and secondary impacts on tropical forests. In contrast to logging, the primary impacts of NTFP harvesting are typically minor, but the secondary impacts, which derive from having more people in forests, can be severe. With an estimated 25% of tropical forests under the control of rural communities (White and Martin 2002) and given the importance of NTFPs as well as timber to community members, the biodiversity impacts of NTFP harvesting deserve more attention, as do the methods for managing forests for both sorts of products. That said, the focus of this overview is on timber, but most of the observations pertain to NTFPs as well.

Although properly planned logging conducted by trained and supervised crews can be an integral component of sustainable forest management, logging, particularly in the tropics, often represents a timber "mining" activity without consideration for sustainability. Instead of timber harvesting operations being used as silvicultural treatments to promote the regeneration and growth of timber-producing species, logs are extracted from forests selectively (mostly owing to the low prices paid for the wood of most tree species) but without regard to residual stand conditions, soils, future yields, biodiversity, or the many ecosystem services that can be provided by managed tropical forests. Furthermore, the many prelogging and postlogging silvicultural treatments with which foresters have experimented (e.g., liberation thinning) are still mostly only applied in research plots. Despite these limitations, with the hopes that tropical forest exploitation will evolve into tropical forest management, it is important to consider the biodiversity impacts of the full range of forest management activities.

7.3 Disaggregating Biodiversity

Logging and other timber stand management activities have direct and indirect but differing effects on all the components of biodiversity, from landscapes and ecosystems to communities, species, and genes (Putz et al. 2001); differentiating these components fosters communication. Nevertheless, discussions of even just the primary impacts of forest activities on biodiversity are complicated by the wide range of possible forest management activities and the even wider range of forests, organisms, ecosystem processes, and responses to these interventions. Even at the species level the direct impacts are difficult to describe because species respond in distinct ways to logging and associated activities (see later).

7.3.1 Landscapes

At the landscape level, logging and other forest management activities can affect biodiversity by changing forest structure and ecosystem functions over large geographic areas. As the intensity and spatial extent of interventions increase, habitat patches increasingly change in size, identity, spatial distribution, and connectivity, all of which affect species distributions. Although the primary effects of logging at the landscape level can be substantial, more severe impacts are generally the indirect consequences of logging, such as increased access to remote areas, fragmentation, and altered fire regimes.

The opening of logging roads affects forests both immediately during their construction and over time as they are used by loggers and other forest users and converters. For example, wide logging roads fragment the populations of some species, particularly understory-adapted species with weak locomotory abilities. Overall, the degree of fragmentation and its impacts on biodiversity depend on whether logging is dispersed over large areas or concentrated in small areas. By opening the canopy and thereby increasing the rate of forest drying while indirectly causing increases in ignition frequencies as more people spend more time in the newly accessible areas, logging results in increased forest flammability, fuel loads, and hence fire intensity, as well as increased fire frequency (Nepstad et al. 1999; but see Blate 2005). Furthermore, when logging roads penetrate far into forests, logging is almost invariably accompanied by increased hunting pressure. Finally, logging is often followed by planned or unplanned deforestation for agricultural land uses ranging from subsistence farms to huge industrial African oil palm (*Elaeis guineensis*) plantations.

7.3.2 Ecosystems

Forest management activities have a multitude of ecosystem-level impacts, which vary with the intensity of logging and other silvicultural treatments and the care with which these operations are carried out. Because ecosystems differ in their responses to even the same intervention, the diversity of possible responses is substantial.

Biomass losses result not only from timber extraction itself, but also from damage to the trees in the residual stand and from subsequent silvicultural treatments. By opening the canopy, and especially by creating large canopy gaps, even light-selective logging increases the risks of uncontrolled fires and increases the intensities of fires that do occur. That this synergy extends to normally moist and wet tropical forests during droughts was made clear by the extensive fires that occurred during the severe El Niño fires of 1998 (Bowman et al. 2009). If managed stands are colonized by lianas and other low-biomass forest weeds (e.g., gingers and pioneer trees with low density wood), forest carbon stocks can be depressed for decades (Pinard et al. 2000). If soils are compacted or otherwise rendered unproductive during harvesting operations, reductions in the carbon storage capacity of the stand can be more permanent. Soil compaction from ground-based yarding also reduces the water infiltration rates and water holding capacity of soils, which leads to increased surface run-off. Impoundments caused by road building, bridge collapse, undersized culverts, and other poor harvesting practices can also have long-term consequences for ecosystem functions and biodiversity. Deposition of sediments in streams during road construction and use also affects the characteristics of water courses; the vast majority of erosion from logged water catchments is from roads, even if they cover only a small proportion of the land surface (Chappel and Thang 2007). Thus, flow regimes of natural streams can be greatly influenced by logging. These changes have grave but avoidable impacts on ecosystem functions and, consequently, on biodiversity.

7.3.3 Community

Logging, especially when followed by silvicultural treatments to promote the establishment and growth of commercial timber species, changes the proportions of species and successional stages in forests. For instance, partially because most commercial timber species, at least those traded internationally, are at least moderately light demanding, produce wood with moderate densities (i.e., 0.48–0.60 g cm^{-3}), and reproduce with wind-dispersed propagules (e.g., many Dipterocarpaceae, Meliaceae, Leguminosae, and Pinaceae), logging alone can result in decreases in the representation of species with this suite of characteristics. If logging is followed by intensive stand refinement treatments to release future crop trees from competition or to promote the regeneration of light-demanding species, rare, threatened, or endangered species may become locally extirpated if they have no commercial value. Similarly, disregard of NTFPs during the planning and execution of logging operations can have serious deleterious socioeconomic and biodiversity effects in multiple-use forests.

7.3.4 Species

The most obvious impacts of logging are on the abundance and age–size distributions of canopy tree species. Depending on the intensity of logging and the care with which it is carried out, the reproduction, growth, and survival of a great number of species can be adversely affected. Most fundamentally from a forest industries perspective, unless great care is taken during logging and, generally, if postlogging silvicultural treatments are not applied, populations of commercial tree species will decline.

The impacts of logging persist for many years after logging is completed. One cause of continued degradation of badly managed stands is the continued high tree mortality rates, often due to apparently slight but eventually fatal damage sustained during logging (Sist and Nguyen-Thé 2002). Also, the proliferation of weeds and losses of the services provided by seed-dispersal agents and pollinators due to population size reductions and fragmentation can have long-lasting effects. Animal species composition also changes in response to the direct effects of logging, such as canopy opening, and to the indirect effects, such as increased hunting pressure, fires, and forest conversion. Disturbance-adapted species, usually native but sometimes exotic, often proliferate in logged-over forests and affect the resident flora and fauna.

Although logging has a variety of deleterious effects at the species level, responses to forest activities are by no means consistent across or even within taxa. For example, unless appropriate silvicultural treatments are applied, populations of the harvested tree species are typically depleted, especially after a few logging cycles (Grogan et al. 2008). Often the result of this depletion is that light-demanding or "weedy" species appear and shade-demanding species with medium-density or high-density wood decline in numbers (Kariuki et al. 2006; Meijaard et al. 2005).

Faunal responses to logging differ within and between species and sites, as well as with the type and intensity of forest management interventions (Haworth and Counsell 1999; Azevedo-Ramos et al. 2005; Lagan et al. 2007). Chimpanzee populations have been reported to increase or to decrease in response to logging, and even not to respond at all (reviewed in Putz et al. 2001). In contrast, populations of terrestrial and bark-gleaning insectivorous birds consistently decline for at least a few years after logging, showing the negative impacts of logging (Thiollay 1997). Frugivorous canopy birds reportedly show slightly negative, neutral, or even positive population-level responses to selective logging (Meijaard et al. 2005). Populations of large and slow-reproducing animals are particularly susceptible to overhunting when forests are made more accessible by the construction of logging roads (Bennett et al. 2002). In general, the more ecologically specialized a species, the more likely it is to suffer population declines or even local extirpation after logging (Meijaard et al. 2005).

The deleterious effects of logging, especially when uncontrolled, persist for many years (Kariuki et al. 2006). Even though logging has no detectable negative short-term effects on many species, the indirect and long-term effects might be quite severe.

7.3.5 Genes

When large proportions of healthy reproductive adults are harvested, especially when coupled with losses of pollinators and seed-dispersal agents, genetic variation declines (for more details see Chap. 8). Unfortunately, the techniques required for assessing the genetic structure of populations are sophisticated and expensive; thus, most concerns about genetic issues in relation to silvicultural activities are based more on theory than on data (but see Lacerda et al. 2008).

7.4 Measuring and Monitoring Biodiversity Impacts

For a variety of reasons, including cost and lack of taxonomic expertise, the effects of silvicultural activities on tropical forest biodiversity are seldom measured, but numerous indirect indicators have proven useful. In particular, management process indicators can be measured that assess changes in ecosystem structure that are thought to be linked with biodiversity (Lindenmayer et al. 2000). Even if all the species present are known, which is never the case, simple counts of the numbers of species can provide a misleading indication of biodiversity impacts when, for example, rare, endangered, or otherwise highly valued species are lost and are replaced by site colonization by widespread and common disturbance-adapted species, resulting in no apparent impact on biodiversity (Cannon et al. 1998). It would appear that some sort of species weighting is needed, but none has emerged that is widely accepted (Scholes and Biggs 2005). Finally, when the biodiversity impacts of forestry practices are discussed, it is important to recognize that all silvicultural interventions affect biodiversity; most are applied with this goal in mind (e.g., to favor well-formed trees of commercial species at the expense of their competitors).

Given that most tropical forests are still being exploited for timber without concern for future yields, ecosystem processes, or biodiversity, it is strategic to prioritize the management practices to be adopted that are most critical. Control of hunting might be a first priority, but because it is not directly related to silviculture, it will not be discussed further here. Perhaps the next most important required changes on the pathway from log mining to forest management are related to maintaining hydrological functions and the biodiversity they support. In particular, protection of riparian buffer zones and minimization of sediment loading of streams have a variety of biodiversity benefits (Fredericksen and Peña-Claros 2007).

In the national "codes of forest management practices" promoted by international agencies (Dykstra and Heinrich 1996) and adopted by most tropical countries (Vanuatu Department of Forests 1998) as well as in the "criteria and indicators" of most forest certification programs, protection of streamside buffer zones figures prominently. Usually the width of the unlogged buffer varies with stream width; in Vanuatu's code, for example, streams more than 20 m wide require a buffer zone of 30 m on each bank. The expected benefits of such buffers include reduction in stream sediment loads by the filtering effect of intact vegetation and soils as well as provision of habitat and corridors for movement of animal species sensitive to canopy opening and other forest-management-related disturbances (Nasi et al. 2008).

Emphasis in environmental regulations on the construction, use, and maintenance of roads and skid trails is justified on the basis of the fact that the vast majority of the erosion in logged catchments is from these traffic corridors. Poorly sited roads with poor drainage (e.g., lack of side and cross drains), improperly sized culverts, and poorly constructed bridges all result in excessive and avoidable soil losses. Where the movements of ground-based log-yarding machines (e.g., skidders and crawler tractors) are not limited to preplanned and rationally arranged trails, the resulting soil damage (e.g., exposure and compaction of mineral soil) has deleterious environmental impacts from which recovery may require decades.

Of all the potential silvicultural treatments that might be prescribed for tropical forests managed for timber, liana cutting is one of the most common. The case of liana cutting serves as a good example of the trade-offs forest managers face when trying to balance timber production goals, worker safety, and biodiversity impacts while keeping management costs low. Silviculturalists have long been aware of a variety of negative impacts of lianas on timber production, and liana cutting has consequently been one of the most frequently prescribed silvicultural treatments. For example, by tying together the crowns of trees, lianas can cause felling gaps to be larger and endanger tree fellers in the process (Schnitzer and Bongers 2002). Then, because lianas are narrow-stemmed and often exceedingly flexible, they mostly survive falling, produce adventitious roots from the prostrate stems, and spout profusely; often more than half of the liana stems in felling gaps and on their margins are the sprouts of individuals that formerly were in the canopy (Alvira et al. 2004). Liana tangles, where dense, can inhibit tree regeneration for decades (Putz 1984; Schnitzer and Bongers 2002). Where lianas are abundant, it makes silvicultural sense to cut those entangled in trees to be felled and future crop trees before logging; blanket prescriptions (i.e., cut all lianas) are much more expensive and certainly have larger biodiversity impacts than the more selective treatments.

That lianas can be a severe impediment to the achievement of timber production goals is obvious, but they are also important components of tropical forest ecosystems and are critical for biodiversity maintenance (Fimbel et al. 2001). Not only do lianas provide food (e.g., flowers, fruits, and abundant leaves), they also provide critical intercrown pathways for a wide range of nonvolant canopy animals. For small animals susceptible to raptor predation, the liana tangles that suppress tree regeneration also provide important refugia. Liana cutting, even where selective, obviously will have biodiversity impacts, but owing to the remarkable capacity of lianas to resprout, recovery is typically rapid (Gerwing 2006).

Low-intensity RIL with no or only mild silvicultural treatments may be the best approach to maintaining preintervention forest structure and composition, but for sustaining timber yields, the gentle approach is not always the most suitable (Fredericksen and Putz 2003; Putz and Fredericksen 2004; Sist and Ferreira 2007; but see Sist and Brown 2004). Although RIL techniques that protect soils and hydrological functions are always appropriate where biodiversity conservation and forest production are concerned (Putz et al. 2008b), sustaining the yields of some commercially valuable species, particularly those that are light-demanding, often requires more forest disturbance than is typically the goal of RIL. The most familiar example of this phenomenon is mahogany (*Swietenia macrophylla*), which in many forests regenerates best after hurricane-like disturbances followed by competition-reducing fires (Snook 1996). In contrast, in forests where there is abundant "advanced regeneration" (i.e., seedlings, saplings, and pole-sized trees) of commercial species, such as in the dipterocarp forests of Southeast Asia, protecting the residual forest is critical and RIL is tantamount to sustained-yield forest management (Meijaard et al. 2005). But more fundamentally, given the many ways in which tropical forests are valued (Frumhoff 1995), whether efforts should be made to sustain timber yields needs to be informed by foresters and researchers but decided by society.

7.5 Improved Forest Management and Global Climate Change

Increased awareness of the speed and magnitude of human-induced global climate change has resulted in increased attention to the contributions of tropical forests to the problem and potentially to the solution. Given that tropical land-use changes, principally deforestation, are estimated to contribute nearly 20% of global emissions of atmospheric heat-trapping gasses, coupled with recognition of the regional contributions of standing forests to moisture maintenance, negotiators of the new global climate change protocol are considering including a program designed to reduce the emissions from deforestation and forest degradation. Although attention is focused on deforestation, the substantial loses of carbon due to forest degradation also need to be considered (Asner et al. 2005; Sasaki and Putz 2009). Fortunately, carbon retention in managed tropical forests can be enhanced easily and inexpensively simply by widespread adoption of environmentally sound timber-harvesting practices (Putz et al. 2008b). Given that most of the biomass carbon in forests is in large trees, further carbon savings can be secured through silvicultural treatments applied to promote the survival, recruitment, and growth of the same sorts of trees that foresters have long favored for timber. Financial incentives for improved forest management as an alternative to degradation and deforestation are especially important where nonforest land uses (e.g., cattle ranching and oil palm plantations) are much more lucrative than management for timber (Pearce et al. 2002). Unfortunately, because many environmentalists are staunch "protectionists" and object to logging and other silvicultural interventions, no matter how carefully applied, the potential contributions of improved forest management to mitigating global climate change and protecting biodiversity are too often disregarded.

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Chapter 8
Management of Forest Genetic Resources

Reiner Finkeldey

Abstract Human impacts on tropical forests are manifold. The consequences are often directly visible and obvious, such as loss of forest cover and forest fragmentation due to alterations of land use. Other consequences are more subtle and difficult to monitor, but pose severe threats to the maintenance of biodiversity. Intraspecific diversity in tropical forests constitutes a fundamental component of global biodiversity. Thus, sustainable management of tropical forests needs to consider the conservation and wise use of genetic diversity.

8.1 Genetic Diversity in Tropical Forests

Our understanding of the levels of genetic variation of tropical forest trees, its spatial distribution, and human impacts on genetic diversity has greatly increased during the past decades (Finkeldey and Hattemer 2007). Investigations of biochemical and molecular gene markers proved high levels of genetic diversity for most tropical tree species. Isozyme studies revealed that genetic variation of trees in comparison with herbaceous plants and most groups of animals is on average considerably higher. The genetic diversity within tropical trees is only slightly lower than the diversity levels within trees of the temperate and boreal zone (Hamrick et al. 1992). This even holds for many tree species occurring in extremely low density in species-rich tropical forests (Hamrick and Murawski 1991). These general statements are based on a large number of investigated species, some of which harbour much less variation than the average. For example, low diversity was observed in natural populations of *Acacia mangium* at isozyme gene loci (Moran et al. 1989).

The combination of particular life history traits, most importantly longevity combined with efficient mechanisms of gene flow, is responsible for the

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maintenance of high levels of diversity. Selfing rates of tropical trees are usually low, and the dominance of animal pollination safeguards efficient transport of pollen even among widely separated trees. This keeps effective population sizes high and prevents losses of diversity due to genetic drift.

8.2 Human Impacts on Genetic Diversity

Humans do not only deliberately change genetic structures of a few plantation species with the objective to increase the yield in plantations through tree improvement. Most tropical tree species have never been included in well-designed improvement programmes, but their patterns of genetic variation have been modified by human impact such as forest fragmentation and silvicultural treatments, notably selective cutting.

Forest destruction rarely wipes out all forests in a particular region completely, but often leaves patches of more or less disturbed forests in a landscape dominated by other land uses. If these patches are reproductively isolated from each other, losses of genetic diversity due to genetic drift and negative effects due to inbreeding are expected in the long run. Thus, the crucial question from a genetic viewpoint refers to the connectivity of forest fragments, i.e. the degree of gene flow among spatially isolated fragments. The consequences depend on pollen and seed vectors of a given species. Since plant–animal interactions, including possible shifts of pollinators in fragmented landscapes (Dick et al. 2003), are species-specific, it is difficult to discuss the genetic consequences of forest fragmentation in general. For example, extraordinarily large breeding units have been estimated for figs (*Ficus* spp.) since the tiny wasps pollinating figs are able to bridge distances of much more than 5 km (Nason et al. 1998). Thus, figs which are spatially isolated from conspecifics by several kilometres may be part of a viable population. On the other hand, fragmentation and conversion of forest to pasture resulted in highly uneven fertilities in a Costa Rican population of the neotropical species *Symphonia globulifera* (Aldrich and Hamrick 1998). Although natural regeneration was more abundant in forest fragments in comparison with a closed forest, the reproductive dominance of the few trees left on the pasture land resulted in elevated inbreeding and induced a genetic bottleneck in the next generation.

Main silvicultural treatments in naturally regenerated tropical forests are selective cutting systems, which are usually based on target diameters and phenotypic assessments of trees. Large trees and trees with favourable traits are preferentially harvested. These harvesting operations are of limited genetic interest unless the trees that were removed had the chance to transmit their genetic information to the next generation. This implies the use of a preexisting natural regeneration for the establishment of a new generation. However, selective cutting usually favours the regeneration from remaining, non-harvested trees. Assuming that diameter growth and quality traits are under partial genetic control, selective cutting as currently practised often favours the reproduction of inferior trees and results in dysgenic selection. Even though the consequences of selective cutting systems in tropical forests have not yet been described at gene loci controlling growth and quality traits, dysgenic selection should be regarded as a main threat to genetic resources of forest trees (Finkeldey and Ziehe 2004).

8.3 Conservation of Forest Genetic Resources and Tree Improvement

Forest destruction and other human activities threaten the biodiversity of tropical forests. The Convention on Biological Diversity emphasises the need for the conservation not only of species, but also of intraspecific diversity. On the other hand, increasing plantation areas in many tropical countries call for an increased use of forest genetic resources by breeding and domestication. Sustainable management strategies for genetic resources take into consideration both the urgent need for conservation and the importance of the utilization of genetic resources.

Most tree species can be efficiently conserved only in natural forests and naturally regenerated secondary forests. In situ conservation is frequently feasible even if only small patches of a previously much wider occurrence of a species are left in a region. For example, high genetic diversity was observed in remnant populations of *Cordia africana* (Derero et al. 2011) and *Hagenia abyssinica* (Taye et al. 2011) from Ethiopia, a country severely affected by forest destruction.

Applications of forest genetic research to tropical forest management are gaining importance not only in view of the urgent need to conserve biodiversity at all levels, but also owing to the importance to provide reproductive material for plantation areas. Most plantation species are in a very early stage of domestication, and considerable improvement may be expected from the establishment of well-designed provenance trials, simple selection methods and the harvesting of seeds in properly managed seed production areas and seed orchards (Finkeldey and Hattemer 2007). More advanced breeding methods, including the production of full-sib families and artificial hybridization, the establishment of progeny and clonal tests and the production of reproductive material in advanced propagation populations are confined to a few taxa such as eucalypts.

8.4 Procurement of Forest Reproductive Material

The enormous success of intensive breeding methods for eucalypts and a few other tropical taxa is undisputed and the need to use high-quality reproductive material for plantation establishment is widely recognized in the tropics (Schmidt 2007). Thus, it is surprising to note that the vast majority of reproductive material used for tree plantation establishment is of unknown origin. Legal regulations controlling the marketing of forest reproductive material (Nanson 2001) either do not exist or are not enforced in most tropical countries. Seeds are often harvested from solitary trees, small plantations or phenotypically inferior trees. The same applies to the collection of wildlings. Ease of harvesting or collection and production costs for reproductive material are usually decisive for the choice of the material used for plantation establishment. The comparatively long time gap between planting poorly adapted reproductive material and harvesting in plantations not exploiting the full growth potential at their respective sites is likely to be a main reason for the dominance of the use of material of unknown origin. Although it is currently impossible to make even rough estimates of the economic losses caused by this practise, the extent of existing tree plantations in the tropics with unknown establishment history suggests that the problem is severe though rarely discussed among plantation managers and in the relevant scientific literature.

A case study compared genetic structures of five natural stands and neighbouring plantations of *Dalbergia sissoo* in Nepal (Pandey et al. 2004). Most *D. sissoo* plantations in Nepal are poorly adapted to their respective sites and show complex symptoms of diseases. Genetic analyses strongly suggested that the origin of the reproductive material used for plantation establishment in Nepal was far from the planting sites. No plantation was established with seed from the neighbouring natural stands. Most likely, seeds were imported from an unknown region in India for the establishment of *D. sissoo* plantations in Nepal. Thus, uncontrolled long-distance seed transfer is likely to be a main factor contributing to the disappointing performance of *D. sissoo* plantations in Nepal. A companion study using a comparable design showed no comparable effects of plantation establishment with material of unknown origin for an even more important plantation species in Nepal, *Pinus roxburghii* (Gauli 2007). Harvesting operations within the country and efficient gene flow by pollen resulted in similar genetic structures of natural populations and plantations for this wind-pollinated pine.

8.5 Outlook: Genetic Fingerprints for Timber Certification

The application of modern methods in molecular genetics to tropical forest trees and in particular the use of molecular genetic markers to assess genetic diversity of numerous species not only enhances our understanding of the dynamics of genetic diversity in tropical forest ecosystems, but also opens up options for new fields of application.

Genetic fingerprints potentially allow the proclaimed origin of wood to be tested and false declarations to be identified. These molecular investigations are of interest to government organizations such as customs controls depending on non-manipulable methods to identify illegally harvested wood or certification agencies using reliable tools to check the chain of custody. The methods depend on the availability of protocols to extract DNA from unprocessed and processed wood, and on the development of DNA-based markers showing strong differentiation among regions (Finkeldey et al. 2010). The feasibility to extract DNA from tropical timber has been proven for the species-rich dipterocarp family (Rachmayanti et al. 2006, 2009). DNA-based markers to identify endemic species of the Dipterocarpaceae have been developed (Indrioko et al. 2006), and markers to distinguish among growing regions are being developed for the most important widespread dipterocarps (Nuroniah et al. 2010). However, the optimization of protocols for DNA extraction from processed wood and the establishment of databases containing relevant spatially explicit genetic data are time-consuming tasks.

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Chapter 9
Managing Short Rotation Tropical Plantations as Sustainable Source of Bioenergy

Jonathan C. Onyekwelu

Abstract Renewable energy has been identified as a promising option for reducing the heavy dependence on fossil fuels. The global use of bioenergy is equivalent to nearly a billion tons of oil and contributes about 10% of world's primary energy demand. Bioenergy is expected to become a key future energy resource. Short rotation energy plantation species are characterized by fast growth rate and high yield. Depending on tree species, rotation age, and biomass yield, energy yield from short rotation energy plantation could range from 312 to 9,792 GJ ha^{-1}. However, allowance must be made for branch and foliage biomasses that are usually left on site after harvesting as well as for the overall low efficiency of converting biomass. The current and expected future increase in world food prices, partly occasioned by the competition between resource use as food and biofuel feedstock, calls for reconsideration in the use of edible crops as bioenergy feedstocks. Future directions will be the use of resources that are inedible, cost-effective, high yielding, and easy to sustain as bioenergy feedstocks, as well as wastes from forestry and agrifood industries, domestic and industrial products.

Keywords Short rotation forestry · Bioenergy plantations · Sustainability · Degraded forestland · Energy yield

9.1 Introduction

In recent years, concerns about the environmental consequences of heavy dependence on fossil fuels have been growing (Bhattacharya et al. 2003; Raison 2006). Global energy-related carbon dioxide (CO_2) emission is projected to increase from 26 Gt in 2004 to 40 Gt in 2030 (IEA 2006). The high prices of fossil fuels and the

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growing concerns over nuclear energy have awakened the interests of both the public and policy and decision makers to bioenergy (IEA 2002). These concerns are encouraging the use of alternative and renewable energy sources. Renewable energy has been identified as a promising option for reducing the heavy dependence on fossil fuels and consequently mitigating its effect on the climate. Bioenergy is the most widely used renewable energy source, representing nearly a billion tons of oil consumption levels comparable to that of natural gas, coal, and electricity (IEA 2002). It is expected to become one of the key energy resources in the future because it is renewable and free from net CO_2 emissions. In 2007, bioenergy contributed about 10% of the 470 EJ world primary energy demand (FAO 2008).

Bioenergy has been in use since the early days of humanity. Today, more than 2.5 billion people depend on bioenergy, particularly fuelwood and charcoal, for their household energy (IEA 2006). Over half of this population live in tropical countries. An estimated 580 million people in sub-Saharan Africa, which is projected to increase to 820 million within the next three decades, rely on bioenergy (Arnold et al. 2003; IEA 2006). Bioenergy is experiencing a surge in interest due to the recognition of its current role and future potential contribution to modern fuel, its availability, versatility and sustainability, global and local environmental benefits, existing and potential development and entrepreneurial opportunities (Onyekwelu and Akindele 2006). The recent years have particularly witnessed a massive and growing expansion of a particular form of biomass based energy: liquid biofuels. Although, biofuels still account for a very small share of global energy consumption, that share is growing fast (IEA 2006; Cotula et al. 2008).

Much attention is currently focused on identifying suitable species that can provide high-energy outputs as well as developing appropriate biomass conversion technologies. The type of biomass required is largely determined by the energy conversion process and the form in which the energy is required. Biomass productivity depends on factors such as species, soil quality, climate, silvicultural practice, etc. Short Rotation Forestry (SRF) tree species have been recognized as one of the main sources of bioenergy, with the potential of being the largest source in the future (IEA 1998; Sajjakulnukit and Verapong 2003). This case study reviews the potentials of managing SRF tropical plantations as sustainable sources of bioenergy.

9.2 Potentials of Bioenergy

The potentials of bioenergy are hinged on their renewability, availability, versatility, sustainability, and land availability. Bioenergy plantations have environmental benefits, such as a positive impact on biodiversity, nutrient capture, and carbon circulation in the soil–plant atmosphere system. They can be established anywhere, even on poor soils and high yields will be expected. In addition, the unpredictable and often high price of fossil fuels, the desire to reduce dependence on them, and the increasing quest to diversify energy sources, have all increased bioenergy potentials. Bioenergy could become the most important renewable energy within the next 25 years (Lal and Singh 2000). Modern bioenergy technologies can be set up in urban and rural locations. Biomass fuelled electricity generating technologies and bio-based refineries and chemical industries are currently available (Bhattacharya et al. 2003) while new ones are springing up, which offer new potentials for bioenergy use.

The inability of many people in developing countries to afford fossil fuels, coupled with the failure of energy infrastructures such as electricity, etc. (Onyekwelu and Akindele 2006) has increased their dependency on bioenergy. For example, between 75 and 95% of energy needs of most rural households in Africa are met from bioenergy. In addition, a significant number of small-scale industries in Africa such as tea, tobacco, cassava processing, alcoholic beverage, wood processing, bakeries, brick and tile industries, etc. rely on bioenergy (Onyekwelu and Akindele 2006).

9.3 Short Rotation Energy Plantations

9.3.1 Land Available for Energy Plantations

A basic question related to plantation establishment is land availability. In short rotation energy plantations, land availability is among the most critical considerations. Large areas of land are needed for energy plantations. In 2004, about 14 million hectares of land (about 1% of the world's arable land) was used for bioenergy plantations, which is projected to increase to 3.5% by 2030 (FAO 2008). To reach the target of 500 million hectares of bioenergy plantations by 2050, an average of 10 million hectares of plantations would have to be established annually (Berndes et al. 2003). The major factor that determines land availability for bioenergy plantations is the demand on land for food production (IEA 2002). The recent high prices of food crops, coupled with an increasing population, have increased the demand for agricultural land (Elauria et al. 2003; FAO 2008) and thus more pressure on available land. Other factors that determine land availability include legislation and policies, opportunity cost of land, financial viability, and market demand (Elauria et al. 2003).

Latin America and sub-Saharan Africa have abundant land resources for plantations (FAO 2008), most of which are unsuitable for agriculture. If bioenergy is to contribute substantially to the world's energy supply, SRF plantations will require degraded forestland, grassland, fallow land, marginal and abandoned land (McKendry 2002; Elauria et al. 2003; FAO 2008). Since high yields can be obtained from bioenergy plantations established on poor and marginal land, it is expected that energy plantations established on the categories of land mentioned above will be productive and sustainable. The focus of bioenergy plantations should be to ensure that these lands are used economically, socially, and sustainably. Annual global deforestation rate was about 13 million ha between 2000 and 2005 (FAO 2005). An estimated 950 million ha of marginal agricultural land, mainly in South America, Africa, and China is expected to be available for bioenergy plantations by 2050 (Dameron 2007). Since these categories of land are increasing due to increasing deforestation and land degradation, it implies that land potentially available for energy plantations is also increasing. With an annual global forest plantation establishment rate of 2.8 million ha between 2000 and 2005 (FAO 2005), it is evident that a high percentage of the land potentially available for bioenergy plantations is underutilized. Only about 6% of the available land will be utilized for SRF energy plantations by 2050 (Dameron 2007).

9.3.2 Silviculture of Short Rotation Energy Plantation

As an industrial product, the aims of SRF plantation should be to make maximal profits by achieving maximum biomass production from a site within a specific period of time at minimal costs. This will entail appropriate silviculture to optimize genotype and/or cultural management. Numerous species have been and are being tested for energy potentials. The choice of a species for SRF plantation depends on fast growth rate, high biomass production, disease resistance, bioconversion options, climate, and soil conditions (McKendry 2002; Laureysens et al. 2004). Generally, the characteristics of suitable bioenergy tree species include:

- Early vigorous and fast growth throughout rotation;
- High yield (maximum production of dry matter per hectare);
- Ability to tolerate a wide variety of climate and site conditions;
- Ability to grow in monoculture;
- Ease of propagation from seeds and cuttings;
- Ease of establishment, with low cost of establishment;
- Low nutrient requirements;
- Good coppicing characteristics with ability to do so over several rotations;
- Resistance to pest and diseases;
- Short rotation period;
- Simple and easy silviculture.

To obtained high yields, bioenergy plantations require intensive silviculture and management. The management of SRF plantations is more like an agricultural practice than a conventional forest plantation practice. Cultural treatments generally include site preparation, high planting density, fertilization, coppicing, etc. Site preparation must be intensive and completed prior to planting. Existing vegetation is killed manually or with suitable herbicides (Ledin and Willebrand 1995). The ground is then deep-plowed, disced, and left fallow. Prior to planting out, the soil is harrowed and a contact herbicide may be applied to create a fine, weed-free tilth. Soil-nutrient analyses should be carried out to ascertain any deficiency, which could be rectified with inorganic or organic fertilizers.

Nursery operations can last between 4 and 6 months, depending on species and growth rate. In tropical rainforest zones, nursery is targeted to be completed by the onset of rainy season. Seeds are usually sown in a mixture of river sand and topsoil and fertilization is necessary. Seedlings are transplanted during the rainy season after about 100 mm of steady rain has fallen. Although early plantations were established manually and are still used for planting small areas, it is usually slow and time consuming. The establishment of larger areas through mechanical means has led to the development of several semiautomatic mechanical planters. Unlike conventional forest plantations, almost all SRF are established using high initial tree density (Ceulemans and Deraedt 1999; Laureysens et al. 2004). Generally, plant density of between 5,000 and 16,670 ha^{-1} is maintained in bioenergy plantations in many parts of the world (Ceulemans and Deraedt 1999). High density will facilitate rapid establishment, reduce the need for early weeding, and result in a higher yield. However, plant material can account for up to 65% of establishment costs and any advantage gained by high density may be outweighed by increased costs. Fertilizers may be applied depending on management intensity, soil-nutrient status, and the expected yield. Herbicides may be applied to control and eliminate early competition from weeds.

Apart from weeding during the initial stages, other tending operations such as pruning, thinning, etc. are not necessary since stem quality in energy plantations is not important. In SRF, rotation age is guided by initial density and growth rate. For most SRF, the denser the initial stand, the earlier the maximum yield is attained. Rotation age varies from 4 to 15 years, depending on species and growth rate. After harvesting at the end of a rotation, the stumps are left to coppice. The coppicing ability of SRF tree species can reduce plant material requirements and establishment costs. Since several coppices arise from a single stump, tending is essential. If left untended, the coppices that will survive will become spindly, leaning, have one-sided crowns, and poor contact with the stool (Evans and Turnbull 2004). Tending is usually done in two stages: the first, conducted at the end of the first year, reduces the number of shoots to about three or four per stump while the second, conducted at about 18 months, reduces the stems to one dominant shoot per stump (Evans and Turnbull 2004).

Box 1: *Moringa oleifera*: A Multipurpose Tropical Tree Species with Great Bioenergy Potentials

M. oleifera or drumstick tree is a fast-growing deciduous, medium-sized tropical evergreen tree species that grows up to 12 m in height at maturity. Moringa is a drought resistant pioneer species, which can easily adapt to various ecosystems. However, it grows mainly in semi-arid tropical and subtropical areas. It is indigenous to sub-Himalayan regions of northwest India, Africa, Arabia, Southeast Asia, the Pacific and Caribbean Islands, and South America, but is now distributed in the Philippines, Cambodia, and Central and North America (Rashid et al. 2008). The species is found from

(continued)

sea level up to 1,000 m altitude and can tolerate poor soil, a wide rainfall range (750–2,250 mm per year), and soil pH from 5.0 to 9.0. It is adapted to a wide range of soil types, but grows best in dry sandy soil and in well drained loam to clay loam soil and does not tolerate water logging. With oil contents of 35–47%, Moringa seeds have great bioenergy potentials. NWFP Digest (2008) reported that it is capable of producing up to 2,000 L of oil per hectare. The Moringa oil can be used for biodiesel.

The establishment of Moringa plantations requires very little silvicultural attention. It is easily established by seed or cutting. Seeds are sown either directly or in containers, without any pretreatment and germinate between 7 and 30 days after sowing. Up to 10,000 seedlings are planted per hectare in Tanzania. The rapidly growing seedlings can reach a height of 5 m in 1 year if sheltered from drying winds and provided with enough water. Cuttings of 1 m in height and 4–5 cm in diameter are typically planted during the rainy months. Seedlings raised from cuttings produce fruits of superior quality and quantity than those from seeds (Ramachandran et al. 1980). Fruit production begins during the first year of planting and continues up to 15 years. Fruit yield during the first 2 years is generally low. For example, 1-year-old plantings yielded 350 L of oil per hectare, but output was expected to increase dramatically in coming years (NWFP Digest 2008) as the trees reach their peak production capacity. In favorable environments, an individual tree can yield 50–70 kg of pods or 600–1,600 fruits in a year (Ramachandran et al. 1980; Sherkar 1993). Moringa coppices excellently. The yield of the coppiced trees has not been reported to be inferior to that from cuttings.

9.3.3 *Energy Yield from Short Rotation Plantations*

The amount of dry matter (t ha^{-1}) produced by a species per unit area determines its potential energy production capacity. Bioenergy plantation species are characterized by fast growth rate and high yield. Biomass yield and increment of some major tropical tree species are presented in Table 9.1. Biomass increment could range from 6.0 to 30 t ha^{-1} year^{-1}, depending on species and age. *Eucalyptus* spp., *Pinus caribaea*, *Gmelina arborea*, *Paraserianthes falcataria*, and *Acacia* spp., all showed high biomass increment (Table 9.1). Research is ongoing on the possibility of increasing yields through hybrid plants. Hislop and Hall (1996 [cited in McKendry 2002]) reported high biomass increment of 43 t ha^{-1} year^{-1} and 39 t ha^{-1} year^{-1} using hybrid poplar and eucalyptus species in Brazil.

Each bioenergy species has a specific yield; hence, it is useful to make assumptions in generating energy values. For wood derived from SRF, McKendry (2002) assumed average Low Heat Value (LHV) of 18 MJ kg^{-1}. At full generation rate, 1 kg of woodchips converts to 1 kWh(e) in a gasifier/gas engine generator, giving an overall 20% efficiency of conversion to electricity (Warren et al. 1995). McKendry (2002) opined that 1 ton of dry matter equals to 1 MWh(e), thus 15 t ha^{-1} year^{-1}

Table 9.1 Biomass yield and increment of some major tropical bioenergy plantation species

Tree species	Rotation age (years)	Mean biomass yield (t ha^{-1})	Biomass increment (t ha^{-1} $year^{-1}$)	Energy yield (GJ ha^{-1})
Acacia mangium	5–14	326.0	15.0–25.0	4,890.0–8,150.0
Eucalyptus camadulensis	4–8	64.7	13.2–21.6	8,54.0–1,397.5
Eucalyptus globules	3–5	64.7	12.0–22.0	776.4–1,423.4
Eucalyptus grandis	5	52.0	6.0–10.0	312.0–520.0
Gliricidia sepium	5–8	62.0	8.0–16.0	496.0–992.0
Gmelina arborea	8–15	265.6	10.5–21.5	2,788.8–5,710.0
Leucaena leucocephala	5–8	65.8	9.0–13.16	592.2–894.9
Paraserianthes falcataria	9	180.0	10.0–20.0	1,800.0–3,600.0
Pinus caribaea	10	158.0	12.0–15.9	1,896.0–2,370.0

yield would provide 15 MWh(e) per year. This was applied for the species on Table 9.1 to generate the energy yield. An energy yield of 312–9,792 GJ ha^{-1} could be obtained from SRF tropical species (Table 9.1). Bearing in mind that the usual practice in many tropical countries is to establish forest plantations by completely clearing degraded natural forests (Chen et al. 2004; Evans and Turnbull 2004; Onyekwelu et al. 2006), the yield on Table 9.1 is expected from bioenergy plantations established on degraded forest lands. However, since the yield in Table 9.1 is total above ground biomass, a 25% reduction in total energy yield should be made to account for branches and foliages, which are usually left on site after harvesting. Although these components could be used for bioenergy, they are better left on the site for nutrient replenishment. The overall efficiency of converting biomass to energy is low. For example, combustion processes using high-efficiency, multi-pass, steam turbines to produce electricity, can achieve an overall efficiency of 35–40%, while integrated gasification combined cycle (IGCC) gas turbines can achieve about 60% efficiency (McKendry 2002). This will increase the amount of biomass needed to produce a unit of energy.

9.4 Future Direction for Sustainable Bioenergy Production

Most biofuels (mainly biodiesel) currently in use are produced from edible agricultural products (e.g., corn, palm oil, sugarcane, rape seeds, sunflower seeds, etc.) or those used in the manufacture of edible products. One of the current contentious issues on the commercial use of biofuel is its role in food price increase (Cotula et al. 2008), especially in developing countries of Africa, Asia, and South America. It is feared that the continued use of edible resources as bioenergy feedstocks will lead to competition with food crops and a significant negative impacts on food security (Cotula et al. 2008). These concerns are particularly relevant for large-scale commercial biofuel production, which tends to consume a large amount of food crops as raw materials as well as consume lands that are suitable for food production.

Consequently, there is the need for reconsideration of current biofuel policy and a shift from heavy dependence on food crops as biofuel feedstocks. Apart from investment in SRF plantations, one of the considerations is to use resources that are inedible, cost-effective, high-yielding, etc. as biofuel feedstocks. Such energy crops include *Jatropha curcas*, *Azadirachta indica* (Neem), *M. oleifera* seeds, and other non-food seeds. These energy crops are attractive because of their low-cost seeds, fast growth rate, short rotation period, ability to grow on good and degraded soil, ability to grow in low and high rainfall areas, high annual seed yield, high oil content, etc. Onyekwelu et al. (2008) reported a good content of ethanol in *G. arborea* fruit pulp, which can be converted to biofuel. The second consideration is to source biofuel feedstocks from forestry and agrifood industrial wastes (e.g., wood and crop residues), domestic and industrial waste products (e.g., waste paper, household rubbish) (Cotula et al. 2008) etc. Pilot plants for biofuel generation from these feedstocks already exist in Japan and the USA.

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Chapter 10
The Silviculture of Tropical Nonwood Forest Products, Between Farming and Forestry

Paul Vantomme

Abstract Reviewing the silviculture of nonwood forest products (NWFP) first requires a clear understanding of its associated vocabulary. This is necessary in order to prevent confusion in the circumstances when the described "silvicultural techniques" are actually part of "forestry" or "agriculture." "Silviculture" of NWFP spans both the "forestry" and "agriculture" domains because most NWFP species are actually in a dynamic process of domestication, moving from traditional gathering/hunting practices in forests toward more intensive cultivation on farms.

Silvicultural interventions favoring the growth of NWFP-bearing species in tropical forests are governed by the NWFP user perspectives, which may range from satisfaction of subsistence needs to the production of commodities for industrial processing and international trade. In this chapter, the complexities of combining silvicultural interventions for managing forests for the production of both timber and nontimber goods and services are described.

The planning of silvicultural interventions for NWFP species through basic forest management is rarely done. It requires a multifaceted approach in order to integrate the many and often conflicting user demands for food, fiber, energy, health, and recreational goods. It also requires the active participation of a much wider range of stakeholders than when dealing with timber alone. International organizations such as the FAO can play a key role in raising awareness and building the required technical and institutional capacities in countries to incorporate NWFP silviculture within overall sustainable forest management.

Keywords NWFP · Silviculture · Sustainable forest management · Domestication · Certification

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10.1 Clarifying Terms and Definitions Related to NWFP and Silviculture in the Tropics

Tropical forests make up 47% of the world's total forested area, estimated at almost 4 billion ha by the FAO (FAO 2000a).

Many terms are used to capture the wide range of forest-based plants and animals from which goods (other than timber) and services are derived: minor forest products, nontimber forest products, nonwood forest products (NWFPs), naval stores, etc. (Belcher 2003), and the debate on terms and definitions is still ongoing. For the purpose of this chapter, the term NWFPs is used and defined as products of biological origin other than wood derived from forests, other wooded lands, and trees outside of forests (FAO 2000b). It is important to note that this definition does not include fuel wood, poles, carvings, or handcrafts made of wood and that silvicultural interventions for their production will not be covered in this section. On the other hand, NWFP can also come from gathering and hunting practices outside forests (such as honey, mushrooms, trophies, or nuts).

10.2 The Dynamics of Producing NWFP in the Tropics: From Hunting/Gathering to Farming

Since human existence, people have gathered plant and animal resources in forests to satisfy their food, shelter, and energy needs. Examples include edible nuts, mushrooms, fruits, herbs, spices, gums, aromatic plants, game, wood, fodder, and plant or animal products for medicinal, cosmetic, or cultural uses. Still today, 1.2 billion people, mostly poor and in rural areas of developing countries, derive a significant part of their subsistence needs and income from plant and animal products gathered in the forests (FAO 2007, 2009). For example, in India alone, the World Bank estimates that some 275 million people depend on forests for part of their livelihoods (World Bank 2006). Some of the NWFP enter national and international trade such as pine resins, rattan, bamboo, medicinal plants, and gum arabic. By large, the use of NWFP is part of the informal sector and as such is characterized by lack of reliable data on production, trade, and on the number of people involved.

Globally, 90% of the total reported output by countries originates from five major categories of NWFP uses (listed in descending order of importance): "food" with mainly fruits, berries, mushrooms, and nuts; "exudates" mainly gum arabic and pine resins; "other plant products" such as bamboo, rattan, and cork; "honey"; and "ornamental plants," mainly Christmas trees.

A breakdown of major NWFP outputs by region is detailed as follows:

Europe: Food, honey, ornamentals, and hunting products. Russia, Germany, Spain, Portugal, and Italy are the major producers.

Asia: Food, other plant products, and exudates. China is by far Asia's largest producer, followed by India.

Africa: Exudates and food. Sudan (with gum arabic) and South Africa (medicinal plants) are reporting the region's largest outputs.

North and South America: Other plant products, food, and exudates are the major categories, mainly reported by USA, Canada, Brazil, and Colombia.

Oceania: Food and raw materials for construction are the major NWFP outputs.

It is interesting to recall that most present agricultural crops and domestic animals evolved from wild growing species once gathered or hunted by man. Gradually, in the course of human history and agricultural developments, these plant and animal species provided the source genetic material for domestication programs (examples of tropical crops include: coffee, cocoa, oil palm, rubber tree, tea, vanilla, cloves, bamboo, banana, mango, etc.). The domestication process of useful plants and wild animals from tropical forests is still ongoing. Some well known and recently domesticated tropical forest species are macadamia nuts (*Macadamia integrifolia*), star fruit (*Averrhoa carambola*), cupuacu (*Theobroma grandiflorum*), durian (*Durio zibethinus*), and animal species like paca (*Agouti paca*) and iguanas *(Iguana iguana*).

In the tropics, NWFP are coming from a wide range of production systems. This can be as simple as the gathering of wild resources in and outside forests by indigenous peoples, such as the case of Brazil nuts in the Amazon with an average yearly production of some 35,000 tons (Vantomme 2004). More intensive production forms include domestication and crop intensification through agroforestry, farming, or industrial scale tree crop plantations, such as with the 2.7 million ha of moso bamboo (*Phyllostachys pubescens*) plantations in China (Perez et al. 1999). Another example is from the rubber tree (*Hevea brasiliensis*) in Brazil, of which natural stands are still tapped for the commercial production of latex by local forest dependent people in the Amazon. In addition, industrial corporations operate industrial plantations of improved rubber tree clones. This practice is not limited to Brazil. The same goes for oil palm in Cameroon or durian in Malaysia. These different production systems have different stakeholders applying different types of silvicultural treatments at different levels of intensity. While managing NWFP species in forests is part of "silviculture," the more intensive production systems through farming and tree cropping are part of farming and horticultural science.

The major producing countries of tropical NWFP are China, India, Indonesia, and Brazil (FAO 2005). At least 150 NWFP are of major significance in international trade. In addition, a large number of botanicals (approximately some 4,000 species) enter international markets. Most NWFPs are traded in rather small quantities, but some such as ginseng roots, natural honey, pine nuts and walnuts, gum turpentine, rosin, rattan, and gum arabic reach substantial levels (FAO 1995).

10.3 Silviculture of Major Tropical NWFP

New silvicultural approaches for the production of NWFP in tropical forests are presented by describing key examples from Amazonia, tropical Africa, and Asia.

Açaí (a local common name that includes the species *Euterpe oleracea*, *E. precatoria*, and some other *Euterpe* spp.) is among the best known NWFP from the Amazon forests. The multistemmed palm species occurs in large quantities in floodplains and is utilized primarily for palmhearts (palmito) and edible fruits. Acai palms grow up to 20 m high with a d.b.h. (diameter at breast height) of up to 40 cm. They grow gregariously with often more then 100 adults/ha. Fruit maturation is from September until December with birds and rodents as seed dispersers (Kageyama et al. 2004; Rocha and Viana 2004). The abundance and rapid growth of *E. oleracea* was noticed early on by indigenous peoples. Pre-Columbian silviculture and indigenous management of açai was limited mainly to harvesting wild stands combined with some cultivation of wildlings at their homesteads to meet their subsistence needs. This extraction system continued undisturbed until approximately the 1970s. Since then, the commercial demand for açai palmito and for the juice made of its seeds has taken off, as the floodplain natural forests have become increasingly accessible. This accessibility is due to the logging of Virola timbers (*Virola surinamensis* and other *Virola* spp.) to supply the growing plywood industry in the Amazon. The logging activities in the Amazon floodplains paved the way for the expansion of a palmito canning industry. Once natural floodplain forests were logged of their virola trees, which dominated the forest's upper canopy layer. The logging disturbances and the increased light penetration in the forest further enhanced growing conditions for açai palms. With the presence of open access natural stands, the silvicultural approach now being implemented is a large-scale rotation system. This is being imposed either by companies or by the rural communities owning or managing these lands. They allow harvested areas to regenerate, while intensifying extraction in undisturbed or recovered palm stands (Taylor 2005). Although plantations with improved genotypes are emerging, the extraction system of managed natural açai stands still provides the bulk of the palmito and fruit harvests because harvesting natural stands is cheaper than establishing plantations.

Gum arabic is a natural gum that can be extracted from acacia trees (mainly *Acacia senegal* and *Acacia seyal*). The gum is harvested commercially from wild and planted trees throughout the Sahel from Senegal to Somalia. The acacia tree is very common in the tropical dry savannahs of Africa and the trees regenerate easily (Fagg and Allison 2004). Traditionally, pastoralists gathered the gum during their seasonal transhumance passage through acacia stands. In some regions of Sudan, the world's largest producer of gum arabic (World Bank Policy Note 2007), farmers tolerate the spontaneous regeneration of acacia seedlings in their slash and burn farm fields and they leave approximately one sapling/sprout every 10 m to grow. After 2 or 3 years of farming, soil fertility drops, and the plot is abandoned to farm a new field elsewhere. The young saplings grow quickly and form an open acacia stand in 4–5 years (World Agroforestry Center 2009), during which, grazing of livestock in these newly established young acacia stands is not allowed. Afterwards, seasonal migrant herders with their livestock are allowed to graze the area and browse on the trees. The herders wound the bark of stems and large branches to stimulate the trees to exudate the gum for collection and sale in local markets. The grazing activity and gum harvesting continues for some 10–15 years until the trees have restored enough fertility in the soil and are large enough to cut down for making charcoal. After this harvest, the farmers take back the fields for farming and a new cycle begins (Rahim et al. 2005). The applied silvicultural rotation cycle for these acacia trees is 15–20 years, pending the availability of sufficient land and favorable weather conditions. This silvicultural system is a nice example of symbiosis between farmers and pastoralists to grow trees on farm plots, which will restore soil fertility, while at the same time provide fodder, gum, fuel wood, and charcoal (Boffa 1999).

In tropical Asia, rattan is a very common NWFP. Rattan or “canes” are the stems of palm trees that grow like lianas in humid tropical forests. There are 13 different genera of rattans that include in all some 600 species (Dransfield 2002). The stems are processed for a wide variety of uses such as baskets, nets, and ropes, but also for furniture making. Rattan palms were once abundant in the forests, but unsustainably high harvesting levels and deforestation have significantly reduced the available stock in the forests. The silvicultural response to a dwindling rattan supply has been twofold, i.e., by “managing” wild growing rattans in natural forests and by establishing plantations of them (Sastry 2002). Rattan first became included into forest management activities with silvicultural interventions to promote its growth (by thinning clumps which were too dense and by leaving a few stems per clump) and by regulating its extraction through licenses. Forest inventory maps now geographically relate rattan stock in the forest by species and size (number of stems per clump) and the harvesting of the mature stems is planned around a 15-year cycle. Also, the forest management plan is designed to leave support trees close by for the rattan stems to climb up in search of light. Where needed, enrichment planting of rattan (particularly with species that provide large diameter cane) in logged-over forests can be done (Tesoro 2002). Plantations of rattan species in cleared forest areas have been tried mainly in Malaysia and Indonesia (eventually in combination with planting rubber trees as support). However, the complexities of growing rattan as a plantation crop are still creating major technical and silvicultural challenges for research to solve. These challenges include, the choice of most suitable support trees, how to shorten the 5–7 year long rosetta stage in which the newly planted rattans remain without growth of their stems, and how to improve the very laborious work of harvesting the cane stems which can be from 20 to 50 m long and contain numerous spines. In addition, rural people are less interested in rattan plantations, as they have lower economic returns than other cash crops such as oil palm or rubber. Therefore, managing rattans in natural forests still prevails as the most suitable option for cane production (Ali and Raja Barizan 2001; FAO 2002).

10.4 Opportunities and Challenges of Introducing Tropical NWFP Silviculture

Opportunities for including NWFP into forest management are now endorsed by various international conventions, such as the Convention on Biological Diversity (CBD) and forest-related initiatives, such as criteria and indicators for SFM and by various national forest programs of many countries. In addition, governments, development agencies, and particularly NGOs have high expectations that an improved use of NWFP can contribute to better conservation of biodiversity and to social welfare programs like poverty alleviation of forest-dependent peoples. Since the 1990s, the importance given to the environmental and social components of SFM has increased dramatically. This has further raised the necessity of incorporating NWFP-related objectives within silvicultural interventions in tropical forests, particularly timber concession arrangements and their certification. However, there is still a significant lack of basic information about the population dynamics of the major NWFP species in tropical forests, on their silvicultural needs, and sustainable management and harvesting prescriptions. Although producers and collectors may have ancestral knowledge on managing different NWFP species found in tropical forests for their subsistence needs, that knowledge is no longer enough to cope with growing commercial demand and/or changes in consumer preferences regarding the quality of the products.

Constraints and challenges for applying silvicultural interventions in tropical forests, which are aimed at enhancing growth of NWFP producing species, can be summarized as follows.

In natural tropical forests, using silviculture to both improve timber production and implement interventions to sustain the supply of commercial NWFP species is a very complex issue. This is due to the fact that there are so many species to deal with and so many different life forms of the NWFP (leaves, roots, bark, fruits, nuts, etc.). Also the intended silvicultural interventions may in many cases be conflicting among themselves. For example, trees providing edible nuts, bark, fruits, leaves, or caterpillars would require wide spacing, full sunlight expositions, and pruning to stimulate the development of a large and low hanging crown to facilitate harvesting. Timber production silviculture would basically aim to do the opposite and plans to concentrate wood growth in a long bole by keepings stands very closed to produce knot free boles. Different stakeholders can have conflicting interests. For example, forest concessionaires in central Africa are interested in logging sapelli trees (*Entandrophragma cylindricum*) for their beautiful timber, while the local population prefers to keep the sapelli trees alive to collect caterpillars feeding on its leaves (ODI 2004). On the other hand, when the commercial value of a given NWFP gets so high, as with cinnamon (the inner bark from the *Cinnamomum verum*), other trees in the forest will be gradually removed and be replaced by the more lucrative tree transforming the forest into a (privately owned) tree garden (Michon et al. 1986). In most cases, the domestication of a NWFP species usually results in removing the original forest in order to set up plantations of the desired species. This can be seen with the production of bamboo in China or for cardamom and cloves in India. This leaves little of the previously available biodiversity.

Global trade of NWFP leads to increased competition to improve the productivity and quality of the goods and their substitutes. Commercially successful NWFP are increasingly being produced in farming systems. Domestication and production of NWFP through farming ultimately reduces the economic value of these species growing spontaneously in the forest. For most commercial NWFP, the gatherers tend to become more marginalized as trade volumes increase and the production becomes more intensified through farming systems. In those cases, the benefits usually go to the traders and farmers.

Forest Certification Schemes, such as the Fair Trade and Sustainable Forest Management Certification, can provide protection for the forest gatherers' production from certified areas. However, the certification benefits cannot be easily expanded to cover all producers in the sector (Vantomme and Walter 2003). Monitoring the social impact of NWFP domestication on the gatherers' livelihoods is essential. The certification of forests and wood products is well advanced with several schemes of certification firmly in place. However, the development of technical and social criteria and standards for certification of NWFP is still in its infancy. Certifications also have market limits. They are aimed at international markets, while most NWFP are mainly traded at local and national levels. Although the market share of SFM certified, organic, or "fair-traded" NWFP is increasing, it is still insignificant.

10.5 Conclusions

Planning silvicultural interventions for NWFP species within basic forest management is still rarely done. It requires a broad multifaceted approach to integrate the many and often conflicting user demands for food, fiber, energy, health, and recreational goods. It also implies the active participation of a much wider range of stakeholders than when dealing with timber only. Knowledge of NWFP management techniques and the related methodologies used to assess the impacts they play on the sustainability of the resource is still needed. In addition, the impact these methods have on the livelihoods of the resource gatherers over time, still needs to be assessed to a greater level. International organizations such as the FAO can play a key role in establishing the silviculture of NWFP species into socially equitable SFM practices. This can be accomplished through raising awareness and building the required technical and institutional capacities needed by the countries.

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Chapter 11
Modelling Forest Growth and Finance: Often Disregarded Tools in Tropical Land Management

Thomas Knoke and Andreas Huth

Abstract While many studies analyse patterns of tropical land management with a backward-oriented approach that utilises data of the past, we propose to consider future-oriented modelling approaches to find sustainable land-use options. This proposal is illustrated with application examples for advanced growth modelling in tropical forests, a short overview on financial performance analyses for tropical land uses, and the introduction of a new modelling approach. This modelling approach sees tropical land management as a financial portfolio of land-use options. Its advantage is the ability to make transparent effects of financial risk reduction that arise from mixing forestry and agriculture-based land-use options. The approach thus does not analyse land uses as stand-alone options, like most other analyses do. The land-use portfolio modelling shows that sustainable land use may also be financially attractive for farmers, if abandoned farm lands are reforested (with a native tree species in our case) and sustainable management in natural forests is carried out. We conclude that the combination of advanced growth with sound financial modelling may lead to improved bioeconomic models. Developed bioeconomic models are necessary to increase the biological realism and acceptability of the results obtained.

11.1 Introduction

Studies on tropical land use often describe given land-use patterns, their possible reasons, and consequences (e.g. Pichón 1996, 1997, Marquette 1998; Sierra and Stallings 1998; Paulsch et al. 2001; Browder et al. 2004; Pohle et al. 2009).

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In contrast, the modelling of consequences resulting from land-use activities is, in fact, rather seldom. Given the problems related to tropical deforestation (e.g. Stern 2006; Malhi et al. 2008), the lack of advanced land-use modelling approaches forms a great research challenge for the future.

The backward-oriented analysis of data recorded from the past certainly helps understanding the current land-use patterns. However, future-oriented forest growth and financial modelling of ecosystems may reveal new opportunities to control undesired and destructive management activities, such as ongoing deforestation. In this line, the inclusion of silvicultural activities into land-use modelling and the consideration of financial consequences bears potential to come to an improved bioeconomic modelling (Carpentier et al. 2000), which will primarily focus on the perspective of tropical land users.

In this chapter, we report briefly about the development of forest growth modelling and demonstrate one modelling example to show the opportunities of modern growth models for the tropics. Subsequently, we review some exemplary results on the financial attractiveness of several silvicultural options collected from the literature. We finish with a new modelling approach and conclusions on how to possibly achieve sustainable land use in tropical ecosystems.

11.2 Growth Modelling of Tropical Forests

Forest models aim at investigating long-term impacts of logging and silvicultural treatment. Thus, if the models work accurate enough, they represent an important tool to assess key aspects of sustainable forest management (SFM). Moreover, they form a crucial precondition for financial valuation. If we want to evaluate the sustainability of management strategies we need models as tools, which allow assessing the long-term dynamics of managed forests.

The first forest models have been very cryptic and had no user interface (e.g. Botkin et al. 1972). New forest models can take advantage of the progress in informatics, provide comfortable user interfaces, and have also the possibility to visualise predicted forest dynamics and used relations. Nevertheless, synthesis of the main dynamic processes in forests is still a laborious task, especially if we assess forests with more than one tree species. Please note that this is not caused by the models, but by the complexity of ecological processes in forests.

Forest modelling has been around since the 1970s providing a valuable approach to understand forest growth and ecology from a mechanistic standpoint (Shugart 1998; Bugmann 2001; Pretzsch 2007; Jeltsch et al. 2008). Due to the fact that empirical records of forest dynamics are limited in time and space, models are useful tools to scale up the available measurements.

Development of forests has been successfully analysed, for example with forest gap models, and the Sortie model for temperate and boreal forests (e.g. Shugart 1998; Pacala et al. 1996; Bugmann 2001). These models describe growth, mortality, regeneration, and competition processes for the different tree species parameterised with field data from long-term research plots. (The main challenge here is to find adequate statistical relations to describe a large range of competition situations.) They have been used to investigate, e.g. regeneration after disturbances and the role of species for the structure and succession dynamics of forests.

Since the 1990s, more and more process-based forest models have been developed (e.g. Landsberg and Waring 1997; Porte and Bartelink 2002). In this approach, forest growth is described on the basis of a carbon balance by modelling ecophysiological processes, especially photosynthesis of leaves, respiration, and allocation. These models also include a detailed description of light competition. There is a growing amount of available physiological data, which allows the application of this model type to more and more regions.

The number of forest models for tropical rain forests is small mainly due to the complexity of these forests (e.g. high tree species richness, tree age data is missing; Vanclay 1994, 1995; Sist et al. 2003). The FORMIND and FORMIX3 model family introduced the process-based approach to tropical forests and has been applied in many different sites (e.g. Huth and Ditzer 2000, 2001; Kammesheidt et al. 2001; Köhler et al. 2003; Köhler and Huth 2004; Rüger et al. 2008; Groeneveld et al. 2009; Gutierrez et al. 2011; Dislich et al. 2010).

For every individual tree, a set of quantities is calculated every time step in the FORMIX model, e.g. tree height, stem diameter, biomass, and leaf area index. The change of these variables is determined by applying fundamental ecological processes as growth, competition, regeneration, mortality, and disturbance. The model follows the gap-model approach, i.e. the forest stand is divided into patches, which have the size of treefall gaps. In each gap the vertical leaf distribution and light climate is calculated, which allows a detailed description of the light competition situation for each tree. This multi-layer description of the canopy is in contrast to the big leaf approach in global vegetation models (Friend 2001) or the use of aggregated competition indexes in statistical forest models (e.g. Kohyama et al. 2003; Vanclay 1995).

The concept of plant functional types is used to cope with the tree species richness in tropical forests (Smith and Shugart 1997; Köhler et al. 2000; Picard and Franc 2003). Tree species are classified into plant functional types based on physical attributes (e.g. maximum potential height and shade tolerance). The model belongs to the FORMIND and FORMIX3 forest model family, which has been extensively tested against field data (e.g. Huth and Ditzer 2000; Rüger et al. 2008; Groeneveld et al. 2009; Gutierrez et al. 2011) and applied by other research groups (Sato 2009; Pinard and Cropper 2000).

11.2.1 Example of Application: Analysis of Logging Scenarios for Dipterocarp Rainforest

We used the FORMIX3 model to analyse the long-term impacts of different logging strategies on yield, forest structure, and species composition. We distinguished conventional logging with high damages and low impact logging with low damages and analysed different cutting cycles for Dipterocarp rainforests in Deramakot, Southeast Asia.

The Deramakot Forest Reserve is situated in North Borneo (Malaysia, 5°25′ N, 117°30′ E). The region has the perhumid climate typical of the inner tropics (mean annual temperature is 27° and mean annual rainfall is about 3,500 mm). The prevailing forest type is Dipterocarp lowland forest. The forest remained essentially undisturbed until this century and commercial logging started in 1956.

For conventional logging with a cutting cycle of 20 years, each logging operation can be recognised due to the sudden decrease of total standing volume in the simulation (Fig. 11.1a). This decrease is composed of the harvested stem volume and losses due to damages. The cutting cycle of 20 years is so short that 20 years after the first logging operations, there are not enough harvestable trees in the forest to allow a new logging operation. Thus, the next logging is carried out later, at year 40. The same situation occurs at the years 80, 100, 120, 160, 200, 220, 300, 320, and 380. The high variation of yield per operation is strong evidence that the forest is overexploited (Fig. 11.1a right). After each logging operation the forest regrowth is constituted largely by increased growth of Macaranga species (group 3) compared

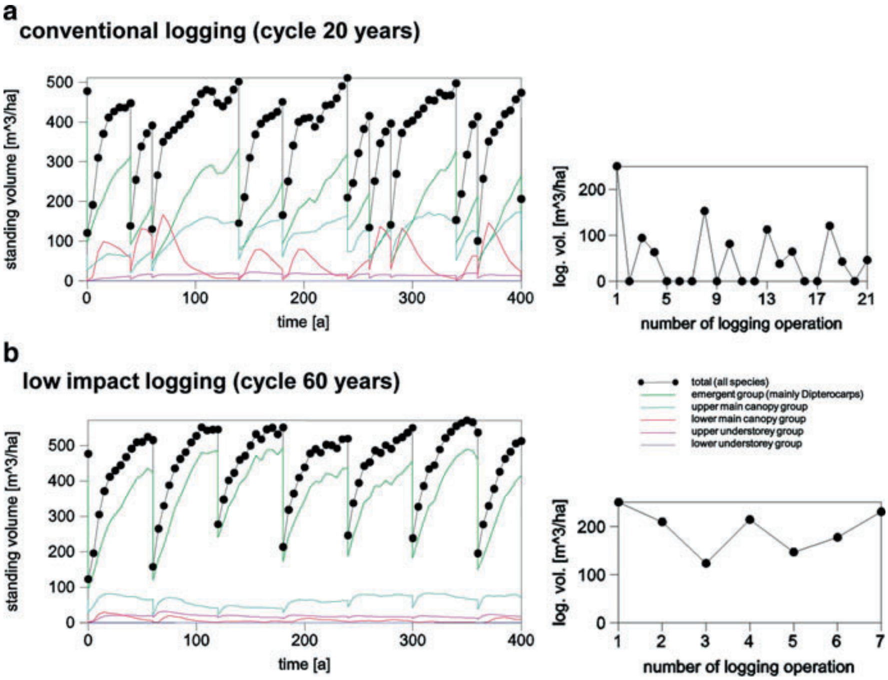

Fig. 11.1 Example for the simulation of different types of logging scenarios. Results are for Dipterocarp rainforest in South East Asia, Deramakot forest reserve assuming conventional logging with a cutting cycle of 20 years (**a**) and reduced impact logging with a cutting cycle of 60 years (**b**). *Left*: Standing timber volume over time for different species groups and all trees (above 10 cm diameter). *Right*: Logged volume over the number of the current logging operation

to the composition in primary forest (initial state). But also the species composition of the Dipterocarp species (groups 1 and 2) shifts to smaller and more light-demanding Dipterocarps (group 2).

In the low impact scenario with a cutting cycle of 60 years, the volume of all trees and the volume of harvestable trees reach nearly the same value as in unlogged forests before each logging event (Fig. 11.1b). Species composition remains rather stable. The logged volumes per cut still show some fluctuations (Fig. 11.1b right).

As shown for this example, growth models can help derive the sustainable timber harvest. A sustainable harvest may be derived for plantations with native tree species as well. Based on this information, financial valuation may be carried out to assess the financial attractiveness of silvicultural options.

11.3 Financial Modelling of Tropical Land Use

Among the manifold of management activities for tropical lands, we identify various silvicultural options (Fig. 11.2). We will first review studies about the profitability of natural forest management and land reforestation, as options

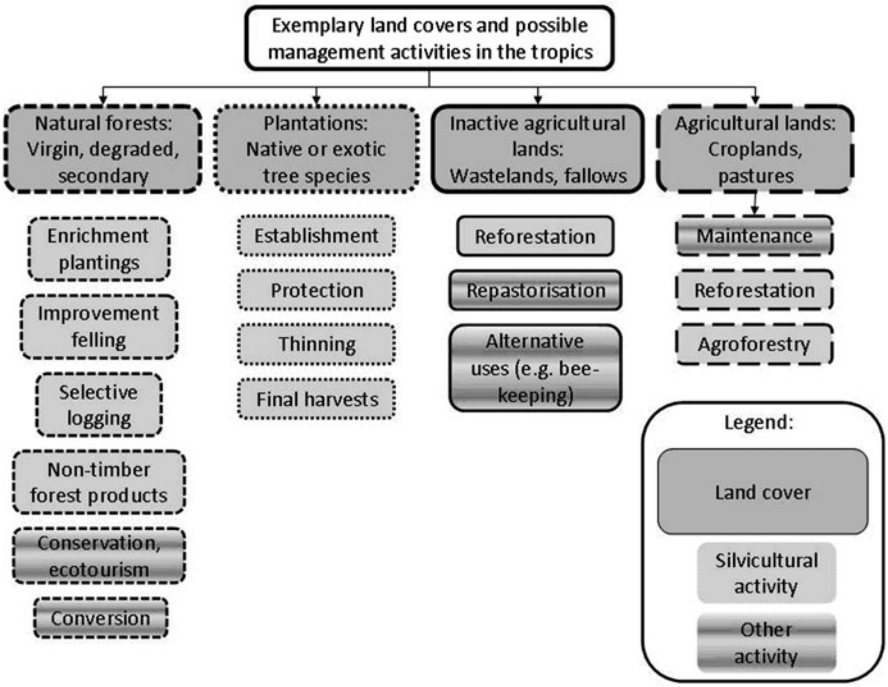

Fig. 11.2 Exemplary management activities on tropical lands

among the most important silvicultural management opportunities. Following this short overview, we will demonstrate a new, model-based approach in analysing and designing sustainable land-use concepts, which include both mentioned silvicultural options.

11.3.1 Profitability of Natural Forest Management

Given a flow of sustainably harvestable timber volumes, which may be predicted by a growth model or derived by data recorded from the past, one can calculate the profitability of SFM of natural forests and other forest management strategies (see Pearce et al. 2003). Various studies in this field show that net revenues from SFM in natural tropical forests are rather small; they may range between US$ 32 and 153 ha^{-1} year^{-1}(Table 11.1). In comparison to alternatives, mostly agricultural land uses, we regularly find, in part overwhelmingly, greater net revenues between US$ 72 and 468 ha^{-1} year^{-1} for the alternative options. From the perspective of a farmer, this rather simple comparison (reasons why simple given below) makes clear why tropical land use often excludes SFM.

Table 11.1 Net revenues from sustainable natural forest management (SFM) compared to those of alternative land-uses (ALU)

Country	Annualised net revenue SFM (US$ ha^{-1} year^{-1})	Annualised net revenue ALU (US$ ha^{-1} year^{-1})	References	Comment
Brazil	69[a]	58[a] (agriculture with no intensification) 90[a] (agriculture with intensification)	Carpentier et al. (2000)	Sustainable yearly harvest, labour costs in forest 10[a] R$ day^{-1}
Ecuador	31	72 (conversion to cattle pasture, 20-year cycle)	Knoke et al. (2009a)	Sustainable yearly harvest, 5% interest for agriculture
Cameroon	32	154 (conversion to small-scale agriculture) 178 (conversion to palm oil)	Studies cited by Turner et al. (2003)[b]	10% interest, 32-year cycle
Sri Lanka	123	468 (cultivation of tea)		8% interest, 20-year cycle
Malaysia	153	189 (unsustainable timber logging)		8% interest, 100-year cycle

[a]Brazilian real

[b]We calculated annualised net revenues as annuities from net present values (i.e. the sum of all appropriately discounted positive and negative financial flows) reported in Turner et al. (2003)

If net revenues to be generated by SFM are very low, an effective way towards long-term conservation of tropical forests is the allocation of market value to standing tropical trees. For example, the allocation of market-based carbon values to standing timber, where deforestation is to be avoided, may be a great opportunity to save tropical forests. However, the expected carbon values alone may often not cover the full land opportunity costs that farmers face when accepting the maintenance of tropical forests (Knoke et al. 2009b). As a consequence, the implementation of carbon values, for example through compensation payments financed by international carbon markets, will probably only be successful in combination with sustainable land-use concepts, which include timber-based SFM (Pearce and Pearce 2001). If, in fact, no or only small financial value is generated by the standing forests and their management, the natural forest has no direct advantage for the farmers, other than as insurance. Insurance means that the harvest of still existing standing timber of natural forests can serve to compensate for possible losses, if land-use options other than SFM perform more poorly than expected (Knoke et al. 2009a). The allocation of direct and permanent value to standing tropical forests – be it through ecosystem products (e.g. non-timber forest products) or services – is thus one of the most challenging future tasks to which tropical silviculture can contribute. However, the available analyses show that the maintenance of tropical forests will often be financially unattractive for tropical land users, given that ecosystem services are financially ignored and that SFM of tropical forests is seen as a stand-alone operation (see below for a detailed discussion of the latter point).

11.3.2 Plantation Forestry with Exotics vs. Reforestation with Native Tree Species

Industrial plantations, mainly established with exotic tree species, form about 6% of the world's forested area, but nevertheless stand for more than 25% of the world's timber production (Siry and Cubbage 2003). Tropical countries, such as Brazil, grow for example Eucalyptus or Pine plantations with internal rates of return (IRR) from 15% up to more than 20% (Cubbage et al. 2007). *Gmelina arborea* even reaches IRR of more than 30% on the Philippines (Harrison et al. 2005), while native tree species result in IRR of maximally 7% in Philippines' study. The annualised net revenues of plantations with exotics (US\$ 248–343 ha^{-1} year^{-1}) go far beyond those of SFM of natural forests and also exceed net revenues of plantations with native tree species, such as Andean alder (*Alnus acuminata*) or Laurel (*Cordia alliodora*) tested for Ecuador (Table 11.2).[1]

[1]Note that various interest rates applied for calculations in Table 11.2 hinder a sound comparison. If we keep in mind that great interest rates reduce annualized net revenues substantially, the comparison between exotics and natives underlines even stronger the superiority of the exotics.

Table 11.2 Net revenues from forest plantations in the tropics (without costs for land, e = exotic, n = native, IRR = internal rate of return)

Country	Tree species	Annualised net revenue (US \$ ha^{-1} year^{-1})	References	Comment
Brazil	*Pinus taeda* (e)	248	Cubbage et al. (2007)[a]	8% interest
	Eucalyptus grandis (e)	434		
Ecuador	*Alnus acuminata* (n)	58	Knoke et al. (2009a)[b]	5% interest, 20-year rotation
		2–25	Dunn et al. (1990)[c]	15% interest, 20-year rotation, Alnus as an agroforestry component
	Cordia alliodora (n)	6	Olschewski and Benitez (2005)	7% interest, 2 15-year rotations
Philippines	*Acacia mangium* (e)	50	Harrison et al. (2005)[d]	15% interest, 10-year rotation
	Gmelina arborea (e)	123		15% interest, 10-year rotation
	Eucalyptus deglupta (n/e)	44		15% interest, 20-year rotation
	Native species	Negative, maximum IRR 7%		15% interest, various rotations

[a]We calculated annualised net revenues as annuities from land expectation values (calculations considering an unlimited time horizon) reported in Cubbage et al. (2007)
[b]We calculated annualised net revenues as annuities from net present value (calculations considering a limited time horizon), we ignored risks considered by Knoke et al. (2009a) for the sake of comparability
[c]We calculated annualised net revenues as annuities from net present value (calculations considering a limited time horizon), we converted 166 sucres to the US dollar. Note that the reported net revenue is additional to pasture net revenues, which can be obtained from the same area. Pasture net revenues were, however, not reported
[d]We calculated annualised net revenues as annuities from net present value (calculations considering a limited time horizon), and converted 50 Philippines pesos to the US dollar

Environmental concern about economic losses in social ecosystem services, as probably induced by intensive plantation forestry with exotic tree species, will certainly outweigh short-term private financial benefits from this strategy. Particularly, if timber companies convert native forests into intensively managed plantations and thus destroy great biodiversity, release much carbon into the atmosphere, and possibly deplete site fertility in the long run, this option cannot be considered sustainable. However, until today the internalisation of the mentioned externalities has mainly not been carried out. Land-use practices in many parts of the world are thus still mainly driven by their expected short-term financial consequences (Pearce et al. 2003); a fact that actually still favours exotic tree plantations compared to natives.

11.3.3 Selective Logging, Reforestation, and Pasturing Combined in a Land-Use Portfolio

Despite the enormous IRRs achievable with exotic trees, an intensive land management, as necessary for a successful “industrial-style” timber production, is often not a favourable option for small land holders. Small holder households may often not raise the necessary investments and work intensity to establish large-scale exotic tree plantations. In the situation of small holders, *diversification* instead *intensification* of land use may be an adequate option to improve their livelihood. For such a diversification strategy, a clever reforestation based on robust and site-adapted native tree species, designed to re-integrate degraded, unproductive “wastelands” (i.e. abandoned pasture lands sensu Silver et al. 2000) into the production process, may be a key option to achieve private benefits for farmers.

In a case study carried out for a virtual farm located in South Ecuador (see Knoke et al. 2009b for details) to better understand financial reasons for the current land-use patterns, the agricultural land-use option “single pasturing” was most profitable among three mutually exclusive land-use options (the farm consisted of 20 ha pastures and 40 ha natural forests). However, this result (Fig. 11.3 and Table 11.1) was obtained only when considering all land-use options isolated and ignoring their financial risks. Financial risk may be measured as the standard deviation of the simulated average net present value (NPV). NPV results from applying the “discounted-cash-flow” method, when summing up all appropriately discounted future financial flows of land management (with 5% discount rate applied in the above study). The financial risk of single pasturing was medium: On the one hand, its risk was double compared to selective logging in the natural forests; on the other hand

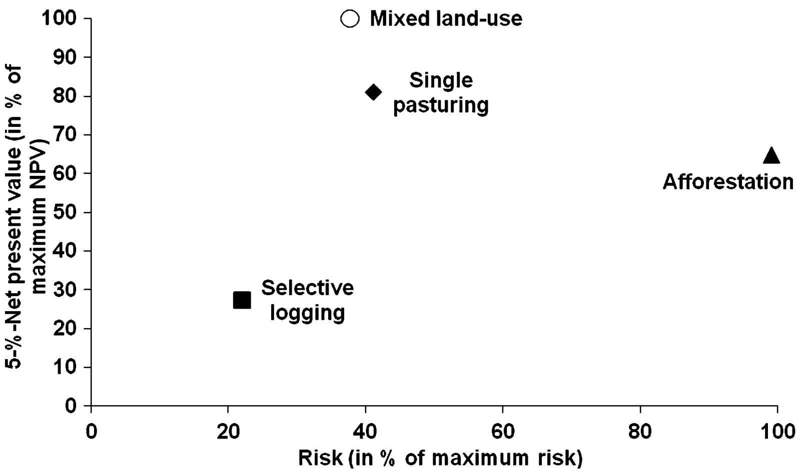

Fig. 11.3 Relative net present values for three land-use options in South Ecuador and their standard deviation, all values relative to the maximum values (data based on simulations carried out by Knoke et al. 2009b, with alteration)

it amounted only to around 40% the risk of reforestation (Fig. 11.3). Reforestation bears higher financial risk, because we assumed only one final harvest operation for this option and excluded thinning operations for this example. We thus have no compensation effects with regards to market-price volatility, which occur for the other land-use options, where yearly financial flows were considered (see Knoke et al. 2001 for a detailed explanation of this effect). In summary, we can say that the results of the shortly presented financial model calculation are well in line with the actual land use in South Ecuador, which mainly consists of cattle pasturing (Paulsch et al. 2001).

However, a direct comparison of mutually exclusive land-use alternatives is too simple of an approach. It ignores diversification options and the fact that most landholders are risk averters (Pichón 1996). Studies from the fields of forestry (Knoke 2008), fishery (Edwards et al. 2004), and managed grasslands (Koellner and Schmitz 2006) have shown that the consideration of less profitable options (which are selective logging and reforestation in the case presented above) is a reasonable option, if diversification effects occur, such as risk compensation from various products. In Fig. 11.3 we find the mixed land-use option, consisting of a combination of three land-use options, involved with the highest NPV (computed for an interest of 5% over 40 years) and only moderate risk (even slightly lower than single pasturing). The better profitability of the farm under mixed land use is mainly a result of the reforestation of areas, which have fallen unmanaged and abandoned because of degradation processes. In contrast to classical land management, the production at farm level under the mixed land use is immediately diversified into agricultural and forestry products and thus land use is stabilised (Fig. 11.4), whereas

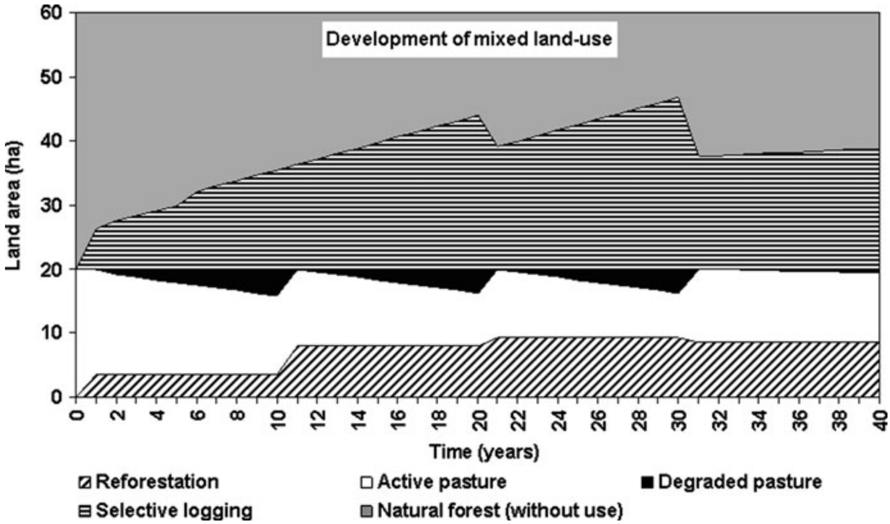

Fig. 11.4 Simulated development of the areas for three land-use options over 40 years under a mixed land-use concept for a virtual farm in South Ecuador (adopted from Knoke et al. 2009b, with alterations)

under classical single pasturing we face considerable pasture degradation and accumulation of wastelands (Knoke et al. 2009a, b).

Forest uses include reforestation, although involved with comparatively great risk. This disadvantage is balanced out in a portfolio of land uses by the risks of pasturing, which are independent from that of forestry. A small part of the pasture area will be reforested with Andean alder, a species native to the sites in South Ecuador, already at the beginning of the considered period with increasing areas when degraded pastures accumulate. Note that a reconversion of a part of the Alder plantations after 20 years back to pasture management was considered.

Besides the establishment of Alder plantations, selective but sustainable logging (SFM) was considered in a part of the natural forests (Fig. 11.4). The freed working capacity by means of reduced pasture area was allocated to selective logging activities in the natural forest area to compensate for missing early revenues from reforested pastures. This mixture of land uses, diversified into agriculture and forestry, stabilises net revenues and accumulates substantial monetary value by the natural growth of the newly established Alder plantations until they are harvested at age 20. In the meantime they deliver net revenues from thinning (from year 11 onwards). After the harvesting of the tree plantations, the areas were again used for pasturing. In summary, the mixed land-use concept stabilises net revenues, enhances livelihood for the farmers, and avoids deforestation, while single pasturing would result in a yearly deforestation rate of 1.3%.

However, forest modelling was based on rather simple assumptions in the reported case study, with transition probabilities for natural forest trees to move from one size class into the next bigger size class (estimated from past diameter increments) and mortality recorded in experimental plots (see Knoke et al. 2009b). Reforestation growth was estimated mainly based on existing references. We would like to point out that a future combination of the advanced forest growth modelling reported above with the demonstrated kind of financial land-use modelling will strengthen the validity of the results and thus bears great potential.

Despite this limitation, the above example shows that reforestation with natives and SFM in natural forests can show great advantage from the perspective of sustainability, because both options fit well into a portfolio of land uses and thus decrease the overall risk. These effects may outweigh relatively poor financial performance resulting from stand-alone financial analyses. When concentrating reforestation only on abandoned agricultural lands (so-called wastelands), this option does not compete with natural forests for land and can be obtained in addition to the classical land use. Moreover, site rehabilitation and consolidation from the effects of native tree species stress the advantages of this option. However, to convince farmers about the reforestation option, when based on native tree species, solid financial modelling is necessary. Particularly the fact of uncertainty needs being considered, since diversification effects between forestry and agricultural products will reduce risks of land management and thus also the demand for agricultural and plantation land. Uncertainty sensitive modelling reveals more realistic and thus rather acceptable results, compared to classical deterministic calculation (Knoke and Seifert 2008). It enables estimating financial consequences in a more realistic way, as it may include all available land-use options, and thus can make reforestation with native tree species and SFM in natural forests viable alternative for farmers (Knoke et al. 2008; Knoke et al. 2009a), while pointing out risk-reducing benefits obtained from less profitable tree species (Knoke 2008) and land uses. When combining the financial mixed land-use model with compensation payments for environmental services, also some limitations, such as the financing of the investment necessary for reforestation, may be eliminated (Knoke et al. 2009b).

11.4 Conclusions

We could show that a comparison of isolated and mutually exclusive land-use alternatives is probably inadequate to support land-use decisions. Based on a modelling approach, fed by growth and production data for forestry and agricultural land-use options, which includes financial risks modelling, the design of sustainable land-use options may be effectively improved. The presented approach may now be applied to improve land use in other parts of the world and to analyse other land-use concepts. We expect great demand for this kind of analysis, as socio-economic aspects of natural diversification are much underrepresented in other studies (Rice 2008). We conclude that linking modern growth modelling with advanced financial valuations, resulting in sound bioeconomic models, would increase the biological realism of economic studies and the acceptability of their conclusions. Improved bioeconomic modelling, based on solid financial theory and advanced growth models, thus bears great potential in supporting sustainable land use.

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Part IV
Silviculture in (Semi-)Natural Humid Forests

Chapter 12
Review
The Ecology, Silviculture, and Use of Tropical Wet Forests with Special Emphasis on Timber Rich Types

Mark S. Ashton and Jefferson S. Hall

Abstract Tropical wet forests were once extensive throughout the high rainfall (>2,000 mm per year) areas of the equatorial latitudes between 23 N and 23 S. Most of the of the forest in Asia, West Africa, Central America, and northern Latin America is either degraded from logging or has been cleared for agriculture. However, extensive forest remains in Central Africa, Amazonia, and parts of Asia. This chapter is a review of the autecology and forest dynamics of the timber-rich forest types of these regions, with special attention to work done on the silviculture and sustainable management of such forests. New insights are provided into management particularly in integrating nontimber forest products with timber for higher economic values that are competitive compared to other land uses, and in providing a strong case for episodic regeneration methods and their variants based on recent studies of the ecology of the tree species and their forest dynamics. More sophisticated silvicultural methods need to be implemented if the remaining forests are to be sustainably managed for timber and nontimber products. It is clear that silvicultural research both on timber and on nontimber species is lacking, and what information there is, is concentrated on only a few species and in particular regions. What is now being realized is that these forests provide much more than wood and nonwood products. Such realized provisions now include: (1) water for downstream urban drinking supply, hydroelectricity, and for irrigated agriculture; (2) climate mitigation through action as a store house and sequestration site for carbon; and (3) as moderator of surface temperatures and variability in rainfall – functions that have global consequences if these forests are destroyed. It is therefore in our own best interests to protect and sustainably manage them.

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Keywords Amazon · Central Africa · Dipterocarps · Entandrophragma · Southeast Asia · Swietenia

12.1 Introduction

This chapter is a review intended to focus primarily on the timber-rich forest types of the wet tropics. In doing so the paper leaves out considerable regions largely in Latin America and Asia that reflect montane and coastal forests that have been covered by others in this book (see Chap. 33). It should also be regarded as an introduction to the ecology and silviculture of wet tropical forests rather than a comprehensive synthesis – which in and of itself should be a book.

This chapter can be divided into as given below.

Sections 12.2–12.4 comprise an introduction to tropical wet forests and contains:

- A definition of tropical wet forest, its regional status, extent, and land use history;
- A description of the physical environment of tropical wet forest with descriptions of climate, geology, and soils in particular; and
- A review of the autecology and description of the major timber-rich forest types.

Sections 12.5 and 12.6 summarize the uses of tropical wet forests and contains:

- An assessment of regional silvicultural history and management knowledge; and
- A description of the uses of important tropical timber and nontimber tree species.

Finally, Sects. 12.7–12.10 describe the ecology and management and contains:

- A summary of the common principles in stand dynamics that can provide a framework for the practice of silviculture and sustainable forest management;
- A description of appropriate silvicultural applications for the different forest types, timber trees, and nontimber forest products (NTFP); and
- A discussion on reduced impact logging (RIL).

The chapter ends with summary findings and conclusions.

12.2 Wet Tropical Forest Distributions and Current Extent

Tropical wet forests can be defined as those forests that typically have greater than 2,000 mm of precipitation per year and that have no dry season or one that is distinctly moderated and considerably shorter than the length of the wet season (Whitmore 1998; Richards 1996) . The forest in general is sufficiently wet to have evergreen foliage year round. Soil moisture availability is usually nonlimiting except during supraannual periods, where annual rains can be extraordinarily low or the dry season extended – usually associated with El Nino and La Nina southern oscillations.

12.2.1 Africa

For Africa, the tropical wet forest occurs along the coast of West Africa from Liberia to Ghana. It starts again in southeastern Nigeria, expanding across Cameroon around the Gulf of Guinea through Gabon and then directly east along the central Congo River basin (Richards 1996). The forests are generally more seasonal than those of Latin America with greater levels of deciduousness exhibited by some canopy species. Because of the ease of access, most of the coastal forest that spanned Cote d'Ivoire, Ghana, Nigeria, and Cameroon has been lost during the periods of French and British colonization as a result of the commercialization of plantation crops such as coffee and cocoa (1930–1960), and now oil palm (Martin and Tsardakas 1991). Forests are now largely restricted to small degraded patches. However because of the difficulty of access and civil strife, the inner core region of the Congo largely remains whole, though current timber extraction is high (1990–ongoing) in countries neighboring the Democratic Republic of Congo (DRC) (Schmidt 1991; Hall et al. 2003a, b).

12.2.2 Central and South America

The tropical wet forests were once expansive covering all of eastern Central America (Atlantic Coast) from Belize through Costa Rica and Panama, and along the Pacific coastal mountains of Columbia and northern Ecuador (Whitmore 1998). However, the majority of the forest covers the eastern part of the northern core of the South American continent – east of the Andes – and filling out the whole upper and lower basin of the Amazon, the upper Orinoco of Venezuela and across the Guyana's (Richards 1996). The wettest forests in Latin America are those straddling the Andes in the region known as the Choco on the Pacific coast range of Colombia, and the upper Amazon of Ecuador (Richards 1996). The Atlantic side region of Central America has been difficult to access by people and still remains extensively forested particularly in Panama. The core inner regions of the Amazon (except around Manaus) remain forested, as well as the upper regions bordering Colombia, Ecuador, and Peru (Nepstad et al. 2001). However, much of this forest has been logged out, with timber evacuated through the use of extensive river systems. Coastal and floodplain forests of the major rivers that flow into the Amazon (varzea) have largely been converted to agriculture (Nepstad et al. 1999). The outer periphery of the basin and the coastal Atlantic forest of Brazil has retracted considerably because of colonization schemes, and large land conversion to commercial soybean and ranching (1970–ongoing) (Laurance et al. 2001).

12.2.3 Southeast Asia

The core Asian rain forests can be considered the most moderated in seasonality largely because the land–sea margin and N–S mountain ranges serve as an important source of convectional and orographic precipitation during intermonsoonal wet seasons (Whitmore 1984). The heart of the tropical wet rain forest would be considered Borneo, Sumatra, New Guinea, and the Malay Peninsula, with other regions – parts of the Philippines, southern Thailand, northeast India/Burma, southeast Cambodia, southern Vietnam, southwest Sri Lanka, and the Western Ghats of India – making up the remaining rain forest region (Whitmore 1984).

Asia has had the longest legacy of rain forest commercialization (dating back 2,000 years) largely through maritime trade between Indian, Arab, and Chinese traders and the regional peoples (Whitmore 1984). India's and Sri Lanka's forests are now largely restricted to the mountains and uplands of the countries, where historical land use for intensive rice cultivation, private tree garden systems, and plantation agriculture (tea, rubber, coconut – 1850–1950) has a much longer legacy than elsewhere in Asia. Most of the tropical wet forest in the Philippines and Thailand is now restricted to degraded patches, first logged over, and then subsequently and incrementally converted to village agricultural projects, which often subsequently failed and are now wastelands (1940–1985) (Whitmore 1984, 1998). The Malay Peninsula had most of its lowland forest converted to plantation crops (rubber and oil palm) starting with the British (1900) but accelerating postindependence (1948) such that most of the lowlands had been converted in 1980. Substantial logged over forest remains in the highlands (Richards 1996). A similar story exists for Sarawak and Sabah, the two east Malay States on Borneo. However, for these states the land conversion of the lowland forests occurred very rapidly and recently (1970–2000). Indonesia embarked on rapid logging and land conversion of its tropical wet forests in Kalimantan (Indonesian Borneo) and Sumatra initially for colonization schemes (1970–1980), then more substantively through a vast expansion of logging concessions. Subsequently, much of the logged over forest has been converted to oil palm plantation (Curran et al. 2004). In Borneo and Sumatra, both countries (Malaysia, Indonesia) have now embarked on clearing the remaining logged over forest for *Acacia mangium* pulp plantations (1995–ongoing) (Curran et al. 2004). What forest now remains is restricted to the most unproductive soils and upland regions that are difficult to access. New Guinea (Papua and Iryan-Jaya, Indonesia) can be considered the last frontier in remaining large intact forest within the region. Much of it has now been gazetted for logging concession (1990–ongoing).

12.3 Physical Environment: Climate, Geology, and Soils

12.3.1 Africa

With a few exceptions, most of the Central and West Africa have rainfall that is generally below or just at 2,000 mm per year. Along the western coast, there are two dry seasons (Dec–Feb; July–Sept). The Harmattan, a dry desert wind, blows from the northeast from December to March, lowering the humidity. The greater expanse of forest in the Congo basin receives one dry season that lasts between three and four months; the forest core within central DRC spans the Equator and has from zero to two dry months during the year. The highest precipitation occurs along the Atlantic coast followed by the central basin and areas to the east (Walter and Lieth 1967; Richards 1996).

West and Central African soils are markedly different in terms of soil fertility. Whereas soils of West African wet forests are broadly mapped as soils within the Alfisoil order (Martin and Tsardakas 1991), those of Central Africa are predominantly Oxisols. In general, Central African upland soils are well weathered and highly infertile. These soils are underlain by a Pre-Cambrian continental shield that mostly comprises sedimentary shale and sandstones (Richards 1996; Juo and Franzluebbers 2003). However, the Cameroon fault line comprises a series of volcanoes and plateaus that make up the greater Cameroon Highlands separating Nigeria from Cameroon. Soils in this region are fertile and would be categorized as Andisols or Inceptisols that are much younger in development (Juo and Franzluebbers 2003). The highest mountain in this range is Mount Cameroon at about 4,000 m amsl. This is also the wettest region in the African tropics with rainfall exceeding 10,300 mm at Debundscha, a town at the base of Mt Cameroon (Walter and Lieth 1967; Whitmore 1998). The other source of fertility is the flood plains of the big rivers, where alluvium has been deposited. Such soils would be defined as Entisols.

12.3.2 Central and South America

Central and South America's climate is more strongly seasonal than the maritime influenced region of Southeast (SE) Asia. Rainfall exceeds more than 2,000 mm a year in five tropical wet areas: (1) coastal Guiana, Suriname, French Guiana; (2) the Amazon River Basin to the foothills of the Andes, and the upper basin of the Orinoco; (3) the Atlantic coastal region of Brazil; (4) the coasts of Colombia and northern Ecuador; and (5) the Atlantic coast of southern Central America (Costa Rica and Panama). The wettest and most aseasonal regions are the coast of Colombia, and the upper Amazon along the border between Brazil and Colombia and northern Peru (Walter and Lieth 1967; Richards 1996). These regions resemble the even rainfall distributions of maritime SE Asia, with rainfall exceeding 3,500 mm per year. Quibdo, Colombia, is the rainiest place in South America, and receives more than 8,900 mm of rain a year (Walter and Lieth 1967). However, the greatest part of all five regions of the continent generally has a dry season, with some, such as part of the lower Amazon Basin, having an extended dry period of more than 5 months with little rain (Whitmore 1998).

The soils of the Amazon and Orinoco basins as well as the Guiana's, being developed on the core of the continental shield, are highly weathered and nutrient poor – either tropical Oxisols or Spodosols (Jordan 1985). Those in Central America and along the coasts of Colombia are more fertile, largely because of volcanism of the central and coastal mountain ranges of these regions. But, also because of a much younger orogeny, soils would be categorized as Inceptisols or Ultisols (Juo and Franzluebbers 2003).

12.3.3 SE Asia

The SE Asian mixed dipterocarp rainforests receive rainfall between 2,500 and 6,000 mm per year with very little dry season (Walter and Lieth 1967). In some regions (SW Sri Lanka, NW Borneo, Malay Peninsula, and SW Sumatra), the forest type receives rain almost all the year from two seasons (Northeast monsoon, November–April; Southwest monsoons May–August) with rainfall from convection and orographic effects (from nearby coasts and mountains) between monsoons making even the dry season relatively wet compared to other tropical forest regions (Whitmore 1984). The region has a moderated maritime climate because the land largely consists of islands and peninsulas surrounded by the Indian and Pacific oceans. The more continental "Indochina" region of SE Asia to the north has greater and more distinctive seasonality – Burma, Thailand, Northeast India, Cambodia, Laos and Vietnam (Walter and Lieth 1967; Whitmore 1984).

The soils of the maritime influenced region of SE Asia consist of low fertility Ultisols or Andisols and Inceptisols. Ultisols are underlain by sediments of ancient continental origin – mostly shales and sandstones along the coastal plains, and granites on the backbone of the mountains (Borneo, Malay Peninsula). In contrast, Andisols and Inceptisols mostly of volcanic origin are found in Philippines, New Guinea, and Sumatra are considerably more fertile (Whitmore 1984).

12.4 Forest Types and Autecology of the Major Timber Trees

The forest types across the three tropical rain forest regions (West and Central Africa, Central and South America, SE Asia) are complex. They can be divided into wet and semievergreen forest types for several geographical areas within each region. For Africa they have been described by the geographical areas of: (1) West Africa and (2) Central Africa. For Central and South America, they have been broadly described as: (1) Atlantic side Central America and the northern Pacific region of South America; and (2) The Amazon, upper Orinoco basin, and the Guyana's. For SE Asia, they have been described as: (1) Indochina and (2) the maritime islands and peninsula. Regions that have been left out of this discussion, because of their relatively low importance in tropical timbers, include: (1) Atlantic coastal forest of SE Brazil; (2) NE Madagascar; (3) Papua New Guinea and wet island systems of Polynesia (Solomon Is, Fiji); and (4) the coast range of NE Australia.

12.4.1 Africa

12.4.1.1 West Africa

West Africa can be divided into forest types that are determined largely by rainfall, in which the wet evergreen type is restricted to the coast. Going toward the interior the forest type gets more strongly seasonal and semievergreen. The core rain forest region comprises an approximate expanse of 500 miles – West to East that includes the countries of Liberia, Côte d'Ivoire, and Ghana. The coastal high rainfall evergreen forests are dominated by the timber trees *Lophira alata*, *Cynometra anata*, and *Heriteria utilis* (Table 12.1a, Fig. 12.1) (Hall and Swaine 1983; Hawthorne and Jongkind 2006). Such forests are structurally complex, with tall emergents, and mixed composition. Further inland it changes to *Celtis-Triplochiton* forest that during the dry season comprises a semi-evergreen canopy with an evergreen understory. The Meliaceae genera *Entandrophragma* and *Khaya* have the highest stand densities in this zone. Much of the region, because of richer soils, has been cleared for small-holder farming – particularly cocoa; and plantation crops – usually oil palm (Martin and Tsardakas 1991). Farmers will often leave the faster growing shade intolerant timbers standing – (*Terminalia ivorensis*, *Triplochiton scleroxylon*). The driest most seasonal rain forest furthest from the coast is dominated by timber trees such as *Milicia excelsa*, *Nesogordonia papaverifera*, *Mansonia altissima*, and *Pterygota macrocarpa* (Hawthorne and Jongkind 2006). Within this wet-to-dry transition, lower lying swamps are dominated by more monodominant stands of *Alstonia boonei*, *Carapa procera*, *Mitragyna ciliata* and *Berlinea* spp. along with the climbing palms, *Calamus* and *Raphia* spp.

12.4.1.2 Central Africa

More than 70% of Africa's rainforests are found in Central Africa. (Richards 1996; Weber et al. 2001). The forest block can again be divided by wetness into a coastal belt that goes from SW Nigeria, across coastal Cameroon (protruding inland along the Cameroon Highlands separating Nigeria from Cameroon), across Equatorial Guinea and north and central coastal Gabon. A second wet zone forest type is found in the heart of the upper Congo River basin of the DRC, ending around Kisangani (Weber et al. 2001). An additional wet zone exists east of Kisangani centered on the Ituri Forest, DRC (Fig. 12.1). Around the Congo basin, more seasonal semievergreen forest dominates.

The coastal belt (Biafran Coast) between Nigeria and Cameroon and stretching over to Gabon comprises ever-wet mixed *Cynometra hakeri-Lophira alata*-dominated forest that is high statured. It is often replaced after swidden agriculture with *Pycnanthus angolensis* and *Coula edulis* (Hawthorne and Jongkind 2006). Many pure interspersed stands of Caesalpinoid legumes – *Brachystegia* spp., *Monopetalanthus* spp., *Cynometra* spp., and *Gilbertiodendron* spp. occupy particular edaphic (nutrient poor)/hydrological (moist seepage) sites across this type (Hart 2001).

Table 12.1a Major Timber Trees from West and Central African Rain Forests with common names, country, non-timber forest product and shade tolerance

Latin species and family	Common name and country	Other products	Autecology
Alstonia congolensis (Apocynaceae)	Alstonia (Nigeria, Congo)	Medicinal	Long-lived pioneer
Aningeria robusta (Sapotacaeae)	Asanfina/samfemamini (Ghana)	Fruit	Shade intolerant
Antiaris africana (Moraceae)	Kyenkyen (Ghana); ako (Togo)	Fruit	Shade intolerant
Aucoumea klaineana (Burseraceae)	Okoumé (Congo, Gabon)	Resin	Long-lived pioneer
Baillonella toxisperma	Moabi (Cameroon, Gabon)	Oil, food, medicinal	Shade intermediate
Borassus aethiopum (Palmae)	Sugar palm, ronier (Congo, Gabon)	Sugar	Shade tolerant
Butyrospermum parkii (Sapotaceae)	Sheabutter (Nigeria), karité (CAR)	Fruit/oil	Shade tolerant
Carapa procera (Meliaceae)	African crabwood (Côte d'Ivoire, Nigeria, Congo	Gum, food/oil, medicinal	Shade intermediate
Canarium schweinfurthii (Burseraceae)	Aiélé (Côte d'Ivoire, Gabon, Ghana)	Food, medicine, resin	Shade intolerant
Ceiba pentandra (Bombacaceae)	Fromager (Côte d'Ivoire, Ghana, Togo); ghe (Liberia)	Medicinal	Long-lived pioneer
Petersianthus africanum (Lecythidaceae)	Abale (Côte d'Ivoire, Gabon, Congo, Nigeria)		Long-lived pioneer
P. macrocarpum	Owewe (Cameroon, Gabon, Congo, Nigeria)	Medicinal	Pioneer
Cordia millenii (Boraginaceae)	Cordia (Cameroon, Côte d'Ivoire, Ghana, Nigeria,Togo)		Pioneer
Cylicodiscus gabunensi (Leguminosae)	Denyao, okan (Cameroon, Côte d'Ivoire, Ghana, Nigeria)	Medicinal	Shade tolerant
Daniellia ogea (Leguminosae)	Hyedua, daniellai (Cameroon, Côte d'Ivoire, Ghana, Nigeria)	Resin	Pioneer
Didelotia brevipaniculata (Leguminosae)	Sapo (Cameroon, Côte d'Ivoire, Ghana, Nigeria)		Shade tolerant
Diospyros spp. (Ebenaceae)	African ebony, Omenowa (Ghana, Nigeria)	Medicinal	Shade tolerant
Entandrophragma angolense (Meliaceae)	Tiama (DRC, Congo, Côte d'Ivoire)		Shade intermediate
E. cylindricum	Sapelli, liboyo (Cameroon, CAR, DRC, Congo)	Food (caterpillars), medic	Shade intermediate
E. candollei	Kosipo (DRC, Côte d'Ivoire, Ghana, Liberia)		Shade intermediate

(*continued*)

Table 12.1a (continued)

Latin species and family	Common name and country	Other products	Autecology
E. utile	Sipo (Cameroon, CAR, DRC, Congo)		Shade intermediate
Erythrophleum ivorense (Leguminosae)	Missandra (Cameroon, CAR, DRC, Congo)	Medicinal	Shade intermediate
Chrysophyllum africana (Sapotaceae)	Longhi (DRC, Congo, Gabon)	Fruit	Shade tolerant
Gossweilerodendron balsamiferum (Leguminosae)	Tola (DRC); agba (Nigeria)	Resin	Shade tolerant
Guarea cedrata (Meliaceae)	Bossé (DRC, Côte d'Ivoire, Liberia)		Shade tolerant
Guibourtia arnoldiana (Leguminosae)	Benge (Cameroon, Côte d'Ivoire, Ghana, Nigeria)		Shade tolerant
G. ehie	Amazakoue		Shade Intermediate
Hallea ciliate (Rubiaceae)	Bahia (Gabon); abura (Liberia)		Shade tolerant
Irvingia gabonensis (Irvingiaceae)	Oba (Cameroon, Côte d'Ivoire, Ghana, Nigeria)	Fruit, medicinal	Shade intermediate
Khaya anthotheca (Meliaceae)	Acajou (Cameroon, CAR, Congo, DRC, Ghana)		Shade intermediate
Khaya grandifoliola (Meliaceae)	Acajou (Togo)		Shade intermediate
K. ivorensis	Acajou (Côte d'Ivoire); mahogany (Ghana, Nigeria)	Medicinal	Shade intermediate
Lophira alata (Ochnaceae)	Azobé (Cameroon, Côte d'Ivoire, Gabon)	Medicinal	long-lived pioneer
Lovoa trichilioides (Meliaceae)	Dibetou (DRC, Côte d'Ivoire); cedar (Nigeria)	Medicinal	Shade intermediate
Mammea africana (Clusiaceae)	Oboto (Cameroon, Côte d'Ivoire, Ghana, Nigeria)	Fruit, medicinal, resin	Shade tolerant
Mansonia altissima (Sterculiaceae	Beté (Cameroon); mansonia (Ghana); ofun (Nigeria)		Shade intolerant
Microberlinia brazzavillensis (Leguminosae	Zebrano (Cameroon, Côte d'Ivoire, Gabon, Nigeria)		Shade tolerant
Milicia excelsa (Moraceae)	Iroko (Cameroon, CAR, DRC, Congo, Nigeria, Togo)	Medicinal	Long-lived pioneer
Millettia laurentii (Legumniosae)	Wengé (Congo, Gabon)		Shade intolerant
Mitragyna ciliata (Rubiaceae)	Bahia (Cameroon, CAR, DRC, Congo, Nigeria)	Medicinal	Shade intolerant
Monopetalanthus heitzii (Leguminosae)	Adoung (Gabon)	Medicinal	Shade tolerant
Musanga cecropioides (Moracaceae)	Umbrella tree (Cameroon, DRC, Congo, Ghana)	Medicinal, fodder	Short-lived pioneer

(*continued*)

Table 12.1a (continued)

Latin species and family	Common name and country	Other products	Autecology
Nauclea diderrichii (Rubiaceae)	Bilinga (DRC, Congo)		Medicinal, food
Nesogordonia papaverifera (Malvaceae)	Kotibé (Côte d'Ivoire); danta (Ghana, Liberia); otutu (Nigeria)xx		Shade tolerant
Parinari excelsa (Sapotaceae)	Sougue (Widely distributed)	Medicinal	Shade intermediate
Pericopsis elata (Leguminosae)	Afrormosia (Cameroon, DRC, Congo)		Shade intermediate
Piptadeniastrum africanum (Leguminosae)	Agboin (Cameroon, Côte d'Ivoire, Gabon, Nigeria)	Medicinal	Shade intermediate
Poga oleosa (Rhizophoraceae)	Ovoga (Nigeria, Cameroon, Gabon, Congo)		Shade intolerant
Prunus africana (Rosaceae)	Mueri (widley distributed)	Medicinal	Long-lived pioneer
Pterocarpus soyauxii (Leguminosae)	Padouk (Gabon, Cameroon, CAR, Congo)); red wood (Gabon)	Medicinal	Shade intermediate
Pterygota macrocarpa (Sterculiaceae)	Koto (Côte d'Ivoire, Ghana)	Medicinal	Shade tolerant
Pycnanthus angolensis (Myristicaceae)	Ilomba (Cameroon, Congo); otie (Ghana)	Medicinal	Shade intolerant
P. kombo	Ilomba (Côte d'Ivoire)		
Ricinodendron heudelotii (Euphorbiaceae)	Erimado (widely distributed)	Food, oil, medicinal	Short-lived pioneer
Scyphocephalium ochocoa (Myrsticaceae)	Sorro (Cameroon, Gabon)	Medicinal	Shade intermediate
Sterculia oblonga (Sterculiaceae)	Eyong (Ghana, Cameroon, Côte d'Ivoire, Nigeria)	Food, medicinal	Shade intermediate
Swartzia fistuloides (Leguminosae)	Dina (Ghana, Cameroon, Congo, Côte d'Ivoire, Nigeria)		Shade tolerant
Heritera utilis (Sterculiaceae)	Nyankom, niangon (Côte d'Ivoire, Ghana, Liberia)		Shade intolerant
Terminalia ivorensis (Combretaceae)	Fraké (CAR); framiré (Côte d'Ivoire); edo (Nigeria)		Long-lived pioneer
T. superba	Fraké (Cameroon); limba (CAR, DRC, Congo)	Medicinal	Long-lived pioneer
Tieghemella heckelii (Sapotaceae)	Makoré (Côte d'Ivoire); makore (Ghana)	Medicinal	Shade intolerant
Triplochiton scleroxylon (Sterculiaceae)	Ayous (Cameroon, CAR, Congo); samba (Côte d'Ivoire); obeche (Nigeria)		Long-lived pioneer

Information based on Hall and Swaine (1983), Hawthorne (1995), White and Abernethy (1997), Van Dijk (1999), Hawthorne and Jongkind (2006)

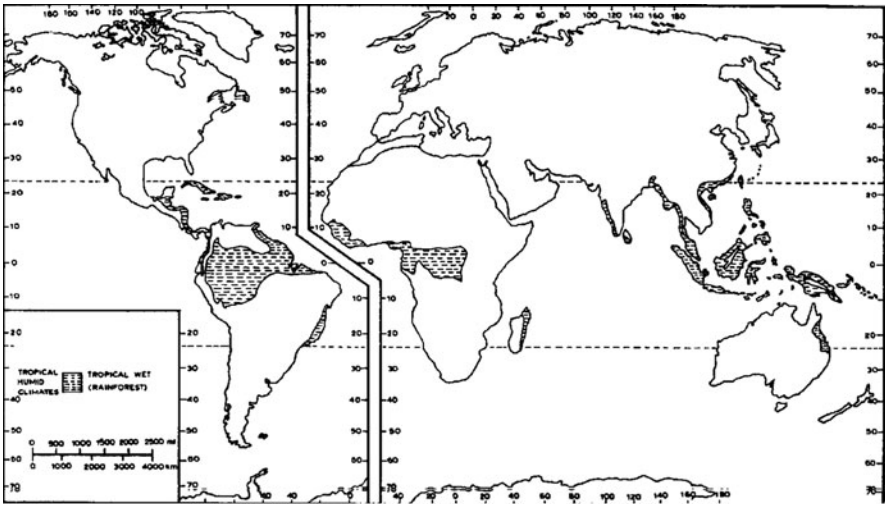

Fig. 12.1 A Map depicting tropical wet forest defined by where rainfall is greater than 2,000 mm per year

The core wet evergreen region of the upper Congo basin comprises many of the same species without the diversity of legumes (Hart 2001). Legumes do, however, represent a significant component of these forests. *Gilbertiodendron deweveri* forms monodominant stands throughout the region, where an individual stand can cover more than 1,000 ha (Hart 2001). Further, in northeastern DRC, although not dominant, both *Julbernardia seretii* and *Cynometra alexandri* can comprise up to 40% of stem density and/or basal area. The mixed forests have noticeable timber additions in the Irvingaceae (*Irvingia* spp.), Ebenaceae (*Diospyros* spp.), and Leguminosae (e.g., *Pericopsis elata, Millettia laurentii, Pterocarpus soyauxii*). Extensive flood plain and swamp forests exist along the major river systems draining into the Congo basin itself. Such forests have been poorly described because generally they are not timber rich. Monodominant swamps of *Guibourtia arnoldiana, G. Demeusei*, and *G.ehie* are the exception (Hart 2001; Weber et al. 2001).

The semievergreen forests ring the ever-wet region with extensive areas on the northern bounds of the Congo basin, including the southern third of Cameroon, southern and northern DRC, and the northern half of the Republic of Congo. While exhibiting different species-specific distributions, many of the same species as in West African semievergreen forest are found in this type (e.g., *Mansonia altisima, Terminalia superba, Triplochiton scleroxylon, Nesogordonia papaverifera, Pterygota macrocarpa, Pericopsis elata*), with the addition of *Gossweilerodendron balsamiferum*, and an abundance of scattered Meliaceae that include, *Entantrophragma* spp., *Guarea cedrata, Khaya anthotheca*, and *Lovoa trichilioides* (Weber et al. 2001).

The forests of Gabon include both wet Atlantic coastal forest and somewhat drier semievergreen forest of its neighbors. The timber species composition is similar to the forests of neighboring countries with two notable exceptions. The exploitation of one timber species – *Aucomea klaineana* – has sustained the timber industry for several decades (White and Abernethy 1997; Reitsma 1988). Further, the forests are generally poor in Meliaceae.

12.4.2 Latin America

12.4.2.1 Central America and the North Pacific Region of South America

The ever-wet forest of this region is restricted to Atlantic/Caribbean side of Costa Rica, and Panama; broadening in the Darien, stretching east to the interior foothills of the Sierra Nevada range (an extension of the Andes) on the Caribbean side of Colombia and into Venezuela; and west down the Pacific coast range of Colombia to northern Ecuador (Fig. 12.1) (Foster and Hubbell 1990; Gentry 1990). The mixed forests on the well-drained soils are the tallest and most stratified in structure. The emergent and canopy timber species are varied in genera with shade intolerant canopy and emergent timber trees such as *Cordia alliodora*, *Brosimum* spp., *Anacardium excelsum*, *Terminalia amazonia* throughout the range; *Virola* spp., and *Pentaclethra macroloba* toward the northern parts of the region (Foster and Hubbell 1990); *Dipteryx panamensis*, *Carapa guianensis*, and *Cavanillesia platanifolia* in the lower lying and swampy region of the Darien, and species in the Burseraceae (*Dacryodes* spp.), and *Ceiba pentandra* toward the southern range of the forest type (Table 12.1b). More shade tolerant and slower growing species such as *Calophyllum brasiliense*, *Dalbergia retusa*, and *Hymenaea courbaril* comprise the subcanopy and canopy (Foster and Hubbell 1990). Genera such as *Cecropia* and *Didymopanax* represent early-to-mid fallow stages of swiddens or after agricultural abandonment (Jordan 1985).

At higher and wetter elevations, Lauraceae (*Ocotea* spp., *Nectandra* spp.,) dominates the canopy particularly in the Pacific coast range of northern Ecuador and the central Cordierra of Costa Rica and western Panama. At such elevations, *Alnus acuminata* is found within riparian and disturbed areas. Infertile lowland soils within this forest matrix can be dominated by pure stands of *Campnosperma panamensis*; riparian flooded forests can comprise *Pterocarpus* spp.; and backwater swamps often comprise pure-to-mixed stands of *Carapa guianensis* and *Prioria copaifera* (Gentry 1990)

Semievergreen rain forest can be considered a transition zone to dry forest on the Pacific side of Central America or toward the interior and lowlands of Colombia Venezuela, and Ecuador, and as an extension down to northern Peru on the Pacific side foothills of the Andes. Such forests have largely shade intolerant species *Andira inermis*, *Pachira quinata*, *Spondias mombin*, *Ceiba pentandra*, and scattered emergent trees in the Meliaceae (*Cedrela odorata, Swietenia* spp.) (Gentry 1990). At either end of the semievergreen forest type in Costa Rica (northern) or Venezuela (eastern)/Ecuador (southern), *Enterolobium cyclocarpum*

Table 12.1b Major timber trees from Central and Latin American rain forests with common names, country, nontimber forest product, and shade tolerance

Latin species and family	Common name and country	Other products	Autecology
Alnus acuminata (Betulaceae)	Aliso (Bolivia, Colombia, Ecuador)	Stabilization	Short-lived pioneer
Anacardium excelsum (Anacardiaceae)	Marañon (Ecuador); espavé (Panama)	Medicinal	Shade intolerant
Anadenanthera colubrina (Leguminosae)	Curupay (Brazil)	Medicinal, beverage, tanin	Long-lived pioneer
Andira inermis (Leguminosae)	Angelin (Costa Rica, Panama, Venezuela)	Medicinal, bees	Shade intolerant
Astronium graveolens (Anacardiaceae)	Goncalo alves (widely distributed)		Shade intolerant
Bagassa guianensis (Moraceae)	Bagasse (Guyana, Brazil)		Shade intolerant
Bertholletia excelsa (Lecythidaceae)	Brazilnut, castanheiro (Brazil)	Fruit	Shade intermediate
Bombacopsis quinata (Verbenaceae)	Cedro espino (Panama); saqui saqui (Venezuela)		Long-lived pioneer
Brosimum alicastrum (Moraceae	Charo (Venezuela); ramon (Mexico)	Fruit, medicinal	Shade intermediate
Brosimum utile	Amapa (Brazil); sande (Colombia, Ecuador)	Fuit, medicinal	Shade intermediate
Calophyllum brasiliense (Clusiaceae)	Santa maría (Guatemala, Honduras); jacareuba (Brazil); maria (Panama)		Shade tolerant
Campnosperma Panamensis (Anacardiaceae)	Orey (Panama)		Shade intolerant
Carapa guianensis (Meliaceae)	Crabwood (Guyana, Suriname); carapa (Venezuela)	Fruit, medicinal	Shade intermediate
Catostemma commune (Bombacaeae)	Baromalli (Guyana); baramán (Venezuela)		Long-lived pioneer
Cecropia peltata (Cecropiaceae)	Trumpet wood		Short-lived pioneer
Cedrela odorata (Meliaceae)	Cedro (Bolivia, Colombia, Ecuador, Guatemala, Peru, Venezuela)		Shade intolerant
Cedrelinga catenaeformis (Leguminosae)	Chuncho (Ecuador); tornillo (Peru)	Medicinal	Shade intolerant
Ceiba pentandra (Bombacaceae)	Kapok (Ecuador); ceiba (Venezuela, Honduras)	Medicinal, other	Long-lived pioneer
Chlorophora tinctoria (Moraceae)	Fustic (widely distributed)	Medicinal, dye	Long-lived pioneer
Cordia alliodora (Boraginaceae)	Laurel (Ecuador, Honduras, Panama)	Medicinal	Shade intolerant
Chrysophyllum spp., (Sapotatceae)	Caimito (widely distributed)	Fruit, medicinal	Shade tolerant
Dacryodes excelsa (Burseraceae)	Candle tree (Costa Rica, Venezuela)	Resin	Shade intermediate

(*continued*)

Table 12.1b (continued)

Latin species and family	Common name and country	Other products	Autecology
Dalbergia retusa (Leguminosae)	Cocobolo (Panama)	Medicinal	Shade tolerant
D. nigra	Jacaranda, rosewood (Brazil)		Shade tolerant
Dialium guianense (Leguminosae)	Jutahy (widely distributed)		Shade tolerant
Didymopanax morotoni (Araliaceae)	Morototo (widely distributed)		Short-lived pioneer
Dipteryx panamensis (Leguminosae)	Tonka, dipteryx (costa Rica, Panama, Colombia)		Shade intolerant
Erisma uncinatum (Vochysiaceae)	Cedrinho (Brazil); moreillo (Venezuela, Peru)		Shade intolerant
Enterolobium cyclocarpum (Leguminosae)	Guanacaste (widely distributed)	Fruit, stabilization	Shade intolerant
Eschweilera spp., (Lecythidaceae)	Manbarklak (widely distributed)		Shade intermediate
Genipa americana (Rubiaceae)	Genipa (Costa Rica, Colombia)	Fruit, medicinal	Shade intolerant
Goupia glabra (Goupiaceae)	Kabukalli (Guyana); kopie (Suriname)	Fruit	Long-lived pioneer
Guarea spp.(Meliaceae)	American muskwood (widely distributed)		Shade intermediate
Hevea brasiliensis (Euphorbiaceae)	Para, rubber (Brazil, Peru)	Latex	Shade intolerant
Hyeronima alchorneoides (Euphorbiaceae)	Pilon (Coast Rica, Panama, Colombia)	Medicinal	Shade intolerant
Hymenaea courbaril (Leguminosae)	Jatobá (Brazil) (widely distributed)	Fruit, resin,	Shade tolerant
Hura crepitans (Euphorbianceae)	Ochoó, catahua (widely distributed)	Medicinal	Long-lived pioneer
Lonchocarpus castilloi (Leguminosae)	Manchiche (Guatemala, Costa Rica)	Medicinal	Shade intolerant
Manilkara bidentata (Sapotaceae)	Balata (Guyana); purguo (Venezuela)	Latex	Shade tolerant
Mora excelsa (Moraceae)	Mora (Trinidad, Guyana, Venezuela)		Shade intermediate
Ochroma pyramidale (Bombacaeae)	Balsa (widely distributed)	Other	Long-lived pioneer
Ocotea rodiaei (Lauraceae)	Demerara greenheart (Guyana, Venezuela)		Shade tolerant
O. rubra	Determa (Guyana, Brazil)		Shade tolerant
Ormosia spp. (Leguminosae)	Baracara (widely distributed)	Medicinal	Shade intolerant
Peltogyne venosa (Leguminosae)	Purpleheart (Guyana); purperhart (Suriname)		Shade tolerant
Pterocarpus spp., (Leguminosae)	Sangre (widely distributed)	Medicinal	Shade intermediate
Prioria copaifera (Leguminosae)	Cativo (Panama, Colombia, Venezuela)	Medicine, cordage	Shade intolerant
Samanea saman (Leguminosae)	Raintree (Costa Rica, Panama, Colombia)	Stabilization	Long-lived pioneer

(*continued*)

Table 12.1b (continued)

Latin species and family	Common name and country	Other products	Autecology
Spondias mombin (Anacardiaceae)	Jobo (widely distributed)	Fruit, medicinal	Shade intolerant
Simarouba amara (Simaroubiaceae)	Marupá (Guyana, Peru); cedro blanco (Venezuela)	Medicinal	Shade intolerant
Sterculia apetala (Sterculiaceae)	Sujo (Bolivia); camaruco (Venezuela)		Shade intermediate
Swietenia macrophylla (Meliaceae)	Mara (Bolivia, Ecuador); mogno (Brazil); caoba (Guatemala, Panama, Peru)		Shade intermediate
Tabebuia rosea (Bignoniaceae)	Apamate (Trinidad and Tobago, Venezuela); cedro rosado (Colombia)		Shade intolerant
Tabebuia spp.	White cedar, ipe (widley disributed)	Medicinal, bees	Shade intolerant
Terminalia amazonia (Combreatceae)	Amarillo (Cosat Rica, Panama); cumbillo (Honduras, Venezuela)		Shade intolerant
Tetragastris spp., (Burseraceae)	Masa (widely distributed)	Resin	Shade intolerant
Virola koschnyi (Myristicaceae)	Palo de sangre (Honduras, Costa Rica, Panama, Colombia)		Shade intermediate
V. surinamensis	Virola (Brazil, Guyana)		Shade intermediate
Vochysia spp., (Vochysiaceae)	Kwari (widely distributed)		Shade intermediate

Information based on Croat (1978), Gentry (1990), Francis and Lowe (2000)

is a characteristically dominant tree. *Tabebuia* as a genus illustrates clinal differences in species from north to south with *T. chrysophylla, T. rosea,* and *T. pentaphylla* (Gentry 1990) Early seral habitat or fallow is often dominated by *Cecropia spp*., and *Ochroma pyramidale* (Jordan 1985; Gentry 1990).

12.4.2.2 The Greater Amazon Basin and the Guyanas

Much of the basin associated with the Amazon and its larger tributaries can be considered periodically flooded (varzea). This forest type is very varied and dominated by palm species (e.g., *Mauritia* spp.) but not usually rich in timber. The upland forest (terra firme) comprises many important timber trees that are scattered or in local populations. A regional arc around the outer basin of the Amazon to the west and south comprise many timber trees associated with greater dryness and seasonality – particularly trees in the Meliaceae (*Cedrela* spp., *Swietenia macrophylla*), Leguminosae such as *Cedrelinga catenaeformis*, as well as *Bertholetia excelsa*, *Anacardium excelsum*, and *Hevea brasiliensis* (Prance 1990).

The other important region for timber is the coastal plain and interior uplands of the Guyana's (Guiana, Suriname, French Guiana) and eastern Venezuela (Richards 1996). The main wet evergreen forest type, primarily in the northwestern region of Guiana, from the foothills of Merume Mountains and Guiana highlands to the coast, can be characterized by the presence of the trees *Eschweilera sagotine – Licania heteromorpha*. Species such as *Ocotea rodei*, *Catostemma commune*, *Hymenea davisii*, and *Peltogyne venosa*, occur in scattered numbers, often with canopy and subcanopy palms in the genera Bacteris and Geonoma (Prance 1990) This type, due to edaphic and rain fall changes, becomes more semievergreen further southeast into French Guiana and Suriname, dominated by *Swartzia leiocalycina*, *Terminalia amazonia*, *Piptadenia suaveolens*, *Tetragastris* spp., *Vochysia* spp., *Manilkara bidentata*, and *Simarouba amara* (Prance 1990) Early seral species such as *Cecropia* spp., and *Goupia glabra* signify disturbance in this forest type. In both types, the backwaters (*Mora excelsa*, *Carapa guianensis*, *Hura crepitans*) and riparian zones (*Hymenaea courbaril*) of river systems draining into the Atlantic play a big role in determining more monodominant timber types (Richards 1996).

12.4.3 Southeast Asia

12.4.3.1 Maritime SE Asia

The ever-wet rain forests that predominate Borneo, Peninsula Malaysia and southern Thailand, and Sumatra can be divided broadly into lowland mixed dipterocarp and hill mixed dipterocarp forests (>300 m amsl) (Fig. 12.1) (Ashton 1980). The lowland type is dominated by the timber tree family Dipterocarpaceae (red meranti group – *Shorea leprosula* and *S. parvifolia*), Dipterocarpus spp. and Hopea spp. (Ashton 1980; Whitmore 1984). Non-dipterocarps comprise Dryera costulata, *Gluta* spp., *Intsia palembanica, Koompasia malaccensis, Sindora* spp., *Tarrietia* spp., and *Palaquium* spp. (Whitmore 1984). Certain species such as *Dryobalanops lanceolata* have an affinity for shale-derived soils; *Shorea* spp., and *Dipterocarpus* spp., is often almost genera monodominant in mixtures on poorer sandstone-derived soils (Ashton 1964; Ashton 1980; Whitmore 1984).

On the coasts peat swamp forests can be dominated by single species such as *Shorea albida* (in Sarawak) or as mixtures comprising *Gonostylus bananas*, and *Shorea* spp.; backwater swamps by *Dipterocarpus costulatus*, *Intsia* spp., and *Koompasia* spp.; and riparian and alluvial forests can be dominated by *Dryobalanops beccari* and *S. parvifolia* (Whitmore 1984).

Hill dipterocarp forests usually comprise shade tolerant *Shorea curtisii* and *S. laevis* as dominants with other non-dipterocarp species such as *Swintonia specifera* (Whitmore 1984). An extension of this type in the Western Ghats and SW Sri Lanka is dominated by *Mesua* spp., *Dipterocarpus* spp., and *Shorea* section Doona (Ashton and Gunatilleke 1987).

Table 12.1c Major timber trees from South and Southeast Asian rain forests with common names, country, non-timber forest product and shade tolerance

Latin species and family	Common name and country	Other products	Autecology
Adina cordifolia (Rubiaceae)	Haldu (Sri Lanka, India, Thailand, Burma)	Medicinal	Shade intolerant
Alstonia macrophylla (Apocynaceae)	Pulai (Sri Lanka, Malaysia, Indonesia, Thailand)		Long-lived pioneer
Anthocephalus chinensis (Rubiaceae)	Kadam (Malaysia, Thailand, Burma)	Medicinal	Long-lived pioneer
Anisoptera glabra (Dipterocarpaceae)	Mersawa (Cambodia, Indonesia); phdiek (Cambodia)		Shade intermediate
Artocarpus heterophyla (Moraceae)	Jak (Sri Lanka, India)	Fruit	Shade intermediate
Balanocarpus spp. (Diptercarpaceae)	Chengal, pinak (Malaysia, Indonesia)		Shade tolerant
Calophyllum spp. (Clusiaceae)	Bitanghor (Philippines)	Medicinal, oil	Shade tolerant
Canarium spp.(Burseraceae)	Kedongong (India, Burrna, Thailand, Malaysia)	Medicinal, resin	Shade intolerant
Chloroxylon swietenia (Rutaceae)	Satinwood (India, Sri Lanka)		Shade intolerant
Chukrasia tabularis (Meliaceae)	Chickrassy (India, Sri Lanka, Malaysia)	Gum, dye, medicinal	Shade intolerant
Dialium spp. (Leguminosae)	Keranji (Malaysia, Indonesia)		Shade tolerant
Diospyros spp. (Ebenaceae)	Ebony, kayu malam (Malaysia, India, Sri Lanka)		Shade tolerant
Dipterocarpus spp (Dipterocarpaceae)	Keruing (throughout SE Asia)	Resin	Shade intermediate
Dryobalanops spp. (Dipterocarpaceae)	Kapur (Malaysia, Indonesia)	Resin	Shade intermediate
Dyera costulata (Apocynaceae)	Jeluntong (Malaysia, Borneo, Sumarar)	latex	Long-lived pioneer
Gluta spp. (Anacardiaceae)	Rengas (Malaysia, Indonesia)	Medicinal	Shade tolerant
Eusideroxylon swageri (Lauraceae)	Belian (Malaysia, Borneo)	Medicinal	Shade tolerant
Gonystylus bancanus (Thymeliaceae)	Ramin (Indonesia, Malaysia)		Shade intermediate
Intsia bijuga (Leguminosae)	Kwila (Papua New Guinea); merbau (Malaysia, Indonesia)		Shade tolerant
I. palembanica	Ipil (Malaysia, Indonesia)		Shade tolerant
Koompasia malaccensis (Leguminosae)	Kempas (Malaysia, Indonesia)	Honey	Shade intermediate
Mangifera spp. (Anacardiaceae)	Mango,machang (India, Sri Lanka, Philippines, Malaysia, Indonesia)	Fruit	Shade intermediate
Melia azederach (Meliaceae)	China berry (India, Thailand, Cambodia)		Long-lived pioneer
Mesua ferrea (Clusiaceae)	Na (Sri Lanka, India), gangaw (Malaysia)	Resin	Shade tolerant

(*continued*)

Table 12.1c (continued)

Latin species and family	Common name and country	Other products	Autecology
Michelia spp. (Magnoliaceae)	Champaca (India, Sri Lanka, Burma, Thailand)		Shade tolerant
Myristica spp. (Myrsticaeae)	Nutmeg, darah darah (Sri Lanka, Indochina)	Food	Shade intermediate
Palaquium spp. (Sapotaceae)	Gutta percha (throughout SE Asia)	Latex	Shade tolerant
Pterocarpus indicus (Leguminosae)	Rosewood (Papua New Guinea); narra (Philippines)		Shade tolerant
Shorea spp. (Dipterocarpaceae)	Dark red meranti/red lauan (throughout SE Asia)	Resin	Shade tolerant
Shorea spp.	Balau (throughout SE Asia)	Resin, food	Shade tolerant
Shorea spp.	Light red meranti/seraya (throughout SE Asia)	Resin	Shade intermediate
Shorea spp.	White meranti/white lauan (throughout SE Asia)	Resin	Shade intermediate
Shorea spp.	Yellow meranti (throughout SE Asia)		Shade tolerant
Sindora spp. (Leguminosae)	Sepetir (throughout Asia)		Shade tolerant
Tectona grandis (Verbenaceae)	Teak (Indochina)		Shade intolerant
Terminalia spp. (Combretaceae)	White chuglam, Indian almond, Indian laurel (India, Thailand, Burma)		Shade intermediate
Tetrameles nudiflora (Datisaceae)	Thipok (Indochina, India, Sri Lanka)	Medicinal	Shade intolerant
Vatica spp. (Dipterocarpaceae)	Resak (Sri Lanka, Malaysia, Indonesia)	Food	Shade intermediate
Vitex spp. (Verbenaceae)	Vitex (Sri Lanka, India, Burma, Thailand)		Shade intolerant

Information based on Troup (1921), Burkill (1966), Joshi (1980), Wyatt-Smith (1963), De Beer and McDermott (1989), Ashton et al. (1997)

Early successional forest, following land clearance or disturbance, is characterized by fast-growing pioneers *Macaranga* spp., *Alstonia* spp., and *Dillenia* spp.

12.4.3.2 Indochina

The semievergreen rain forests of India, Sri Lanka, Burma, Thailand, Cambodia, and Vietnam are both more fragmented and isolated from each other because of landform and climate compared to similar forest in the Neotropics and Africa. All are driven by relatively short dry seasons and lengthy and high rainfall monsoonal periods. The forest type ranges across South Asia and Indochina from the NE coastal plains of Sri Lanka, the western foothills of the Ghats, NE Indian hinterlands (Assam), the coastal mountain ranges adjacent to the Bay of Bengal in Burma, the northern part of the Malay peninsula (below 11 N latitude) in Thailand, the southern side (facing the Gulf of Thailand) of the Elephant Mountains in Cambodia, and the coastal hills of southern Vietnam (Whitmore 1984, 1998). *Shorea guiso*, *S. hypochra*, *S. assamica*, *Dipterocarpus tubinatus*, *D. alatus*, *D. griffithii* and *Vatica* spp., are all dipterocarps well represented within this type. Noticeably absent is the red meranti group of *Shorea* spp. (Ashton 1980). Other species comprise *Adina cordifolia*, *Artocarpus* spp., *Chloroxylon swietenia*, *Chukrasia tabularis*, *Diopyros* spp., *Michelia champaca, Swintonia floribunda*, and *Terminalia* spp. (Whitmore 1984)

The canopy tree species are all shade intolerant. The dipterocarps often predominate on the more infertile soils (sandstone derived) as compared to the non-dipterocarps. Early successional forests arising after swidden agriculture or more permanent land abandonment often comprise *Vitex spp*.

12.4.4 Regional Comparisons

In comparison with the other tropical wet forest regions, the maritime influenced region of equatorial SE Asia has forests, which are higher in standing timber volumes, higher densities of NTFPs compared with other tropical forest regions (Richards 1996; Whitmore 1998). In addition, compared with Africa and the Neotropics, tree diversity in Asia is largely nested within species-rich genera of a variety of families (*Dialium, Diospyros, Dipterocarpus, Gluta, Mangifera, Myrystica, Palaquium, Shorea, Sindora, Terminalia*). The approximate total number of Asian timber tree genera is lower than either numbers in Africa or the Neotropics. Tree families in the Neotropics that comprise high numbers of timber trees are the Anacardiaceae, the Meliaceae, and the Moraceae. Given a related biogeographic history to that of South America (Prance 1990), the African tropics has similar families of timber trees – Meliaceae, Moraceae, and the Sterculiaceae (Gentry 1990; Hawthorne and Jongkind 2006). Across all tropical wet forest regions, the timber trees in the Leguminosae are important – many of them extremely slow growing. In addition, Central Africa has a group of Caesalpinioideae legumes that can form extensive monodominant stands in uplands. All three regions can form monodominant stands of trees along water courses and swamps, usually containing species of the Leguminosae and Meliaceae (Richards 1996; Whitmore 1998). Compared to Asia, no one timber-rich genus stands out in the Neotropics, while in Africa the genera *Diospyros, Entandrophragma*, and *Khaya* are important (Hawthorne and Jongkind 2006).

Across the whole Asian region, and using a quick, but superficial assessment of the shade tolerance of timber trees, the Neotropics comprise 34 (56%) shade intolerant trees versus 11 that are considered shade tolerant (Table 12.1b). For Africa, 45 tree species (74%) are listed on the shade intolerant side compared with 16 (26%) considered shade tolerant (Table 12.1a). Within continental regions of both Central America and Indo-China (India, Burma, Thailand, Cambodia, Vietnam, Laos), timber trees are proportionately more shade intolerant than those of South America and maritime influenced Asia. This is perhaps because both maritime Asia and South America have proportionately had larger areas of relatively longer-term stable continental scale environments that have promoted longer-lived, slower growing trees (Whitmore 1998). The family that dominates the forests in maritime SE Asia is the relatively slow growing, shade tolerant, and shade intermediate Dipterocarpacaeae (Ashton 1980).

12.5 Regional Silvicultural History and Management Knowledge

Timber-rich forest types were most extensive in Asia, and are represented by mixed dipterocarp forests – a type dominated by one timber-rich tree family (Dipterocarpaceae – *Shorea, Dipterocarpus, Dryobalanops*). Standing commercial timber volumes are far higher per hectare (60–200 m^3 ha^{-1}) than any other tropical forest type, largely because of the monodominance of the timbers in the forest canopy. The African and Neotropical wet tropical forests have timber forest types that are largely confined to particular sites or that are sparsely represented in the canopy such that merchantable volumes are much lower (5–50 m^3 ha^{-1}). Usually, no timber family dominates the canopy. Silvicultural research on timber-rich forests of the wet tropics is vastly unequal across regions and countries within regions.

12.5.1 Africa

Africa is in the worst shape regarding silvicultural knowledge and research infrastructure. Where extensive work has been done (Ghana, Nigeria, Uganda), little forest remains (Mergen and Vincent 1987; Hawthorne and Jongkind 2006). Where the forest remains (Central Africa and Congo River basin), only a handful of published papers can be found with some amount of gray literature. Fortunately, many of the most highly valued timber species and forest types in West Africa and Uganda also exist and are exploited in Central Africa. Thus, relatively recent work by Swaine, Hawthorn, and colleagues that builds on the classic work of Taylor (1960) and others has some application, particularly with reference to the semi-evergreen forests bounding the northern border of the Central African forest block. Important historical work can be found in the Oxford Forest Memoirs, Commonwealth Forest Review, and the Nigerian Forestry Information Bulletin.

During the colonial period, French foresters worked in West Africa, but more on plantation forestry rather than natural forest management. Most studies were either published in the gray or difficult to access literature; however, summaries from relevant silvicultural information that can be gleaned have been published in Bois ets Forets des Tropiques. Belgian foresters working in the former Belgian Congo (now DRC) published works that help lay the ground for silvicultural research in a series produced by the Institut National pour l'Etude Agronomique du Congo Belge.

In spite of some classic early works, forestry research on natural forest management in Central Africa came virtually to a halt at independence. Unfortunately, the studies that have been undertaken in postcolonial Central Africa have rarely made it into the peer-reviewed literature.

12.5.2 Central and South America

Silviculture research has a long history in the Caribbean islands, in particular, in Puerto Rico through the International Institute of Tropical Forestry (IITF), US Forest Service and through the early work by Beard and colleagues, (e.g. Beard 1946) in Trinidad and Tobago. This experience, implemented in recent years through numerous partnerships, has been ongoing for more than 60 years. Much of this history is documented in the journal Caribbean Forester by F.H Wadsworth in the 1950s.

Latin America has a strong but relatively recent literature base in Costa Rica (1960–ongoing) through the Tropical Agricultural Research and Higher Education Center (CATIE) and in Belize (Lamb 1966). Budowski's (1961) work on succession was the historical trigger toward documenting and testing experimental silviculture treatments in Costa Rica (Finegan and Camacho 1999; Finegan et al. 1999). Some work has been done in Colombia, Venezuela, and Bolivia (Peña-Claros et al. 2008a, b), and Surinam (De Graaf 1986). Little silvicultural research has been undertaken in Panama, Peru, and Ecuador. Brazil has done a substantial amount of work, again in the recent years (1980s onward) (Verissimo et al. 1992, 1995) but much of this is in gray literature.

12.5.3 Southeast Asia

Voluminous work has been done in India and Peninsula Malaysia starting in 1850. Sir Dietrich Brandis founded the Indian Forest Service in 1850 in response to forest exploitation of what was then British India's teak (*Tectona grandis*) and sal (*Shorea robusta*) forests (Brandis 1897). Later, research officers were appointed to work on the seasonally wet forests of Burma, ca. 1910 (Troup 1921). The British also established a forest research administration in Malaya in 1883 (Hill 1900), which later developed into the Forest Research Institute at Kepong. Research officers were also appointed within the forestry department in Colombo, Sri Lanka, ca. 1890 (Holmes 1957). Such a research infrastructure in Malaysia and Sri Lanka was set up in-part to develop sustainable techniques for managing mixed dipterocarp forest; a type that transcended lowland and hill regions across the aseasonal wet tropics of SE Asia. A landmark summarization of this work was done by Wyatt-Smith (1963). In addition, a wealth of gray literature from this research has since accumulated in the archives of these various agencies, national libraries, and universities (Ashton and Peters 1999). Much of this literature is not regularly accessible to today's researchers, and this is corroborated by an absence of citations of past work. It is also, by modern standards, generally statistically flawed, both in sampling design and in analysis, but the interpretations by those early researchers are largely valid.

The remaining parts of Asia have seen relatively little research – with some work done during the 1930–1960s in the Philippines, and little to no work that has been done in Indonesia until recently (1990–ongoing) (e.g., Sist et al. 1998). New Guinea has almost no work published in the peer-reviewed literature, but there is some gray literature mostly based at the Forest Research Institute, Lai.

12.6 Uses of Important Tropical Timber and NTFPs

Compatible and sustainable technologies for management of nontimber and timber forest products are essential if tropical wet forests are to be conserved outside of preserves and park. Studies have shown that in combination net present values can double or even triple with NTFPs often providing the majority of income (Peters et al. 1989a, b; Godoy et al. 1993; Boot and Gullison 1995; Ashton et al. 2001b). Regional examples are provided in the following sections.

12.6.1 Africa

Although known since the early 1900s, African tropical timbers became important commerce on international markets after World War II. The most accessible forests at that time were those of the former colonies of French and British West Africa (Ghana, Nigeria, Togo, Cote d'Ivoire, and Cameroon), where high quality timber supplied European luxury wood markets. By 1985, 72% of West Africa's rain forests had been logged over and transformed into fallow (FAO 2001, 2006). In West Africa, of the approximately 180 large tree species, only about 25% can be used for timber, and of these only 15–20 are on international markets (Table 12.1a). This is similar to the forests of the America's and Central Africa. However, in Asia, just because of the dominance of the Dipterocarpaceae, and the diversity of commercial species within this family, many more trees are considered commercial timbers but under more general grouped common names (Table 12.1c).

In recent years, two trends have marked timber exploitation in Africa. First, there has been an ever-increasing trend from supplying European markets to supplying those of SE Asia, particularly China. Second, a significant portion of exploited timber is consumed locally and never makes it to international markets (Barbier et al. 1994).

For Central Africa, the Meliaceae and the luxury export timbers of *Khaya* spp. and *Entandrophragma* spp. are a large factor in forest exploitation. However, often West and Central African timber trees also have important medicinal, food, latex, and beverage uses by the regional and local people (Table 12.1a). Exploitation for timber can be problematic for NTFPs (especially – medicinal, food, latex) when tree populations are depleted from logging. Examples are numerous of timber trees in which logging has depleted their ability to sustain viable production of NTFPs. *Entandrophragma cylindricum* (sapelli) provides sustenance to a caterpillar that is a seasonally abundant source of protein for local people. *Pterocarpus soyauxii* (padouk), is a high value wood for furniture and flooring, is a preferred local wood for building canoes, musical and agricultural implements; and when powdered is also has important cultural and medicinal uses (Laird 1995). *Milicia excelsa* (iroko) is a tree species considered sacred and used medicinally that is now heavily depleted due to logging pressures (Laird 1995). Finally, *Aucoumea klaineana* (Okoume), a timber species used for plywood, produces a flammable resin that is used within the populace to start fires.

12.6.2 Central and South America

The period following World War II also saw an increase in international trade in timber from coastal South and Central America. Areas first exploited included Belize, and the Atlantic forest of Brazil – for both European and American luxury imports (Lamb 1966). By the mid-1980s, the Atlantic coastal forest in Brazil was markedly deforested.

A high proportion of South and Central America timber have other medicinal and food uses (Mendelsohn and Balick 1995), but as compared to other regions a smaller proportion play significant roles as nontimber products. NTFPs of nontimber trees therefore comprise a much higher proportional value of the forest than timber – particularly in the Amazon varzea and terra firme forests (Peters et al. 1989a, b; Godoy et al. 1993). Obvious examples of important NTFPs from timber trees in Amazonia are: latex trees – *Hevea brasiliensis* (rubber) and *Manilkara spp*.; fruit – *Brosimum alicastrum*, *Genipa americana*, *Spondias mombin*; nut – *Bertholletia excelsa* (Brazilnut); and medicinal – *Chlorophora tinctoria*. However, *Hevea brasiliensis* and *Bertholletia excelsa* dwarf other NTFP in gross value, as compared to all other NTFPs of timber trees across the whole wet tropics (Prance 1990; Godoy et al. 1993; Boot and Gullison 1995). These products are so dominant that in certain regions of the Brazilian Amazon forest management practices of both indigenous and colonists are devoted toward their sustainable extraction (Anderson 1990; Fearnside 1990).

In Central America, there are higher volumes of merchantable timber trees per hectare with a larger proportion also yielding NTFPs. However, particularly relative to Asia, NTFPs appear to have greater proportional values than timber – primarily because timber yields are lower. Fruit trees and medicines have been shown to have a high proportional value particularly in the seasonal semievergreen rainforests of the Caribbean and the Yucatan (Mendelsohn and Balick 1995; Ricker et al. 1999).

12.6.3 Southeast Asia

As with the other two regions, the post-World War II period also saw an increase in Asian timber on world markets. The first areas exploited include the Malay Peninsula, Sabah, and the Philippines, where timber supplied European, Japanese, and American plywood imports. Deforestation rates here rivaled those of West Africa and the Atlantic coastal are of Brazil.

Asian timber trees also have many nontimber uses (Table 12.1b). Some species have very important past histories – for example *Palaquium gutta* (gutta percha) produced latex that was originally the sole source of insulation for submarine telephone lines in the early 1900s (Burkill 1966). Now gutta percha is used in dentistry to fill the temporary space of a tooth cavity before a permanent filling (De Beer and McDerrmott 1989). Many of the trees in the Dipterocarpacae produce resins (e.g., *Shorea javanica, S. stipularis*) (Damar) that is not only important incense for religious ceremonies but also used for varnishes and paints (De Beer and McDerrmott 1989). *Aquilaria spp*., (Gaharu) is tree that is relatively rare and scattered within the Asian rain forests, is also highly sought after for the perfume scent of its wood and its resin has been traded since the first century for production of incense (jos sticks) in religious ceremonies (Burkill 1966). Other species of *Shorea* (*Shorea stenoptera* –Borneo; *Shorea section Doona* –Sri Lanka) produce nuts (illipe) that are an important source of food, or oil for cooking and is a base for creams and lotions (Ashton et al. 2001b). Some of the legumes (*Koompasia* spp., *Intsia* spp.,) are the largest emergent trees in the forest and, because of their structure and open ventilation, provide important sites for hives of honey bees (De Beer and McDerrmott 1989). *Mesua ferrea* and *Eusideroxylon* spp. produce some of the densest wood used in temple construction and roofing, respectively (Ashton et al. 1997).

Many of the commercial edible fruits are from *Mangifera* spp., (mango), *Durio* spp., (durian), *Artocarpus* spp., (jak, breadfruit), *Garcinia* spp., (mangosteen), *Syzygium* spp., (jambu)., *Euphoria* spp., (langsat), and *Nephelium* spp., (rambutan). These fruits are recognized throughout the world's supermarkets. What is not recognized is that there are many more fruit varieties and species from these same genera sold on local markets and obtained directly from the rain forest compared with the single species cultivated for each genus for mass production (Burkill 1966; Jessup 1981).

Forest disturbance from agricultural clearance, cultivation, and subsequent fallow is a common practice in the forests of Asia. Swidden farmers have long used early successional species that come in after agriculture for medicinal and food crops (De Jong 1997; Ashton et al. 2001b). Plant families that produce important medicines and are also considered mostly early successional include the Solanaceae (potato family), Rutaceae (citrus family), and Verbenaceae (verbena family) (Burkill 1966).

Lastly are the monocots – palms and grasses (Bamboos). NTFPs include cordage and rattan, mostly from the climbing palm (*Calamus spp*., rattan). Taken together across Indonesia, Thailand, India, Cambodia, and Vietnam, this is a multimillion global industry in basketry, furniture, and artisan use (Weinstock 1983; Ashton et al. 2001b). Sugar palms (e.g., *Caryota urens*) that produce sugar and in many cases beverages, wines, and whiskeys (Ratnayake et al.), and the betel nut (*Areca catechu*), an important component of the Asian chewing addiction "betel" are now multimillion worldwide commodities that are mostly cultivated from village gardens or from within the forest itself. The bamboos in addition are widely used for village housing, basketry, and weaving and in the local and national construction industry for scaffolding. Taken together, timber and NTFPs are an important, but undervalued commodity within the Asian rain forests (De Beer and McDermott 1989).

12.7 Forest Stand dynamics

To practice silviculture in rainforests to insure the sustained production of timber and NTFPs, it is imperative that a solid understanding of rainforest dynamics is available. Unfortunately in most rainforest regions, this appears not to be the case. Most management has been applied based on assumptions and false perceptions of rain forest dynamic. Based on the growing experimental evidence from research, there are six common principles that hold true for evergreen and semievergreen rain forests across the wet tropics (Ashton 1998; Ashton et al. 2001a). We list them in order.

12.7.1 Disturbance and regeneration

Principle 1. After disturbance both pioneer and late seral species are simultaneously released together to form a new stand. Research has clearly demonstrated that initial floristics is the main driving mechanism for successional development of temperate forests (Oliver 1981; Oliver and Larson 1996). There is a growing body of literature that is showing this to be the case in tropical forest regions – especially dominated by the timber-rich tree species, most of which are relatively shade intolerant (Ashton and Peters 1999; Hall et al. 2003a, b; Baker et al. 2005) (Fig. 12.2).

A lot more research needs to be done documenting pattern and process of regeneration but many studies suggest episodic disturbance regimes that promote initial floristics are a much more important driver than what was once assumed. Many of the larger disturbance regimes can be attributed to hurricanes and typhoons (Lugo et al. 1983; Basnet et al. 1992); droughts and fires (Leighton and Wirawan 1986; Meggers 1994; Hammond and ter Steege 1998) and flooding and landslide

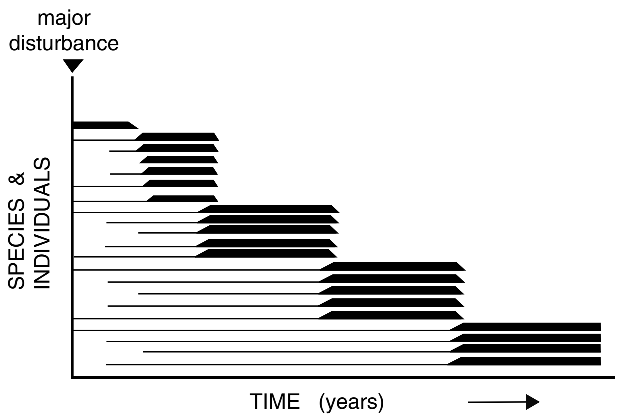

Fig. 12.2 The Initial Floristics model. Lines depict when establishment of individuals and/or species occurs after a major stand replacing disturbance to a forest. Thickened bars depict when species dominate the canopy of the new initiating stand over sucessional time. Species with short lines can be considered pioneers while those with the longest lines would be long-lived late successional. This graphic is adapted from Oliver and Larson (1996)

events (Gullison et al. 1996; Gunatilleke et al. 2006). In addition, large areas of tropical forests have long records of land clearance and cultivation currently practiced through swidden agriculture particularly on poor soils of the ever-wet rain forests (Conklin 1957; Weinstock 1983; De Jong 1997; Ashton 2003) or through permanent agricultural systems that have long since been abandoned – usually on more fertile soils of semievergreen forest regions (e.g., Meggers 1994; Cooke et al. 1996; Piperno and Becker 1996).

Principle 2. Disturbances are not uniform in severity, type, and extent across the forest topography. Most work now demonstrates that release type (allows ground story plants to survive) natural disturbances dominate over lethal disturbances (eradicates ground story plants). In addition, disturbances vary in relation to topography (slope position, aspect) (Gunatilleke et al. 2004; Ediriweera et al. 2008); or soil type, hydrology, and fertility (Becker et al. 1988; Grogan et al. 2003). Droughts affect ridge forests with shallow to bedrock soils more than lower lying deeper soils (Becker et al. 1988; Ashton 1992; Ediriweera et al. 2008). Ridge soils with perched water tables and deep weathered in-situ soils can endure droughts more consistently than lower lying deep sandy soils (Grogan et al. 2004).

Greater topographic relief promotes greater differentiation in the type, frequency, and scale of tree fall disturbances (Whigham et al. 1999). The nature of size and frequency of tree falls change with soil type (Kapos et al. 1990; Poorter et al. 1994), with elevation (Brokaw and Grear 1991) and with landform (Bellingham 1991; Gale 2000). For example in Sri Lanka, lightning strikes are more frequent on higher grounds; tree fall size and frequency differs, with greater numbers and more frequent tree falls in lower lying more water-logged areas than on soils with deeper rooting (Gunatilleke et al. 2006; Ediriweera et al. 2008).

Stronger and larger scale episodic disturbances are found in higher latitude forests of the seasonal tropics (fire, hurricanes, typhoons – Lugo et al. 1983; Whigham et al. 1999; Baker et al. 2005) as compared to rain forests around the equator (Whigham et al. 1999; Ediriweera et al. 2008).

Summarizing, forest types are more shade tolerant, longer-lived, and more resource conservative when evolved with small-scale, intermittent, noncatastrophic disturbance regimes. Forest types exposed to frequent, large-scale, more catastrophic disturbances regimes comprise more shade intolerant, faster growing, and more opportunistic species.

12.7.2 Regeneration Guilds

Principle 3. Tree species in wet tropical forest can be dispersed and can establish in a number of different ways based upon regeneration origin, successional status, and growth habit (see Table 12.2). Compared to other forest types in boreal, temperate or dry tropical regions, the number of species groups (guilds) with similar characteristics of regeneration is more diverse (Grubb 1977). However, each group

Table 12.2 Autecological descriptions of six major regeneration guilds of tropical trees. Guild have been organized by successional status and growth habit

Guild	Autecology	Genus examples
Pioneers of initiation	Frequent production of small flowers and seeds. Seeds dispersed by wind, water, small birds, and bats. Very fast growing – with often umbrella-shaped crowns. Seed germinates best on mineral soil.	*Cecropia*, *Macaranga*
Pioneers of stem exclusion	Similar reproductive traits as above. Fast growing – with columnar – often monodominant stems, dense foliage.	*Alstonia*, *Alnus*, *Ochroma*
Canopy dominants of later succession	Infrequent, large flowering and fruiting events. Dispersed by gravity, predation by rodents, large birds. Shade intermediate/tolerant seedlings. Advance regeneration that carpets forest floor. Site-restricted distribution. Growth habit – columnar crown that becomes expansive as a canopy emergent.	*Dipterocarpus*, *Shorea*
Canopy non dominants of later succession	Regular flowering and fruiting. Larger, relatively few flowers and fruits. Dispersed by large animals. Site generalist. Density dependent, prone to Herbivory and pathogens	*Mangifera*, *Myristica*, *Durio*
Subcanopy trees of later succession	Shade-tolerant trees, Columnar crowns. Similar to canopy-non dominants in reproduction, dispersal, and establishment. Greater reliance on vegetative propagation	*Garcinia*, *Semecarpus*
Understory trees of later succession	Planar crowns with low spreading growth habits. Shade tolerant. Flowering and fruiting in small amounts. Reliance on vegetative propagation	*Humboltia*, *Psychotria*

Examples of genera are from Asian tropical forests tends to comprise species with more restricted abilities to regenerate via other means, meaning that species tend to rely on one dominant method for successful establishment (Ashton et al. 2001a). Diversity of regeneration guilds in tropical forests has been well documented – for example examining only dispersal agents – wind, small birds, bats, large birds, and bats and mammals – are all important. Seed can remain dormant in the mineral soil waiting for a change in environment after a disturbance; or it can be dispersed after a disturbance (usually by wind, small birds, or bats), and germinate immediately – both these modes of regeneration would be regarded as a pioneer strategy (Whitmore 1998). Most seeds of pioneers are small, desiccation resistant, and require high light regimes and mineral soil for best germination and seedling growth (Swaine and Whitmore 1988; Whitmore 1989; Turner 2001).

However, the largest numbers of canopy tree species in rain forest rely upon modes of regeneration establishment that allow for their survival as seedlings and seedling sprouts (canopy dominant and nondominant trees) in the forest understory (Richards 1996; Turner 2001). However, species show very different abilities to endure and survive in understory shade (Ashton et al. 1995; Turner 2001). Another mechanism for regeneration to survive is in the form or dormant vegetative buds on the basal stem and roots of existing trees (mostly subcanopy and understory shrub/tree species). In both instances, species take a conservative "waiting game" enduring low light and soil resources until a change in the canopy environment through disturbance. In the case of the seedlings, the growth release is immediate but reliant upon their own existing but small root system (Turner 2001). In the case of vegetative buds that become coppice shoots and root suckers, the growth response can be very strong given the reliance upon an existing parent tree root system that can be large and extensive. The terminology for this adaptive strategy is called advance reproduction (regeneration). Forests that have co-evolved with release disturbance regimes are heavily reliant upon both modes with greater proportional representation of vegetative regeneration (including canopy trees) in tropical climates that have greater, large-scale release events such as hurricanes, typhoons, and floods (Whitmore 1998). In addition, what is also important to note is that the diversity of the regeneration guilds, and in particular those that rely upon advance regeneration, can be directly related to the complexity of succession and the structural diversity (strata) of late successional forests (Table 12.2) (Turner 2001).

12.7.3 Stand Stratification

Principle 4. Two forest canopy stratification processes contribute to the dynamics of the species mixtures (Ashton and Peters 1999, Ashton et al. 2001a). One process is whereby the long-lived species mixture occupies different vertical strata within late-successional stages of a stand canopy development. This structure has been commonly documented in the classic tropical ecology literature by Davis and

Richards (1933, 1934), Ashton (1964), and many others. In this process, different species attain maturity at different heights (representing different strata). For simplification, we have identified three strata – understory, subcanopy, and canopy (including emergent) (Table 12.2).

The second process considers species mixtures of different successional status. In this case, species attain the top of the canopy sequentially, all having originated or been released after the same initiating disturbance. This process has been less well documented primarily because of the difficulty in identifying rings that can be related to annual or periodic growth and age (Bormann and Berlyn 1981); but some studies exist demonstrating this for tropical rain forests where there is more marked seasonality (Worbes et al. 2003; Baker et al. 2005; Zimmer and Baker 2008). The alternative is to record dynamics of diameter and height growth differentiation in permanent long-term plots over time from very early to mid- successional periods when stratification in the canopy is at its highest. Very few plots exist that either capture early-to-mid successional processes (most plots have been placed in mature rain forest) or that have been monitored a sufficient length of time documenting height as well as diameter increment (the longest measured plots in the tropics are 60+ years old – within mature forest). However, many studies of species mixtures in moist temperate broadleaf forests have documented this process (Oliver 1978; Oliver and Stephens 1977; Kelty 1986; Smith and Ashton 1993; Liptzin and Ashton 1999).

12.7.4 Mechanistic Drivers of Canopy Tree Distribution

Principle 5. Many species within the same genus are site restricted to differing combinations of disturbance type, soil nutrient, and soil moisture status within the forest topography. Such species represent high basal areas within the localized stands (Ashton 1988, 1989), and usually represent the canopy-dominant late-successional category defined in Table 12.2. Species are site restricted because of either one or a multiple of factors that promote their relatives to be in close proximity, and therefore to have a clumped distribution.

The most obvious tree family that shows restricted species distribution is the Dipterocarpaceae of tropical Asia. Many studies have demonstrated their distribution in relation to soil type (Ashton 1964; Ashton and Hall 1992; Gunatilleke et al. 1996); elevation (Gunatilleke et al. 1998); soil nutrition (Gunatilleke et al. 1997; Palmiotto et al. 2004; Paoli et al. 2006); soil water status (Delissio and Primack 2003; Ashton et al. 2005); and shade tolerance (Ashton and Berlyn 1992; Ashton 1995; Ashton et al. 1995; Bloor and Grubb 2003; Leakey et al. 2003; Aiba and Nakashizuka 2005; Barker et al. 1997).

Other important timber trees exhibit similar patterns in relation to soil and light resources: West Africa (*Khaya* – Riddoch et al. 1991; *Triplochiton* – Swaine 1996); Central Africa (*Entandrophragma* – Hall et al. 2003a, b, 2004); Central America (*Dipteryx* – Clark and Clark 1987; *Virola* – Fisher et al. 1991; *Cordia* – Popma and

Bongers 1988; Moraceae – Strauss-Debenedetti and Bazzaz 1992) and the Amazon (*Swietenia* – Grogan et al. 2003, 2004).

More generally tropical forests have been demonstrated to exhibit species, genera, and family differentiation in relation to shade and soil resources as ecological communities within the forests (Turner 2001; Silvertown 2004). Regional examples are for: West Africa (shade tolerance – Kwegsisa and Grace 1986; Veenendaahl et al. 1996; Agyeman et al. 1999; nutrients and soil – Nye 1960; Hoberg 1986; Veenendaahl et al. 1996) and Central America (nutrients and soil – Clark et al. 1998; 1999; Johns et al. 2007; shade tolerance – Brokaw 1985; Strauss-Debenedetti and Bazzaz 1992; Poorter and Markesteijn 2008); soil moisture availability – Englebrecht et al. 2007) and the Amazon (nutrients and soil – Johns et al. 2007).

In summary, there is a voluminous and growing literature demonstrating many tropical tree species to have limited site and distribution patterns. Most important tropical timber species exhibit such tendencies.

Principle 6. Some tree species can be found throughout the forest but at relatively low populations irrespective of site and topography. Such species are usually density dependent, meaning that for their successful germination and survival seedlings do better further away from their parent tree seed source. This is because regeneration closer to parent trees has a greater susceptibility to pathogens, predators, and herbivores.

There are numerous studies demonstrating density dependence in tropical trees (Janzen 1970; Augspurger 1983; Harms et al. 2000; Volko et al. 2005). This phenomenon increases in forest types toward the equatorial latitudes such that it is highest in tropical wet evergreen forests. It is an important attribute to consider when managing such forests because simplifying species composition and structure in forests with strong density dependence will make them much more susceptible to tree disease and herbivory. Such tree species usually represent the canopy nondominant late-successional category defined in Table 12.2.

12.8 Applications to Silviculture

Using the principles outlined above, regeneration methods for wet evergreen and semievergreen topical forests can be divided into two main methods – *shelterwoods* or *selection* and the variants of both. A third method – *seed tree* – can be regarded as a viable possibility for shade intolerant, short- or long-lived pioneer timber trees that are heavy seeded but need very open conditions, with at least partially scarified soils, for best recruitment and establishment. Seed tree methods can be considered the technique that emulates the most extreme disturbances to site (e.g., a combination of hurricane followed by fire; or sublethal land clearances such as that practiced by swidden agriculturalists). In general across the forest landscape, such disturbance regimes are rarer and trees that rely upon them tend to be very localized and patchy in distribution.

Both Shelterwood and selection methods focus on the establishment and release of advance regeneration and as such are focused on facilitating and protecting advanced regeneration of canopy trees before removal of the canopy. Shelterwoods can be considered more appropriate for more shade-intolerant canopy trees or forest types that are driven by strong episodic disturbance regimes, where initial floristics and site-specific treatments are clearly demarcated across topographic gradients by floristic associations that are distinct.

Selection methods are more appropriate for shade-tolerant, slower growing canopy trees or forest types and where disturbances regimes are small and frequent ensuring the progressive movement of trees upward toward the canopy. Selection methods are more appropriate for floristic associations that less distinct and more general and extensive in distribution.

Throughout the tropics both regeneration methods have a varied history of development. In the more recent literature, the terms *polycyclic* and *monocyclic* have been used to emphasize the difference between selection and shelterwood methods in the nature of the periodicity of entry. In reality, this is a gross simplification since most shelterwoods can and do work with multiaged cohorts (2–3 age classes); and most selection methods have, by economic and biological necessity, been pushed toward longer and longer cutting cycles because of logistical, economic, and ecological necessity.

The following sections describe the regeneration methods in detail. However, it is important to emphasize that though these methods have been developed – and have long histories in certain tropical forest regions – in reality only a small fraction of the rain forests that are even considered as "managed" are actually employing such techniques with careful regulation. Unfortunately, most tropical forests are logged with no silvicultural regeneration method in mind. In addition, most countries with tropical forests have not developed a sufficient knowledge base on ecology or silviculture to manage such forests. We would consider even some of the "best managed" forests with applications of "RIL" to be in this category – which in practice comes down to insuring careful logging but not necessarily continued sustainability.

12.8.1 Shelterwood Methods and Their Variants

The shelterwood method of regeneration comprises a set of canopy treatments focused on securing advance regeneration when absent, and then releasing this regeneration as a single cohort once established across the stand in a relatively uniform manner (Smith et al. 1997).

Shelterwood systems in Asia have a diverse and long history. Mixed dipterocarp forest types that are relatively shade intolerant (red meranti type) and that contain merchantable volumes of timber greater than 40 m^3 ha^{-1} are very suited to the classical uniform shelterwood whereby preparatory treatments are focused on establishing advance regeneration of the shade-intolerant canopy trees prior to canopy removals. Mixed-dipterocarp forests can range in merchantable timber volumes from a low of 25 m^3 ha^{-1} on infertile or drought-prone ridge sites or low lying hydric swamps, to a high of more than 200 m^3 ha^{-1} on fertile lowlands (Nicholson 1979; Bertault and Sist 1997; Ashton et al. 2001b). This contrasts with the mahogany-rich forests of Africa and South America where volumes of commercial timber rarely exceed 40 m^3 ha^{-1} (0.5–2 stems ha^{-1}), when averaged across a large forest area (Verissimo et al. 1995, Barreto et al. 1998, Sist and Nguyen-The 2002). However, such studies often ignore the clumped distribution patterns of the timber trees (i.e., *Swietenia, Khaya, Entrandrophragma*), which, if examined as stands (e.g., distinct floristic association), instead of averaging across the whole forest, may sometimes match volumes for mixed dipterocarp forest.

Shelterwoods are also ideal systems to use where canopy-dominant species, such as dipterocarps and many of the long-lived canopy emergents in the leguminosae (*Adenanthera, Cedralinga, Intsia, Koompasia*) and Meliaceae (*Carapa, Cedrela, Entandrophragma, Guarea, Khaya*) mast or exhibit differential seed production from year to year. After germination, these species often require a dramatic increase in light for satisfactory establishment beneath the forest canopy and subsequent release.

Simple systems that were developed for the Asian mixed-dipterocarp forests were first successfully implemented, where advance regeneration was easily secured, or almost always present before any treatments were made. Such systems could therefore bypass the preparatory routines used to first secure it. Examples of shelterwoods whereby the canopy is removed usually in one cutting would be: (1) Malay Uniform System for lowland forest stands dominated by *Shorea leprosula*, *Dryobalanops* aromatica, and other red meranti associates (Wyatt-Smith 1963); (2) the seed tree system adopted for *Dipterocarpus zeylanicus* stands in the lowland southwest of Sri Lanka (Holmes 1957); or (3) the single cutting systems developed for moist sal (*Shorea robusta*) stands in Uttar Pradesh, India (Joshi 1980). These systems worked primarily because they were applied to forest which were dominated by one-or-two shade-intolerant dipterocarp species that mast and regenerate prolifically. These forest types were usually on fertile lowlands (riverways, valleys, floodplains, coastal), fast growing, and typically had high merchantable volumes. Much of this forest has been cleared for agriculture (oil palm, rubber) and only fragments remain.

In more floristically rich upland hill regions of tropical Asia, dipterocarp forest types are often poorly stocked with advance regeneration. More complicated shelterwood treatments that follow the classical methods to secure advance regeneration are used. Here, shelterwood treatments first ensure establishment of advance regeneration through a series of preparatory cuttings that remove the understory and subcanopy before the canopy trees are removed. Mixed-dipterocarp forests of the uplands are slower growing, more shade tolerant with lower merchantable timber volumes. Once advance regeneration has been satisfactorily established and released, canopy trees can often be left within new stands as an older overstory reserve age class for increased structure and habitat for other values and increased shelter for the young stand beneath. For example, in the Andamans, the understory is gradually raised in a series of preparatory and establishment cuttings such that the regeneration grows to a pole-size class before the overstory is partially taken off (Chengappa 1944). This means that at any one time at least two age-classes of trees are present within the stand and over many periods there are three.

In the mixed-dipterocarp forests of the Western Ghats, the partial removal of the canopy and the complete removal of the understory and subcanopy over about a decade allows for the establishment and release of very shade-tolerant regeneration susceptible to dieback from overexposure to direct sun (Kadambi 1954). Recent work in Sri Lanka, for a similar forest to that of the Western Ghats, has developed shelterwood systems that differ in their aggressiveness and timing of canopy removals in relation to topographic position from valley to ridge (see the case study in Chap. 13). On ridges fifty percent of the overwood is retained as reserves for shelter. Mid-slopes retain twenty-five percent of the overwood as reserves for satisfactory establishment, whereas on valley sites regeneration is usually already present and requires release through overstory removal (Ashton et al. 1993, Ashton and Peters 1999).

Few examples exist in the use of shelterwoods in Africa. In Ghana, there were systems that were tested in the 1950–1960s and proved promising (Mergen and Vincent 1987); however, most of this work has been lost and the forests converted to Cocoa. It would appear that unlike the dipterocarp forests of Asia, the use of shelterwoods can only be applied in forests with high standing basal areas of shade intolerant timber trees that were driven by distinct cohorts – such as many of the canopy and emergent shade-intolerant long-lived legumes and species in the Meliaceae (see the case studies by Hall and Grogan et al. for a neotropical example, this volume).

The classic work on shelterwoods in the neotropics was originally conducted in Trinidad and Tobago in the Mora forests (*Mora excelsa*) using the periodic block system that comprises a one-cut removal and release of regeneration (Bell 1969). However, research on this stopped several decades ago. Recent work in the neotropics, primarily in Costa Rica and Bolivia, would support the suggestion that shelterwoods are suited to forest types with shade-intolerant canopy timber trees. In a series of treatments for a *Pentaclethra*-dominated forest that reflected intensity of canopy removal, the understory and subcanopy shade-tolerant species grew slowly with only slight increases in response (particularly Sapotataceae) to heavy canopy removals, but the regeneration of canopy and emergent species considered long-lived pioneers and shade intolerants (e.g., *Vochysia ferruginea* and *Jacaranda copaia*) showed good growth response with median increments of 1.6 cm per year (Finegan et al. 1999). However, Finegan and Camacho (1999) suggested that increments were higher under more directed liberation treatments for most commercial timber species and that trends over time in mortality rates under the most intense silvicultural regime required further study before any conclusions could be made.

In Bolivia, Peña-Claros et al. (2008) found that the most intensive silviculture treatment showed the greatest density and growth rates for regeneration of the commercial species, which were mostly long-lived pioneers (Meliaceae, Leguminosae), even through the shade-tolerant species were more common but did not respond as well. This is corroborated by an older set of studies in the upper Amazon of Peru, where, through a community landowner cooperative, 30–40 m wide strip shelterwood cuts released advance growth of seedlings and sprouts of mostly fast-growing commercial shade-intolerant and pioneer species (Hartshorn 1989, 1995).

Finally in a study in lower Amazon of Pará State, Brazil 11–12 years after being subjected to differing levels of above-ground canopy removals, postharvest basal area growth generally increased with harvest intensity, such that by 10 years the canopy removal treatments were 60–80% of the basal area of those in the control plots. Treatments stimulated recruitment and growth of residual trees, particularly in the smaller diameter classes, but had little effect on species richness, while the tree flora within all harvest treatments was broadly similar to the undisturbed (control) plots. The more intensive canopy removal treatments, especially the one-cut shelterwood, were dominated by a higher proportion of short-lived, early successional tree species.

12.8.2 Selection Methods and Their Variants

Selection regeneration methods rely upon short cutting intervals to promote small gap disturbances and to open up growing space enough to insure continuous conditions for regeneration establishment and timely release in small heterogeneous patches, with the objective of creating an all-aged, all-sized stand (Smith et al. 1997). Such all-aged, all-sized stand structures can be maintained in relatively simple mixtures of shade-tolerant species such as those systems first developed in the beech-spruce-fir mixtures of the alps, where the book-keeping that monitored the stratification process was manageable and where stands are thinned and regeneration liberated at the same time as the larger timbers are cut (Smith et al. 1997). Application of selection treatments in tropical rain forest is an altogether different phenomenon. Imagine trying to keep track of more than 50 species per hectare.

The Philippine Selective System was the first selection system in the Asian region to be adopted for mixed-dipterocarp hill forests (Reyes 1968; Appanah 1998). The system was developed in the 1950s and purposefully attempted such techniques with moderate success by insuring that 70% of all timbers between 20 and 60 cm remained after removal of the large timbers, thinning the canopy, and liberating advance regeneration. However, many of these forests had logging operations that did not adhere to such guidelines and were overcut at each entry, the land degrading to grassland and scrub.

In the 1970–1980s, selection systems were developed in Indonesia and Malaysia for mixed-dipterocarp forests (Appanah 1998). In Malaysia, this was primarily because all lowland forest had been converted to oil palm and all that remained were the more difficult to regenerate hill forests. The system can be described as a repeated diameter-limit cutting (all stems $\geq$50–60 cm dbh are removed). The cuttings are done on a cycle aka polycyclic fellings – Whitmore 1998) that were originally planned at 15-year intervals, but when the expected growth and species composition did not develop, intervals moved to thirty, and are now in some cases at between forty and sixty years (Sist et al. 1998), depending upon the site and soil fertility. Such developments suggest these systems are moving away from attempting balanced all-sized stands to systems that now resemble an irregular shelterwood in age class distribution but not in size class and stratification (structure), or species composition. This is because even the regeneration of the most shade-tolerant canopy tree species need timely assistance in reaching the canopy, and this consideration has been ignored (note the Philippine system attended to this with thinning and liberation treatments). Shelterwoods attain this through insuring that the canopy stratification process is protected by avoiding any major, potentially disruptive, entries into the stand until after the stem exclusion stage of development (Sensu Oliver and Larson 1996).

Selection systems in other regions that have proven workable are few and far between. The best known system is the CELOS system developed in Suriname for the relatively shade tolerant forests comprising *Mora excelsa*, *Ocotea* spp., and *Peltogyne venosa* (De Graaf 1986). In this system, careful tending of remaining commercial crop trees through thinning and liberation treatments that selectively killed vegetative competition was important at each entry (20- to 30-year intervals between entries) that also harvested 20–40 m^3 ha^{-1} of merchantable timber. In a comparison with a more extensive 100-year return interval with little silvicultural input, the CELOS system has a higher financial and development return (De Graaf et al. 2003). However in the Atlantic coastal forest of Costa Rica, dominated by a related species *Peltogyne purpurea*, densities of adult trees remained constant 15 years after selective logging and density of trees 10–30 cm dbh actually decreased (Lobo et al. 2006). Predictions, based on a 15-year cutting cycle, as practiced in Costa Rica, will lead to a significant reduction in reproductive individuals (Lobo et al. 2006).

Although species such as *P. purpurea* may have an autecology compatible with selection systems, there are more than enough examples of other species generally more shade intolerant, where “selective logging” has proven incompatible. For example, in the Atlantic coastal forest of Nicaragua, selective harvests at a 70 cm dbh diameter limit of the shade-intolerant *Carapa nicaraguensis* (Meliaceae) trees removed 6.3 trees ha^{-1} yielding a merchantable volume of 45.8 m^3 ha^{-1} reducing stand basal area by 20%, with 18% residual damage (Webb and Peart 1999). No evidence was recorded of successful regeneration and growth. Similarly in the *Entandrophragma* spp. (Meliaceae) dominated forest of Central Africa, Hall et al. (2003a, b) demonstrated the degradation of commercial timber species and shifts in forest structure and composition through selective logging.

Selection systems have largely been favored in the recent decades mostly because we now concern ourselves with what we think forest structure should look like by slavishly attempting to imitate small-scale natural disturbances, rather than maintaining the integrity of the forest dynamic, particularly the stratification process in a more logistically feasible management unit. Selection has also been a term used in reference to “selectively” taking the largest and most valuable trees (Hawley 1935), and leaving the rest without insuring both the satisfactory establishment and release of the regeneration of those species selectively removed.

Selective cutting is desirable financially in the short-term (large initial financial returns with high discount rates), but not in the long-term (Ashton et al. 2001b). Nevertheless, selection systems when applied appropriately can be suited to certain shade-tolerant forest types with lower commercial timber volumes, slower-growth, and fragile soils and slopes.

12.8.3 Seed Tree methods

Further examination should be given to the use of seed tree regeneration methods, which could be very appropriate in small stand scale treatments to patches of shade intolerant long-lived canopy tree species that are site restricted, require a nearby seed source for effective dispersal, disturbance to the soil surface (scarification) for successful germination, and partial eradication of competing advance regeneration for satisfactory seedling establishment and growth. Seed tree methods have worked satisfactorily within the semievergreen forests of Indochina for species such as *Anthocephalus chinensis*, *Chukrasia tabularis*, *Dipterocarpus* spp., *Tectona grandis*, *Terminalia* spp., *Tetrameles nudiflora*, *Vitex* spp. (see the voluminous Indian literature reviewed in the classic volumes of "Troupe's Silviculture of Indian trees Vols. 1–8" revised edition 1992).

No such studies have been done in the neotropics, but given the nature of the distribution and natural disturbance regimes of mahogany (*Swietenia macrophylla*) in the upper Amazon (Gullison et al. 1996) and the Yucatan, such treatments would appear very suitable for this species along with others with similar autecological traits such as *Anacardium excelsum*, *Brosimum* spp., *Cordia alliodora*, *Dipteryx* spp., *Enterolobium* spp., *Tabebuia* spp., and *Terminalia amazonica*.

Based on the work by Hall et al. (2003a, b, 2004) in Central Africa, seed tree treatments that include adequate site preparation may be suitable on particular sites for *Entandrophragma* spp. and *Khaya* spp., and their associates, *A. congolensis*, *Petersianthus* spp., *Lovoa trichilioides*, *Nesogordonia papaverifera*, *Terminalia* spp., *L. alata*.

Similarly, the short-lived pioneers (see Tables 12.1a–c), and some of the long-lived pioneers (e.g. *Ceiba pentandra*, *Ochroma* spp., *Triplochiton scleroxylon*, *Aucoumea klaineana*) would be best regenerated through true patch clear-cut systems that are discreet and site specific. However, no research has been done to fully investigate such methods in a systematic and experimental way and that examines such effects on adjacent stands managed for other longer-lived species.

12.8.4 Added Benefits: NTFPs

NTFPs have been touted as a savior to tropical forest conservation (Peters et al. 1989a, b). However, NTFPs are as likely to be overexploited with increased commercialization and market demand as timber, and given very patchy distributions their valuation has often been overestimated. Accurately assessing impact of harvesting and the development of silviculture systems for NTFPs is essential for their sustainable management (Hall and Bawa 1993).

In regeneration methods for timber trees (seed tree, shelterwoods, selection), other resources can be and should be captured. NTFPs can be a significant complementary asset to timber production, particularly in low wage, rural economies. Some studies exist that have examined the affects of logging on forest structure, damage and regeneration for useful plant species and for the plant community as a whole (Salick 1992; Salick et al. 1995; Peters 1996; Van Dijk 1999). However, knowledge of NTFPs and their incorporation into silvicultural systems has been lacking.

This should not be the case – timber and NTFPs are interrelated in many ways. Most commercial timber species have important nontimber uses (Tables 12.1a–c). But poor timber harvesting treatments can reduce availability of both and can also directly affect NTFPs that are extracted from understory plants destroyed in the logging operation. Even when good silvicultural practices are used to liberate commercial tree species from forest understories, attention must be given to insure understory NTFPs are protected and cultivated. However, silvicultural treatments to remove canopy timber trees can create habitats for the many NTFPs that prefer open conditions.

NTFP cultivation and purposeful forest management will work best when there are well-established regional markets for forest spices, medicines, and forest foods. For example, a large number of the NTFPs sold in markets in Bata, Equatorial Guinea, such as *Afrostyrax* spp., *Ricinodendron heudelottii*, *Aframomum* spp., *Tetrapleura tetraptera*, and *Garcinia kola* are imported across sometimes great distances from Cameroon (Sunderland 1999). A similar case can be made for various food/fruit varieties of *Durio* spp., *Mangifera* spp., *Garcinia* spp., *Artocarpus* spp., *Parkia* spp., along with *Calamus* spp., in markets in Singapore that contribute significantly to the income of forest peoples in Borneo (de Beer and McDermott 1989).

In shelterwoods, NTFPs can be managed to mature sequentially over the course of stand development. Early seral species of herbs (spices, medicinals) can be cultivated and harvested within the first five years after a regeneration treatment. Vines and lianas (often used for medicinal purposes or basketry) can be harvested after twenty years of cultivation. Subcanopy and canopy trees of late seral stand development can provide important supplies of fruits, nuts, and syrups. An example of a sequential series of NTFP harvests over time within a shelterwood has been developed in Sri Lanka (see Ashton et al. 2001b and the case study by Ashton et al., this volume). Financial analyses show that cultivating and harvesting NTFPs within stands managed for timber can double or triple net present values (NPV) (see case study by Ashton et al., this volume; Ashton et al 2001b). However, selecting the right sites and species for NTFP cultivation is critical.

For selection methods of timber tree regeneration, all stages should be represented within the same stand and therefore NTFP in such stands represent all successional phases. This can be difficult given the cyclical and intrusive nature of the timber cutting cycle. What this does suggest is to cultivate NTFPs that are compatible with the length of the cycle (e.g., rattan, and other lianas of medicinal value; cardamom and other early seral shrubs that provide medicines and spices).

Certain forests are depauperate of NTFPs. Other forests have numerous species used for subsistence, which can be very useful for local communities, but are of low economic value. However, for the long-term sustainability of the forest, strong and diverse commercial NTFP markets can help enormously in complementing income derived from timber. Such enterprises also beg collaboration between state and local values and users for timber and nontimber resources, respectively. Provided proper silviculture is used and such crops are not overexploited, then forests stand to be a viable economic entity compared to other land uses. However, exploitation of NTFPs, like logging, has often led to forest degradation and eventual conversion to another land use.

12.8.5 The Need for Stand Level Planning Within Tropical Forest Landscapes

A stand can be defined as associations of species that comprise a species mixture with the same developmental and age-class structure, similar stocking, and that is often restricted to the same site quality. Stands are the base management unit for carrying out silvicultural operations to insure such treatments match the constraints of change in species composition, forest productivity, and therefore product yield. In tropical rain forests, their delineation is also the scale upon which land use planning identifies and protects riparian, wetland, and watershed values; ecological sites and floristic associations of unique richness, refugia, and rarity; and habitats of high importance to wildlife for foraging, breeding, or migratory corridors.

Like other regions with forests that have been managed for the long-term production of various timber and nontimber products, the landscape template should reflect an integrated network of stands allocated to production and/or protection. Stands as management units within tropical forests have all but been neglected in current management systems. Instead cook-book prescriptions with diameter limits and cutting cycles have been developed that ignore variations in site productivity and species distributions. Such prescriptions have been applied uniformly across very large forested landscapes (Appanah and Weinland 1990; Appanah 1998). Better stand delineation and the development of a more sophisticated partitioning of protection versus production represents the single greatest improvement that can be done within the most tropical forest management regimes.

12.9 Why Is Silviculture Not Applied? Is RIL Next Best Thing?

It is important to stress that the regeneration methods, silvicultural techniques, and spatial stand level planning for timber and NTFP described in the preceding sections is hard to find being applied in any tropical rain forest. It may be future action to strive for but it is an anomaly now. Instead, studies during the past twenty years have focused on "RIL." To most temperate foresters, this would be something that would seem to be common sense. Most forest laws in developed countries guard against such problems that physically reduce the fertility of the soil or affect the hydrology. Carefully planned skid trails, landing and road design; directional felling; and preparatory treatments to reduce damage – such as liana and vine cutting should all be done; but it is not good silviculture – it is what should be done in good harvesting operations. In most developing countries of the tropics, this may be required by law, it may be on the books, but such planning is largely ignored in practice. Numerous studies in tropical Asia and Latin America have now demonstrated that RIL is both cost efficient and logistically feasible (see Asia for: Pinard et al. 1995; Dykstra and Heinrich 1996; Pinard and Putz 1996; Bertault and Sist 1997; Sist et al. 1998; Sist and Nguyen-The 2002, See Latin America for: Verissimo et al. 1992; 1995; Johns et al. 1996; Holmes et al. 2002; Boltz et al. 2003). However, fundamental changes in the nature of the cutting cycle and method of cutting based on a diameter limit have not occurred (Putz et al 2008).

12.10 Summary Findings and Conclusions

Tropical ever-wet and semievergreen rain forests provide much more than products. They are usually the headwaters for downstream users of drinking water and irrigated agriculture. They serve to mitigate climate as a store house and sequestration site for carbon, and act to moderate surface temperatures and patterns and amounts of precipitation – functions that have global consequences if the forest is destroyed. It is therefore in our own best interests to protect and sustainably manage these forests.

Summary findings:

- Timber-rich forest types were most extensive in Asia with mixed dipterocarp forests – a type dominated by one tree family (Dipterocarpaceae), where commercial timber volumes are far higher per hectare (60–200 m3 ha^{-1}) than any other forest type.
- Starting in 1850, south Asia (India) and Peninsula Malaysia have developed the most voluminous and comprehensive silvicultural research compared to all other tropical forest regions.
- Given a related biogeographic history, South America and Central Africa have similar families and autecology of timber trees – Meliaceae, Moraceae, and the Sterculiaceae, as compared to Asia.

- The Leguminosae are important timber trees across all tropical forest regions and many form monodominant stands of trees (along with the certain Meliaceae species), along water courses and swamps. However, the most spectacular monodominant stands in Africa (*Gilbertiodendron*) are on upland sites, not in swamps.
- There are six common principles that hold true for evergreen and semievergreen rain forests across the wet tropics that concern the nature of disturbance, the role of advance regeneration, stand stratification, and species distribution.
- Shelterwoods can be considered more appropriate for more shade-intolerant canopy trees or forest types that are driven by stronger episodic disturbance regimes, where initial floristics and site-specific treatments are more clearly demarcated across topographic gradients. They are methods that are often neglected.
- Selection systems are more appropriate for shade-tolerant, slower growing canopy trees or forest types and where disturbances regimes are small and frequent ensuring the progressive movement of trees upward toward the canopy. They are methods that are often corrupted and applied to the wrong forest autecology.
- NTFPs can be a significant complementary asset to timber production, particularly in low wage, rural economies.
- At present, there is far too little careful stand level delineation and planning. Its implementation could markedly improve forest management, watershed protection, and biodiversity conservation.

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Chapter 13
Sustainable Forest Management for Mixed-Dipterocarp Forests: A Case Study in Southwest Sri Lanka

Mark S. Ashton, B.M.P. Singhakumara, Nimal Gunatilleke, and Savitri Gunatilleke

Abstract Mixed dipterocarp forests (MDF) were one of the most timber productive forests in the world. The forest region is named after the dominant canopy tree family Dipterocarpaceae within which two major genera (*Dipterocarpus*, *Shorea*) usually represent a disproportionate amount of the basal area. Compared with Latin America and Africa, MDF have much higher standing volumes of merchantable timber per hectare, but they were also the first to be over exploited and degraded (1970-ongoing). Most are now restricted to upland regions and require various restorative techniques if they are to be successfully managed. This study describes the silviculture of a financially viable management regime for southwest Sri Lanka. Our studies demonstrate that managing MDF for a combination of timber and enrichment plantings of nontimber forest products (NTFP) can be comparable to the most competitive adjacent land use – tea plantations. By managing for NTFP and timber, forest managers have new opportunities to solve the old problems of high-grading and land-use conversion. In addition, payments for ecosystems services that include water quality and yield, and credits for carbon would double forest value when compared with other competing land use values.

Keywords *Calamus zeylanicus* · *Caryota urens* · Cardamom · Dipterocarps · *Elettaria cardamomum* var. *major* · Enrichment planting · Light-hardwoods · Nontimber forest products · Rattan · Silviculture · *Shorea* spp.

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13.1 Introduction

Mixed dipterocarp forests (MDF) can be defined by the intimate mixtures of canopy trees, all in the family Dipterocarpaceae, that dominate the late-successional rainforest within the aseasonal wet tropics of Asia (Malaysia, Sumatra, Kalimantan, southwest Sri Lanka, and parts of the Philippines and Greater Celebes) (Appanah 1998). The tree family Dipterocarpaceae comprise 15–19 genera and 470–580 species (depending on differing taxonomic interpretations – Maury-Lechon and Curtet 1998). By far the majority of the genera and species are in Asia, with some species in Africa, and several in South America. Well-known genera within the family are *Dipterocarpus*, *Dryobalanops*, *Hopea*, *Shorea*, and *Vateria*. Many provide timber under various market names (e.g., luan, meranti, seraya, Philippine mahogany), making the mixed-dipterocarp forest region the largest exporter of hardwood timber on world markets during the major period of their exploitation (1965–1995).

The climate of mixed-dipterocarp forests can be considered perhumid with rainfall usually exceeding 3,000 mm year^{-1}, and mean annual temperatures of about 25°C with greater diurnal variations (±5°C) than seasonal and with annual dry seasons not usually exceeding 3 months. Soils comprise weathered in-situ Ultisols (red-brown earths) or less commonly Oxisols (latosols) (USDA 1975), which would be considered relatively infertile when compared with the volcanically derived soils of Central America, but comparable or slightly more fertile to the weathered Oxisols of Central Africa and the lower Amazonian terra firme region of South America.

Mixed-dipterocarp forests can be divided coarsely into lowland and hill forest types. Lowland forests are those below 200 m asl, usually restricted to the coastal plains and to areas not subject to flooding or inundation. Hill forests are restricted to elevations between 200 and 1,000 m asl and make up the foothills of the central mountain ranges of Borneo, Sumatra, Sri Lanka, the Philippines, and peninsula Malaya. After logging, many of the more accessible lowland forests were cleared for permanent agriculture and plantations (rubber, tea, oil palm). The remaining dipterocarp forests are now mostly restricted to the hill and mountain forests. Most have been logged over and are in poor condition.

In this study, we describe the ecology and protocols for sustainable management of hill mixed dipterocarp forest of southwest Sri Lanka. We believe it is one of only a few tropical forest examples of a silvicultural system that has been grounded on the ecological science of the forest with demonstrated financial sustainability. The management described in this study is based on 30 years of studies investigating the regeneration ecology, breeding biology, phenology, silviculture, and economics of trees, lianas, and shrub species that provide products and services for local and regional markets.

13.2 The Case Study Site Description

The greater region of Sinharaja (an MAB reserve and World Heritage Site) and surrounding forest reserves comprises about 50,000 ha, and is located in hill mixed dipterocarp forest in the ever-wet south-western region of Sri Lanka (6°21–26′N, 80°21–34′E). In Sri Lanka, the forest has been classified as a Mesua-Shorea community (De Rosayro 1942). The floristics has been well described (Gunatilleke and Gunatilleke 1981, 1983, 1985; Ashton et al. 2001a).

The forest largely comprises ridge-valley topography (200–300 m elevation difference) aligned northeast–southwest, with spurs and drainages that run across the slopes. The topography generally increases in elevation from the southwest toward the northeast from 200 to 1,000 m asl (Fig. 13.1). Seepage ways and many perennial streams cut across these slopes and run along the valleys. This landform is a result of differential weathering and erosion of less-resistant Precambrian metamorphic bedrock along structurally controlled parallel strike ridges and valleys (Cooray 1984; Erb 1984). The region receives 4,000–6,000 mm of rainfall per year. Most rain falls during the southwest (May–July) and northeast monsoons (October–January). Seasonal temperatures range between 25 and 27°C with a greater diurnal variation of (±4°C). Soils are classified as ultisols following the USDA (1975) terminology, or red-yellow latosols, using the classification system of the Food and Agriculture Organization (Moorman and Panabokke 1961).

13.3 Ecology of the Forest

To summarize the dynamics of the forest type, seven ecological principles can be considered. First, natural regeneration of mixed dipterocarp forest ultimately arises after many kinds of disturbance that are nonlethal; with late-seral canopy tree species occupying growing space before a disturbance, pioneers establishing immediately after a disturbance, and most shrubs and small treelets reproducing vegetatively. The process promotes the simultaneous initiation of pioneers and release of existing advance regeneration, and sprout growth into a new stand that is termed allogenic (initial floristics) – meaning that *all regeneration starts together* (Oliver and Larson 1996).

Second, all the tree species that dominate the canopy late seral-stages of forest development are dependent upon existing in the understory as advance regeneration. Disturbance regimes are therefore largely nonlethal to the vegetation in the understory.

Third, advance regeneration usually grows for at least a period of time beneath the intact canopy and so has to tolerate high amounts of shade (i.e., 0.5–1.0% of full sun – Ashton 1992a) and competition from understory shrubs. Advance regeneration species differ in shade tolerance, therefore promoting differing degrees of survival (Ashton and Berlyn 1992; Ashton 1995; Ashton et al. 1995; Gamage

Fig. 13.1 Location map illustrating remaining mixed dipterocarp forest in southwest Sri Lanka. The study site is depicted by the S and *arrow*. Alignment is north–south. Scale 1 cm to 15 km

et al. 2003; Singhakumara et al. 2003; Ashton et al. 2006). For example, *Shorea trapezifolia* has frequent masting events with low survival in understories (90% of a cohort is dead after 2 years), whereas *S. worthingtoni* has infrequent masting events once in every 5–7 y with high survival (50% of a cohort survives 20 years after germination) (Ashton et al. 1995). However, all species maintain the same capacity to initiate rapid height growth whenever a canopy disturbance releases them (Ashton et al. 1995).

Fourth, natural disturbances are not uniform in severity, type, and extent but change in relation to topographic position. In southwest Sri Lanka, trees on ridge sites and thin to bedrock slopes are susceptible to drought effects, particularly during El Nino years (1983; 1991–1992; 1997–1998), and also to lightning strikes. Mid slopes are prone to landslides, particularly with the onset of monsoons; differences in aspect also make exposed slopes prone to multiple tree falls and monsoon downbursts than more sheltered slopes (Ediriweera et al. 2008). Areas with streams and rivers that are seasonally flooded have trees with shallow roots that are prone to wind throws from sudden downdrafts channeled into valleys (Ashton 1992a; Ediriweera et al. 2008).

Fifth, tree species can be grouped into six broadly defined guilds based on ecological similarity of regeneration origin (advance regeneration, vegetative sprout, buried seed, and/or seed that colonizes after a disturbance), stage of stand development (stand initiation, stem exclusion, understory reinitiation – after Oliver and Larson 1996), and growth habit (understory, subcanopy, canopy). The six guilds and their approximate species numbers are as follows: (1) pioneers that dominate stand initiation (animal dispersed postdisturbance or buried seed banks; 10–20 spp.); (2) pioneers that dominate stand stem exclusion (wind or animal dispersed postdisturbance; 15–25 spp.); (3) late-successional canopy dominants (advance regeneration; masting; 55–75 spp.); (4) late-successional canopy nondominants (advance regeneration – density dependent; 15–25 spp.); (5) late-successional subcanopy (advance regeneration – density dependent; vegetative sprout; 35–45 spp.); and (6) late-successional understory (vegetative sprout; 60–80 spp.). The majority of guilds clearly represent regeneration that exist prior to disturbance ("advance" regeneration, vegetative sprout) for representation in the new stand (Ashton 1992b).

Sixth, two forest canopy stratification processes can contribute to the dynamics of mixed-dipterocarp forests (Ashton and Peters 1999). The first process includes those long-lived species that occupy different vertical strata within a mature forest stand. We have referred to this kind of stratification at the mature phase of stand development as "static stratification," with understory species representing smaller (as defined by their diameter and height class distribution) and more numerous individuals than the true canopy trees species. In a combined way, they represent the four late-successional regeneration guilds of the six described earlier (canopy dominant, canopy nondominant, subcanopy, and understory). The second forest canopy stratification process involves species of different stand developmental (successional) status that sequentially gain dominance of the canopy. We have defined this as "dynamic stratification." In this forest, *Macaranga peltata* (pioneer of initiation phase) attains the canopy of the mixture early in stand development, but this position is usurped first by *Alstonia scholaris* (pioneer of stem exclusion phase) and then by *S. trapezifolia* (late-successional canopy dominant). Both stratification processes occur over the course of stand development following initiation, stem exclusion, understory initiation, and old growth phases as described by Oliver and Larson (1996) (Fig. 13.2a, b).

Seventh and last, most of the late-successional canopy dominants of the forest (that often exhibit masting) are restricted to particular topographic positions within the landscape (Fig. 13.3a–d). Species differentiate in relation to frequency and type of disturbance, and differences in soil-water availability and soil nutrition (Ashton 1995; Ashton et al. 1995, 2006; Gunatilleke et al. 1997, 1998, 1996). Most late-successional canopy nondominant species are usually site generalists (see Fig. 13.3e, f), though they represent fewer numbers of species than canopy dominants. Late-successional site generalists in this forest type are usually dependent upon medium to large-sized animals (bat, bird, civet, primate) for effective long-distance seed dispersal. Many of the site generalist species exhibit density dependence (e.g., seedlings do not do well near parent trees because of proneness to herbivory,

Fig. 13.2 (**a**) A simplified diameter distribution for a mature mid slope mixed dipterocarp forest depicting: *Shorea trapezifolia* (A – late-successional dominant canopy emergent); *Garcinia hermoni* (B – late-successional subcanopy tree); *Humboltia laurifolia* (C – late-succesional understory tree); and *Agrostistachys hookeri* (D – late-successional understory shrub). (**b**) A vertical photographic profile illustrating "Static canopy stratification" for the same mixed dipterocarp forest as illustrated by the adjacent diameter distribution. (**c**) A simplified illustration of the hypothetical change in diameter distributions for tree species that attain the canopy at early, middle, and late-stages of stand development in "dynamic canopy stratification" of a mixed dipterocarp forest (1 – stem exclusion stage 15-years; 2 – stem exclusion stage at about

pathogens, insects). Site-restricted species are usually dispersed only a limited distance from parent trees by small territorial animals (rodents) or gravity (aided by wind).

Silvicultural prescriptions for regeneration methods in this forest type must cater to all regeneration guilds to maintain floristic diversity (Ashton 1992b, 1998). However, the late-successional canopy dominants (dipterocarps in particular) form the structural complexity of the mature phase. Given their masting phenology, restricted distributions, and reliance upon advance regeneration, prescriptions are focused on emulating disturbances that promote their establishment and release. Such prescriptions need to be site specific to topographic position, soil type, and elevation to cater to their differing shade tolerances and edaphic specialization (Ediriweera et al. 2008). The core structure of the forest type is therefore an assemblage of species associations unique to different parts of the landscape – making for a complex forest mosaic (Ashton 1998). In this forest type this is the base building block upon which to develop silvicultural treatments.

13.4 The Forest Condition and Floristic Associations

13.4.1 Dynamics of Undisturbed Forest

Let us now go further into the specifics of the ecological principles described earlier for the purposes of applying sound ecologically based silviculture. Three main floristic associations for silvicultural application can be categorized for the study region and can be defined by topographic position – valley, midslope, and ridge (Ashton and Peters 1999). This is a practical simplification of the eight ecological habitats derived from aspect (2) and topographic position (4) (Gunatilleke et al. 2004, 2006).

The three floristic associations can be broadly defined by the site-restricted late-successional canopy dominant tree species. For example, valley associations are composed of *Dipterocarpus hispidus*, *Shorea megistophylla*, and *Mesua ferrea*. Midslope sites are represented by *S. trapezifolia*, *Shorea disticha*, and *Syzygium rubicundum*. The ridge association is composed of *S. worthingtonii* and *Mesua nagassarium* (Ashton and Peters 1999; Gunatilleke et al. 2006).

Fig. 13.2 (continued) 45 years; 3 – understory initiation stage at about 80 years). (**d**) Dynamic canopy stratification depicted in the vertical photographic profile of a 15-year-old mixed dipterocarp stand in the stem exclusion stage, Sinharaja forest. The photograph provides a snapshot of an early phase of stand development on a valley site with the pioneer of stand initiation *Macaranga peltata* (A) in the canopy, the pioneer of stem exclusion *Schumacheria castaneifolia* (B) in the subcanopy, and the late-successional canopy dominant *S. megistophylla* (C) in lower subcanopy. Species representative of truly below-canopy growth habits (those that comprise the different "static" below-canopy strata in a mature stand) are also present in the understory of this photograph (modified after Ashton et al. 2001a)

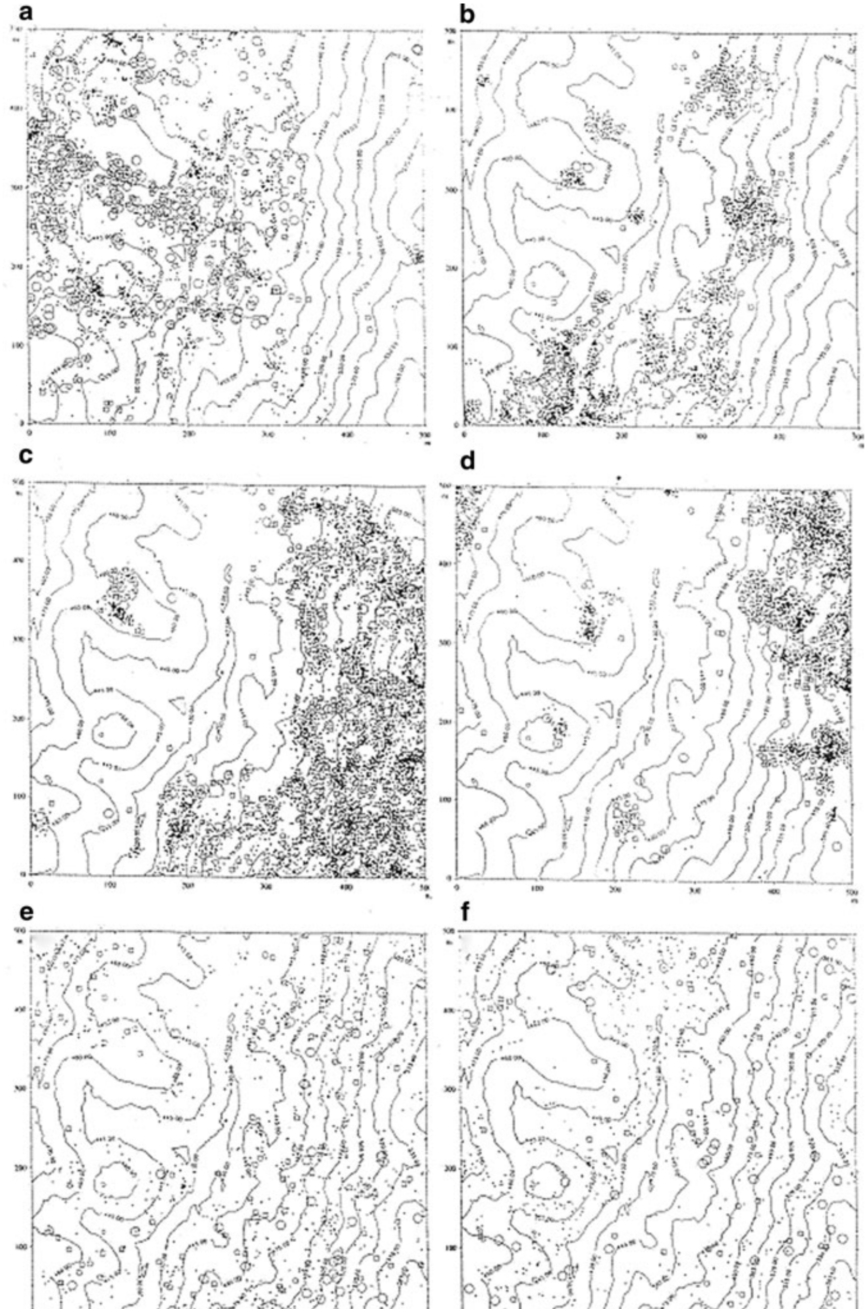

Fig. 13.3 Shows examples of site-restricted and site-generalist tree species distributions across the ridge-valley topography of the Sinharaja forest. Stem maps for each of six dominant tree species with the same 25 ha plot of mature mixed dipterocarp forest; all stems >2 cm dbh of each species have been depicted separately in each stem map. The *contour lines* over each map are at

13.4.1.1 Comparisons Between Valley, Mid Slope, and Ridge Associations

During high rain events (monsoons), the valley association endures seasonal high water tables forcing trees to have shallow rooting systems. This periodic water-logging process, in combination with downdrafts from convectional windstorms channeled through valleys, results in multiple tree falls (Ashton et al. 2001a). Rain-soaked soils from monsoonal events make the midslope association prone to earth slips and landslides.

In the valley association, soils are therefore wetter and average understory conditions have higher amounts of light with much greater degrees of variability (mean PPFD 1 mol m^{-2} day^{-1}, ± 1 SD) when compared upslope with midslope and ridge associations (mean PPFD 0.5 mol m^{-2} day^{-1}, ± 0.25 SD). This is because the valley association has the following: (1) an intermittent but tall stature canopy (55 m) that promotes greater light reflection and diffusion; (2) larger canopy gaps and the edge effects that they create; and (3) more frequent multiple wind throws especially along streams and rivers when compared with the ridge association (Ediriweera et al. 2008).

Compared with the valley association, the ridge association has soils that are shallow, more prone to desiccation, a shorter canopy tree stature (25 m), and understories that receive lower amounts of light (Ashton 1995; Ashton and Berlyn 1992; Ashton et al. 1995; Burslem et al. 2001). Because the ridge association is susceptible to water shortage, particularly during El Niño years, and also to lightning strikes (Ashton et al. 2001a, b), the gaps created by these events are smaller with trees that die standing creating minimal soil disturbance.

The valley association has the lowest density of individuals ≥ 1 cm DFH ($>2{,}000$ ha^{-1}), whereas the ridges have the highest densities ($>10{,}000$ ha^{-1}; Gunatilleke et al. 2006). Mean basal area among associations ranged from 30.5 m^2 ha^{-1} in valleys, to 60 m^2 ha^{-1} on ridges. However, the greatest number of larger diameter trees (>50 cm DBH) are on the lower slopes and valleys, but they are still relatively few in number compared with the number of canopy trees on the ridge association (Gunatilleke et al. 2006). Canopy crown densities increase and canopy crown size decrease from valley association to ridge association, driven by the water

Fig. 13.3 (continued) 15 m intervals. The *circles*, *squares* and *dots* depict the stems of a species. For this composite figure it is important to note the general distribution of stems for a given species rather than the nature of whether it is a *circle* (big tree), *square* (medium sized tree) or *dot* (sapling). Species have been divided as follows (1) four late-successional canopy tree dominant species that have restricted site distributions **(a)** *Shorea trapezifolia* – distributed across mid-slopes with a southeastern aspect; **(b)** *Shorea disticha* – distributed across mid-slopes with northwestern aspects; **(c)** *S. megistophylla* – distributed along the toes of slopes and seepages with northwestern aspects; **(d)** *S. worthingtonii* – restricted to the ridges and knolls of slopes primarily with a northwestern aspect; and (2) two late-successional nondominant canopy trees considered to have general distribution (e) *Mangifera zeylanica*; and (f) *Bhesa ceylanica* (modified after Ashton et al. 2001a) relations of the slope (Ediriweera et al. 2008). Forest canopies are therefore more compact and uniform on the ridge than in the valley (Ediriweera et al. 2008).

Faster-growing more shade intolerant canopy dominant tree species (e.g., *D. hispidus*, *S. megistophylla*, *S. trapezifolia*, *S. rubicundum*) establish more successfully in these lower-elevation valley and mid slope sites (Ashton et al. 1995; Ashton et al. 2006). The three most abundant species $\geq$1 cm dbh in the valley association are the shrub species *Psychotria nigra*, *Urophyllum ellipticum*, and *Schumacheria castaneifolia*, with densities of 240, 160, and 142 individuals ha^{-1}.

On ridge sites, the three most abundant species found are a canopy tree, *M. nagassarium*, a treelet, *Agrostistachys intramarginalis*, and an understory tree, *Humboldtia laurifolia*, which comprised exceptionally high densities of 552, 752, and 978 individual ha^{-1}, respectively (Gunatilleke et al. 2006). *Shorea worthngtonii* and *M. nagassarium* are shade-tolerant and slow-growing species adapted to regenerate in such conditions (Ashton et al. 1995, 2006).

13.4.1.2 Trends in Diversity Across Topography

In the Sinharaja region, topographic and edaphic specializations play a significant role in the spatial distribution of species. Intermediate disturbance conditions may contribute to higher species richness and diversity in the more disturbance prone valley association (78 spp. ha^{-1}) than on the more stable conditions of the ridge association (55 spp. ha^{-1}) (Connell 1978; Gunatilleke et al. 2006). Higher tree species diversity in valleys than nearby ridges have also been reported for lowland dipterocarp forests in Sumatra (Rennolls and Laumonier 2000) and Sabah (Nilus 2003).

13.4.2 Species Distributions of Economic Concern

Over 30 years of permanent plot records in stands representing all three associations have allowed us to record changes in species composition and growth from logged vs. unlogged stands (De Zoysa et al. 1991; Gunatilleke and Gunatilleke 1981, 1983, 1985; Ashton et al. 2001a).

13.4.2.1 Valley Association

Canopy dominant timber trees of the valley association comprise *D. hispidus*, *S. megistophylla*, and *S. stipularis* (used for framing, heavy construction, decking) (Ashton et al. 1997a, b). Other large canopy nondominant timber trees include *Mangifera zeylanica* and *Bhesa ceylanica*.

During mast years, the seeds of *S. megistophylla* provide an important source of flour that is made into a sweetened cake (halapa) by villagers. *S. stipularis* provides resins for temple incense and *M. zeylanica* provide an edible fruit (Ceylon mango) (Ashton et al. 1997a, b). The fruit of a subcanopy nondominant tree, *Garcinia morella*, (Goraka), is used as a spice in fish curries. In large tree fall areas, *M. peltata* (kenda) grows quickly because of the high light and freely available soil moisture – the wood is used in light construction and the leaves in wrappings (Ashton et al. 1997a, b). In addition, within disturbed areas, fast-growing vines *Coscinium fenestratum* (wenewal getta) and *Calamus zeylanicus* (rattan) are used as a medicinal and for craftwork/furniture, respectively. The stem of the medicinal is harvested and its juices used as an antiseptic to treat wounds, or it can be taken as a boiled potion for fevers and colds (Ashton et al. 2001b).

The most important function of this floristic association is riparian protection given that almost all valley sites have fast-flowing water that floods during the SW monsoon. These upper stream and river systems are the main sources of water for downstream agriculture (rice) and drinking water for almost 1/3 of Sri Lanka's population.

13.4.2.2 Midslope Association

The *S. trapezifolia–S. rubicundum* association of mid slopes is the most commercially productive zone, mainly because of the high proportion of light-hardwoods (De Rosayro 1942; Merrit and Ranatunga 1959). The midslope association can be considered the most management productive zone because of a mix of relatively fast and slow-growing timber trees, an abundance of nontimber forest products (NTFP) of high commercial value, with soils that are logistically workable, away from sensitive riparian and wetland areas and the steeper slopes of ridges. It is intermediate in site productivity (as measured by tree growth rate) between ridge (less productive) and valley sites (more productive) (Gunatilleke et al. 2006). In mature stands, nearly 90% of the merchantable volume is in the fast-growing light-hardwood species of the canopy (mostly *S. trapezifolia*). Light-hardwoods are not represented as smaller trees below-canopy unless as advance regeneration in the forest understory. Mature stands of this type have standing merchantable volumes of 60 m^3 ha^{-1} (Ashton et al. 2001a). Heavy hardwoods are the slower growing, more valuable timber species,[1] but they represent only about 10% of the merchantable volume. The most valuable timber is a canopy nondominant, *Diospyros quaesita* (a variegated ebony prized for its streaked black and yellow wood for traditional furniture), which is rare and must be supplemented by enrichment planting if desired in the new stand.

Species that provide NTFP on mid slopes increase in abundance after logging in this forest type. *C. zeylanicus* (rattan) increases in stem density from two stems ha^{-1} to over ten stems ha^{-1} after diameter-limit cutting, with stem growth rates that are 1>m $year^{-1}$ (Gamage 1998; Ashton et al. 2001b). Similarly, *Elettaria*

[1]Heavy-hardwoods comprise the slower-growing, more shade-tolerant tree species with denser wood than the light-hardwoods.

cardomomum var. *major* (wild cardamom) regenerates well in disturbed patches, by originating from the soil seed bank (Singhakumara et al. 2000). *Caryota urens* (fishtail palm), tapped for sugar by villagers, is found within the mature forest at very low densities of less than one mature individual ha^{-1} (Gunatilleke and Gunatilleke 1981, 1983; Ratnayake et al. 1988; Ashton et al. 2001b). Poor seed dispersal and sensitivity to predation make *C. urens* difficult to regenerate through natural recruitment, but after seedling establishment it can grow well in high light environments (Ratnayake et al. 1988).

13.4.2.3 Ridge Association

At the other end of the continuum, the ridge association has dense, shorter stature stands with relatively small crowns, and uniform canopies of site-restricted timber trees such as *M. nagassarium* and *S. worthingtonii*. Both are slow growing heavy hardwoods, highly prized for their timbers in temple construction and high-end house construction. Leaves that are used as roofing thatch are harvested from *A. intramarginalis* (behru), an abundant understory shrub (Ashton et al. 1997a, b, unpublished data).

13.5 Silviculture

Important groundwork on the silviculture of mixed dipterocarp forests (MDF) of South Asia started in the 1900s in the Sinharaja region of Sri Lanka (Holmes 1957), the Western Ghats of India (Troup 1921; Kadambi 1954), and the Andamans of India (Chengappa 1944). Taken together, and given the varied regional and topographic differences in shade tolerance among floristic association, all this work argued for different kinds of shelterwood techniques as the most ecologically compatible and economically viable way to regenerate these forests.

For the hill dipterocarp forest of Sri Lanka, our work suggests the same overall approach – the use of shelterwoods. But they are site specific, with the amount of parent tree and residual overstory that are left after regeneration establishment increasing to serve as an increased source of seed and shade on progressing upslope. Greatest retention is therefore on the ridges that comprise the most shade tolerant canopy tree species (*S. worthingtonii–M. nagassarium*).

13.5.1 Silviculture for the Ridge Association

On the ridges such a system would resemble the work done by Chengappa (1944) in the Andaman Islands. The best silvicultural term to define this method would be a "cyclical irregular shelterwood." To initiate the process, the subcanopy is first removed to increase understory light regimes and to release the growth of advance regeneration of the canopy tree species. Sprouting species of NTFP such as *A. intramarginalis* need no other treatment other than periodic harvest of leaves as an NTFP. Other species of NTFP or timber tree species that are density dependent with widely scattered dispersal need to be enrichment planted at this time. Care must be given to judiciously control the sprout growth of the understory and subcanopy species. After a 10-year-period about half the basal area of the canopy is removed leaving the rest to close canopy before another 50% canopy removal 30 years onward to allow the original regeneration, now pole size class, to attain the canopy (Ashton and Peters 1999). The process is repeated at year 60, and again at year 90 such that three cohorts (age-classes) of canopy trees ascend through the strata at any one time (Ashton et al. 1993; Fig. 13.4a). Based on growth data the canopy trees that move through these entries would be harvested at approximately 90–100 years of age with a dbh of 40–50 cm. Compared to the valley and mid slope sites a relatively higher basal area and stem density can be carried within the prescriptions at all times.

13.5.2 Silviculture for Valley Association

On valley sites, the canopy tree association is much more shade intolerant and a more heavy-handed approach can be taken that resembles a one-cut shelterwood (Ashton and Peters 1999; Ashton 2003). Trees left behind are designed to not be removed until the next regeneration entry. Enrichment planting of NTFP and liberation treatments follow establishment of the pioneers of initiation and stem exclusion that quickly form an umbrella canopy of light shade after a single entry cutting that removes the overstory. Release work of advance regeneration and plantings will be important for the first 10 years before leaving completely alone. No other major intrusions should be done until the start of the next rotation at about 50–60 years. It is a relatively simple system but the harvest of early (*M. peltata*, *A. scholaris*, *C. fenestratum*, *Calamus zeylanica* at 10–20 years) and mid seral (*G. morella* at 25-onwards) NTFP should be important economic additions to the management of the stand for timber (Ashton et al. 1993; Fig. 13.4b).

13.5.3 The Midslope Association: A More Detailed Silvicultural and Economic Analysis

Under shelterwood treatments on the midslope association less than a quarter of the basal area remains after logging (<12 m^2 ha^{-1}) usually comprising overstory canopy reserves of *S. trapezifolia*. Basal area rapidly increases to 20 m^2 ha^{-1}

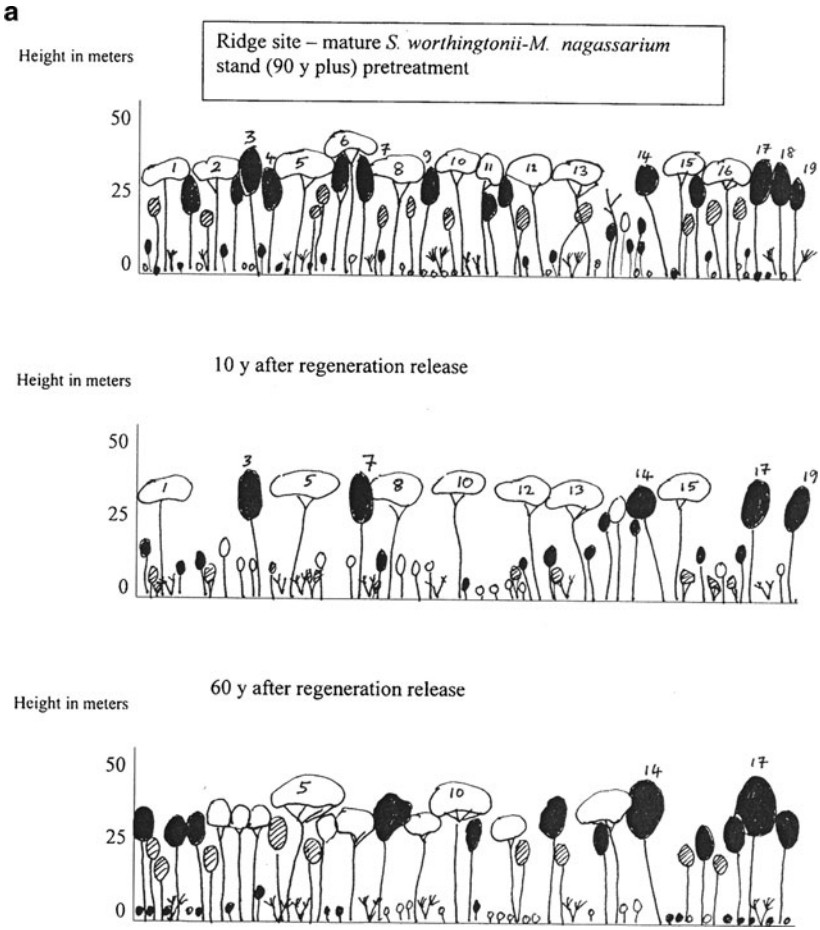

Fig. 13.4 (**a**) Profile A depicts the stylized and simplified condition of a ridge site stand prior to regeneration treatment, with *S. worthingtonii–Mesua nagassarium* trees in the canopy (represented by *open* and *darkened crowns* respectively), and their advance regeneration in the groundstory as saplings and seedlings. Subcanopy trees are represented by slanted stripes, and understory trees and shrubs by frond-like hands. Profile B depicts the stand after the subcanopy has been removed, to release growth of advance regeneration. Other species that are density dependent have been enrichment planted. Judicious control of sprout growth of the understory and subcanopy species has been completed and about half the basal area of the forest canopy has been removed. Profile C depicts the stand after two more entries each removing 50% of the canopy. Three age-classes are now present of canopy dominant trees *S worthingtoni* and *M. nagassarium* – the original canopy trees prior to start of treatments (illustrated by numbers 5, 10, 14, 17); the original regeneration that has now grown and considered 60-year-old canopy trees; and their advance regeneration waiting for the next partial canopy removal at year 90. (**b**) Profile A depicts the stylized and simplified condition of a valley site stand prior to regeneration treatment with *S. megistophylla* and *D. hispidus* trees in the canopy (represented by *open and dashed canopies*). Subcanopy trees are represented by slanted stripes and understory trees and shrubs have darken crowns. Profile B depicts the stand structure and composition10 years after overstory removal with reserve trees (1, 2, 3) judiciously left behind because of ecological or economic value and can be removed only at

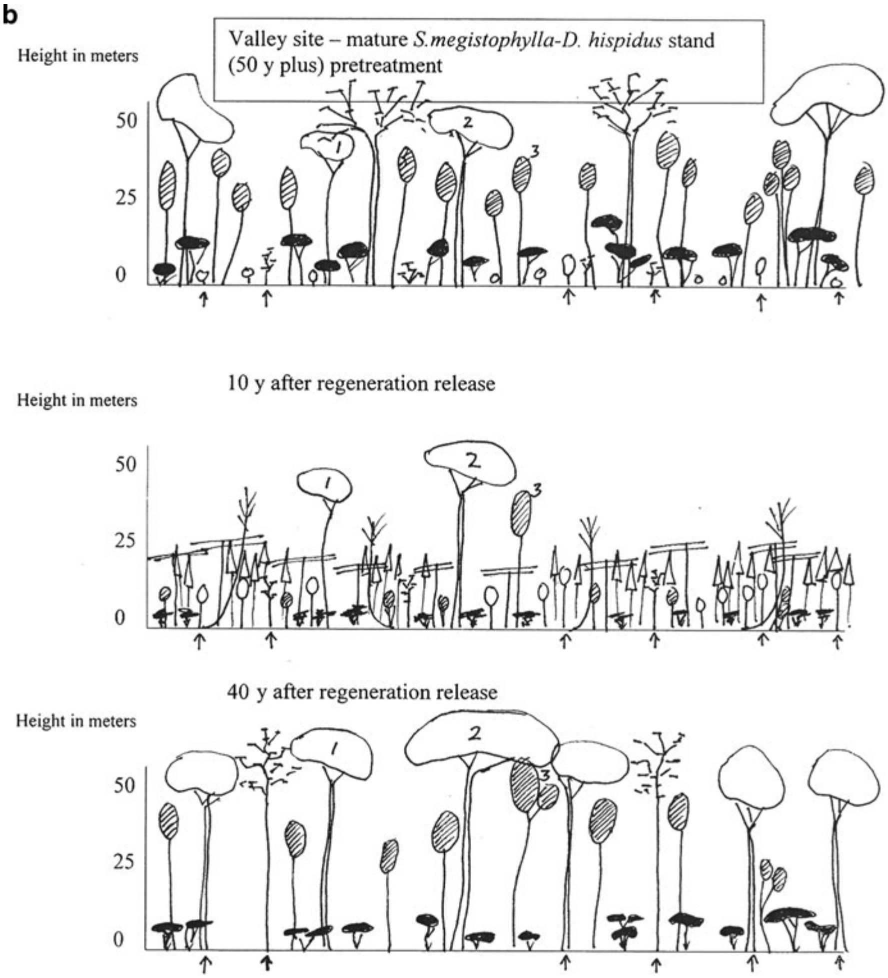

Fig. 13.4 (continued) the next regeneration entry in 50–60 years time. NTFP's (depicted by the fronds of *Calamus zeylanica* supported by the pioneers) and density dependent timber trees have been enrichment planted with follow up cleaning treatments of planted and natural regeneration beneath pioneers of initiation (depicted by *horizontal lines*) and stem exclusion (depicted by *triangles*). Profile C depicts an essentially even-aged, mixed stratified stand after 40 years of growth and development (with a few older reserves). No major intrusions have been done other than to harvest the early successional NTFP's (*Coscinium fenestratum*, *C. zeylanica* at 10–20 years), and mid seral fruits of NTFP's from subcanopy trees (*Garcinia morella* at 25 years and onwards). The pioneers have either been harvested or died from being over topped by the current canopy of 40-year-old (plus) *S. megistophylla* and *Dipterocarpus hispidus*. Subcanopy and understory trees of the same age but largely of vegetative origin fill out the strata. Understory reinitiation of advance regeneration of the canopy tree has not yet started

(1,600 stems ha^{-1}>2 cm dbh) after 10 years regrowth and 25 m^2 ha^{-1} (1,260 stems ha^{-1}>2 cm dbh) after 20 years (see Ashton et al. 1993, 2001a; Ashton and Peters 1999; Ashton 2003). During early stand regrowth, mortality from self-thinning is 1.9% year^{-1}.

Stocking on such sites is good with advance regeneration of canopy timber species normally occupying the groundstory (<1 m) in numbers ranging from 100,000 to 200,000 stems ha^{-1} (De Zoysa et al. 1991). Seldom do seedlings of *S. trapezifolia* and *S. rubicundum* species survive longer than 3 years (Ashton et al. 1995), unless released by the creation of a canopy opening. New cohorts supplement regeneration annually (Dayanandan et al. 1990).

Supplemental enrichment plantings of *D. quaesita*, *C. urens*, *Elletaria ensal*, and *C. zeylanica* can be done using containerized stock. *D. quaesita* should be planted together in groups of three since investigations have shown nearly 50% mortality after planting, and therefore group plantings are more desirable to insure full enrichment stocking. At least two cleanings are required at 3 and 6 years after planting. We estimate that at the rotation end merchantable volumes of *Diospyros* will amount to 10 m^3 ha^{-1}.

Planting of *C. urens* at 25 m spacing is an alternative that could also supplement natural regeneration. The palm is susceptible to porcupine damage and a foliar fungus, both of which have been recorded to significantly affect survival (Gamage 1998). Cleanings around the planted palm need to be done once established. However, the palm can grow well in partial openings, and matures within 15 years providing an inflorescence for tapping once a year for the next 10 years (Ratnayake et al. 1988; De Zoysa et al. 1991). Flowers can be tapped on average for 30 days yielding 3.0 kg of sugar day^{-1} tree^{-1} (Ashton et al. 2001b).

E. cardomomum var. *major*, the native cardamom, can yield over 100 fruit year^{-1} 3 years after planting (Gamage 1998; Gamage et al. unpublished data). However, the shrubs require yearly cleaning and an open environment providing periods of direct sun for best yields. This limits suitable planting sites to skid trails and landings (not more than 10% of the stand area in a well-planned timber harvest). Plants can only productively exist until canopy closure over skid trails. This occurs by year 8 in shelterwoods (De Zoysa et al. 1991; Gunatilleke et al. unpublished data). For shelterwood systems, the productive cultivation period is estimated to be a 5-year-period at the beginning of a 60 years rotation (Ashton et al. 2001b).

C. zeylanicus (rattan, cane) is a climbing palm that can grow more than a meter in a year, reaching lengths of over 20 m in 15 years after planting (Gamage 1998, unpublished data). Cane requires open conditions and a young regenerating forest stand for its best growth. It survives well after planting but its reliance for support on young timber saplings can affect their form. Judicious spacing is important.

In forest stands with extensive regeneration, the most economical silvicultural technique is to use a one-cut shelterwood harvest (leaving single and group reserves for other ecological and structural values) supplemented with enrichment plantings.

At a 4% interest rate, we estimate that these forest stands are worth about \$23,000 ha^{-1}. In forest stands without regeneration, the most economical approach is to use uniform shelterwood with single tree reserves and enrichment plantings (Ashton et al. 2001b). These forests are worth about \$20,000. This compares very favorably to the routine timber-only diameter-limit cuttings (30 year entry cycles) that are widely practised across MDF in Asia, which in our case earns a mere \$7,000. The combination of cultivating NTFP and timber combined can be comparable to tea plantations that are worth about \$26,000 ha^{-1} (Ashton et al. 2001b; Table 13.1). Given other important services that forests provide such as carbon sequestration and watershed protection, and the negative environmental values of tea cultivation (soil erosion, poor drinking water quality, pesticides, loss of biodiversity), the value difference between tea cultivation and sustainable forest management would place clear advantage with the forest management options we have described, when compared with tea (Ashton et al. 2001b; see Table 13.1).

Table 13.1 Net present values (NPV) of silvicultural regimes that (1) rely entirely on natural regeneration; (2) enrich with planting NTFPs and timber; and (3) that include ecosystem services

Regime/landuse	Diameter-limit cutting interest rate		Shelterwood treatments		Tea cultivation	
1. *Natural regeneration ONLY*	4%	6%	4%	6%	4%	6%
Initial timber values	6,097	6,097	7,525	7,525	8,000	8,000
All future timber	434	254	1,293	398	– –	– –
NTFP’s with no enrichment	842	858	1,165	820	– –	– –
Total	7,173	7,009	9,983	8,743	– –	– –
2. *With enrichment of Timber/NTFP’s*						
Fishtail palm	5,853	4,293	11,013	7,768	– –	– –
Cardamom	609	509	1,148	1,062	– –	– –
Rattan	161	101	128	87	– –	– –
Calamander	768	232	768	232	– –	– –
Intensive tea cultivation	– –	– –	– –	– –	17,735	14,937
Total	14,564	12,144	23,040	17,892	25,735	22,937
3. *With service values*						
Water quality and regulation[a]	6,532	5,234	7,653	6,738	−4,652	−3,298
Carbon credits	+ +	+ +	+	+	–	–
Biodiversity conservation	+	+	+	+	–	–
Recreation/open space	+	+	+	+	–	–
Total stacked value	21,096	17,378	30,693	24,830	21,083	13,839

[a] Forest provides strong regulatory and filter control over surface water runoff as compared tea cultivation which has poor control of surface runoff, high nutrient leachate (often from over fertilization) and sediment loss, and high amounts of pesticide contaminants in the water

The financial analysis has been done for a midslope association in a mature mixed-dipterocarp forest comparing the usual management protocol of diameter-limit cutting with the ecologically based shelterwood regeneration method. The clearance of the forest and replacement by intensive tea cultivation is compared as the alternative land use (NPV in \$USD ha^{-1}; + positive value; – negative value; ++ strongly positive value; – – strongly negative value) (modified after Ashton et al. 2001b)

13.6 Concluding Remarks About Future Directions

Sophisticated techniques to manage multiple forest products were once employed in precolonial systems for various NTFPs garnered from the forest. Products such as early-seral spices and medicinal herbs (e.g., cardamom), vines (e.g., rattan), sugar palm, and other fruit trees (e.g., durian, mangosteen) can be harvested as they mature sequentially within the growing structure of the dipterocarp forest over the course of its rotation (Ashton 2003). Such compatibility makes these systems both ecologically and economically more viable than selection systems based on diameter-limit cutting (Appanah and Weinland 1990; Ashton et al. 2001b).

In addition, the level and sophistication of stand-level delineation and prescription have yet to be incorporated into management that takes into account the unique attributes of the floristic association, topographic relief, soils and hydrology that makes silvicultural treatments stand specific. All too often for ease of management, careful integrated spatial and temporal planning that allows for judicious allocation of stands for different use values is absent. Use values such as protection (services – ecological or hydrological sensitivity) or production (timber and nontimber products) need to be carefully partitioned into dominating uses within the forest landscape. Current technologies in GIS, remote sensing, and landscape management system (LMS) models allow for this, but are seldom used in anything other than a rudimentary way.

Lastly, the majority of the remaining MDF are currently in various states of degradation making their management a large challenge before any degree of sustainability can be achieved (Ashton et al. 2001a). Most also now need rest, and in some cases substantial treatment to enrich and restore some semblance of forest structure and species composition (Adjers et al. 1995; Kuusipalo et al. 1995; Ashton et al. 1997a, b, 1998), which can provide “tomorrow’s” new products and services for the citizens of the region. Since most of the remaining forests are restricted to uplands, they therefore potentially have critical future roles as surface watershed protection for downstream drinking water supplies and agricultural irrigation; and for carbon sequestration and global and regional climate stabilization. These are two growing service values that society is now recognizing as irreplaceable.

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Chapter 14
Natural Forest Silviculture for Central African Meliaceae

Jefferson S. Hall

Abstract African Meliaceae have long been appreciated for their timber, with exports from coastal West Africa beginning over 100 years ago. Exploitation of African mahoganies of the genera *Khaya* and *Entandrophragma* has sustained much of the timber industry in Central Africa for several decades. Low levels of high-grade logging have led to the situation where Meliaceae still remain an important component of the timber harvest in the Central African subregion, particularly in more remote areas. Thus, the potential for natural forest silviculture where Meliaceae are targeted still exists. African Meliaceae are found in low densities within the subregion, with densities of between 0.5 and 2 individuals per hectare of exploitable individuals (more than 80-cm diameter at breast height, DBH) in unlogged forest being common. Typically there is an abundance of individuals within the 10-cm-DBH size class, with relatively few individuals in intermediate (20–70-cm-DBH) size classes. African mahoganies are classed as nonpioneer light demanders or species that can regenerate in the shade but need light to recruit to larger size classes. Seedlings can suffer high rates of mortality and many species distributions within the forest have been found to be related to microsite soil fertility. Because of these challenges, management of these species should be undertaken in association with management of nonpioneer light demander species from other genera and families where the exploitation intensity of mahoganies is reduced and volumes are replaced by a suite of timber species. Management for regeneration and recruitment of these species requires increasing understory light and liberating individuals belonging to the small and intermediate size classes. Highly selective logging will likely lead to long-term impoverishment of African mahogany populations where these species no longer play an important role in the timber industry.

Keywords African mahoganies · Central Africa · *Entandrophragma* · *Khaya* · Logging · Management · Meliaceae · Timber harvest

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14.1 Introduction

Timber exploitation has played an important role in the economic development and establishment of a transportation network in forested West Africa and Central Africa. The timber industry first established itself around the more easily accessible forests adjacent to port cities and in countries with favorable economic and political conditions. Thus, the West African countries of Ghana, Côte d'Ivoire, and Nigeria underwent early and sustained exploitation, with the coastal forests of Cameroon and Gabon not far behind. The export of logs from the Democratic Republic of Congo and to some degree Congo (Brazzaville) has been greatly slowed by the logistical challenges of circumnavigating the massive rapids between Stanley Pool and the port at the mouth of the Congo River. With the exception of the Atlantic coastal forests of Gabon, Cameroon, and Congo, much of the early timber exploitation was focused on the exploitation of species within the Meliaceae, particularly those of the genus of "true" African mahogany, *Khaya*, and the closely related genus *Entandrophragma*. Other genera of African Meliaceae of economic importance include *Guarea* and *Lovoa*.

High-timber-value Meliaceae are broadly distributed throughout West Africa and Central Africa. Much of West Africa's Meliaceae-rich forests have experienced intensive logging pressure and multiple entries. However, until recently, large tracts of Central Africa's forest have been subjected to high-grade logging where only the largest individuals with the best form are harvested. The distinction of level of previous logging intensity is important as silvicultural manipulations to favor the growth and development of these species for future harvests range from rehabilitating highly impoverished stands (or populations) to planning for long-term management where these species remain in significant volumes or values of annual harvests, or both. The focus of this case study is on Central African lowland forests where planned silvicultural interventions could be undertaken to favor both the long-term commercial viability of these species and the timber industry.

The species of Meliaceae that have fueled the timber industry in much of Central Africa are the African mahoganies, particularly two species of the genus *Entandrophragma*, *E. cylindricum* and *E. utile*, and *Khaya anthotheca*. *Entandrophragma angolense* and *Entandrophragma candollei* have been extracted to a lesser extent. *Lovoa trichilioides* and *Guarea* spp. are also exploited within the region but are found in much lower intensities.

14.2 Overview of General Silvicultural Characteristics

14.2.1 Density, Distribution, and Stand Structure (Size Class Distribution)

African Meliaceae are typically found in densities of 0.5–2 stems per hectare of exploitable stems for all species combined, with the highest abundance in semideciduous forests (see Hall et al. 2004). African mahoganies are more abundant in mixed-species lowland forests than in evergreen monodominant forests of *Gilbertiodendron*, a major forest type within Central Africa (CTFT 1985; Hart et al. 1989). These mixed forests are characterized by an uneven canopy where the predominant nonanthropogenic disturbance consists of single-stem to multiple-stem tree fall gaps. *Entandrophragma* spp., *K. anthotheca* and *L. trichilioides* grow to occupy the emergent stratum within the forest.

Some data exist on species of African Meliaceae in relation to resource gradients. Hall and Swaine (1981) and Hawthorne (1995) reported them to be common throughout the moist semideciduous forests of Ghana; Hawthorne (1995) reported that in relative terms *E. cylindricum* does better than congeners in drier forest types. Within the semideciduous–evergreen forest mosaic of southeastern Cameroon, southwestern Central African Republic, and northern Congo, Guillot (1981) found *K. anthotheca* to be associated with relatively high fertility status soils. Hall et al. (2004) found clear associations for *Entandrophragma* spp. with edaphic factors at the meso scale, where *E. cylindricum* was found on relatively high status base cation soils and *E. candollei* was associated with extremely low status base cation soils. Nursery trials, where seedlings were grown in forest soils of differing fertility levels, showed results consistent with a niche specialization hypothesis where the distribution of these two species is at least partially determined by soil microsite fertility within a given forest (Hall et al. 2003a). *E. angolense* showed an affinity for low-phosphorus (plant-available phosphorus) and high-pH soils. Hawthorne (1995) reported an affinity of *L. trichilioides* for acidic, base-poor soils in Ghana, a finding supported by Medjibe, Hall, Harris, and Ashton, (unpublished data) for the forests of southwestern Central African Republic. Thus, it is clear that within-stand soil heterogeneity is an important factor in determining the local distribution of trees within these species.

14.2.2 Seed Dispersal

African Meliaceae seeds are contained within fruits borne high in the canopy, where seed dispersal is wind-aided (Hawthorne 1995; Medjibe and Hall 2002; Makana and Thomas 2004). Although none of these species have been described as mast-fruiting, fruiting can be somewhat unpredictable. Forests containing *E. cylindricum* have been shown to fruit anywhere from never to twice during a given year, and different stands containing this species within the same forest can vary between copious production of seeds to virtually no seed production during the same year (Hall et al. 2003b). Plumptre (1995) found a minimum 80-cm diameter at breast height (DBH) for seed production of *Entandrophragma* spp. and 40-cm DBH for *K. anthotheca* in Uganda. In a study of five seed trees in the Central African Republic (including one individual of 73-cm DBH), Medjibe and Hall (2002) found seed dispersal of *Entandrophragma* spp. in closed forest to be related to branch direction, with most seeds falling within 20 m of the bole.

14.2.3 Germination, Establishment, and Early Seedling Growth

African Meliaceae seeds are not stored in the seed bank and generally germinate in the shade through imbibition (Synnott 1975; Hawthorne 1995; Swaine et al. 1997; Hall 2008). Although fruiting can lead to the production of a virtual seedling carpet across large areas of the forest floor (Hall et al. 2003a), mortality rates are typically very high within the first 6 months for recently germinated seedlings. In mixed forest in the Central African Republic, Hall (2008) found 12% survivorship over 190 days for *E. cylindricum* seedlings, where 55% of the mortality was due to insects. In contrast, small mammals were found to have killed 48% of the recently germinated seedlings in fallow forest within the first week in the same study and fungal attack was the highest source of mortality for seedlings in *Gilbertiodendron* forest. Hall (2008) attributed the higher incidence of fungal attack in *Gilbertiodendron* forest as compared with other forest types to conditions favoring fungi in this forest type, including decreased irradiance and higher relative humidity.

African Meliaceae seedlings can persist in the shaded understory for some time but typically require gaps to recruit to the sapling stage (Hawthorne 1995; Hall et al. 2003c). Pieters (1976) determined *E. cylindricum* seedlings grow best between 19% and 39% of full sunlight. Hall et al. (2003c) found no difference in total mass relative growth rate between 8.4 and 29.7% of full sunlight for seedlings of either *E. cylindricum* or *E. angolense* grown in a shade house experiment for 1 year in the Central African Republic. In contrast *E. utile* showed significantly higher total mass relative growth rate in the lower light level. All of these species performed significantly worse in full sunlight. Makana and Thomas (2005) found aboveground biomass for seedlings of both *E. utile* and *K. anthotheca* in forest understory plots to be less than half the value of those planted in gaps over the 18-month period of their study in the Democratic Republic of Congo (also see Swaine et al. 1997).

14.2.4 Insect Damage

As noted already, insects can be an important cause of seedling mortality for *Entandrophragma* spp. in undisturbed forest. African Meliaceae, including *Entandrophragma* spp. and *Khaya* spp., are known to be susceptible to shoot borer (*Hypsipyla robusta*) attack. In a study assessing *Hypsipyla* attack on *K. anthotheca* and *Khaya ivorensis* under different light levels in a forest in Ghana, Opuni-Frimpong et al. (2008) found significantly reduced attack under 26 and 11% canopy openness. In their study, attack on *K. anthotheca* was 11 and 0% for these two treatments, respectively. Although the authors concluded that growth under these two shade treatments was insufficient to justify use of shade as a strategy for controlling *Hypsipyla* attack in *Khaya* spp., the results may point to a strategy for controlling *Hypsipyla* attack on *Entandrophragma* spp. Indeed at photosynthetic photon flux densities of 20.36 mol $m^{-2}day^{-1}$ (26% openness) and 5.30 mol $m^{-2}day^{-1}$ (11% openness), these lower light levels span the range of optimal growth for *Entandrophragma* spp. (Pieters 1976; Hall et al. 2003c).

This may help explain why the shoot borer does not appear to be a serious problem in logging concessions in Central Africa; however, where little or no attempt is made to increase regeneration, reduced attack may also be due to low densities of saplings.

14.3 Silvicultural Interventions

Silvicultural interventions in Central Africa are constrained by both concession regulations and species' silvicultural characteristics. Efforts by the World Bank and other international donors have led to regulatory reforms whereby governments are unifying the forest management regulatory framework across the subregion (Debroux et al. 2007; Singer and Karsenty 2008). Forestry regulations have been changed to emphasize best management practices and now provide for concessions of up to 30 years. This is a big improvement as it provides companies with the incentive to manage forests for the longer-term production of timber and other benefits.

The management of African Meliaceae, in general, and African mahoganies, in particular, is nevertheless constrained by the silvicultural characteristics and size class distribution of individuals within a given forest stand. Although silviculturalists should strive to regenerate the species being harvested, a 30-year rotation cycle means that even under the best circumstances the subsequent two to three logging cycles (60–90 years) will rely on recruitment. Forest inventories for African mahoganies in unlogged forest have typically found a relative abundance of individuals in the 10-cm-DBH size class (about one per hectare), with very few individuals in subsequent larger size classes but with sufficient densities of large individuals (e.g., 80-cm DBH or more; 0.5–2 individuals per hectare) to justify entry into the stand (Hall et al. 2003b). Diameter-limit cutting regulations where large individuals (e.g., 80-cm DBH or more) are harvested provide a challenge for forest management as evidenced by a simple growth model using size class distribution data for *E. cylindricum* from Hall et al. (2003b). Even in a very well stocked unlogged forest and employing the lowest published mortality rates and highest published growth rates for this species, only 0.18 individuals per hectare would recruit to the minimum diameter cutting size class in 30 years (Fig. 14.1; also see Karsenty and Gourlet-Fleury 2006). Employing average growth and mortality rates would lead to approximately 0.06 exploitable individuals per hectare for this same period.

A constraint to understanding the potential for recruitment has been that studies tend to average growth and mortality across all size classes rather than measure sufficient individuals to provide rates independent of size class. Drawing inferences from size class distribution and crown exposure data for four *Entandrophragma* species on a 100-ha plot in the Central African Republic, Hall et al. (2003b) and Hall (2002) hypothesized that the abundance of individuals within

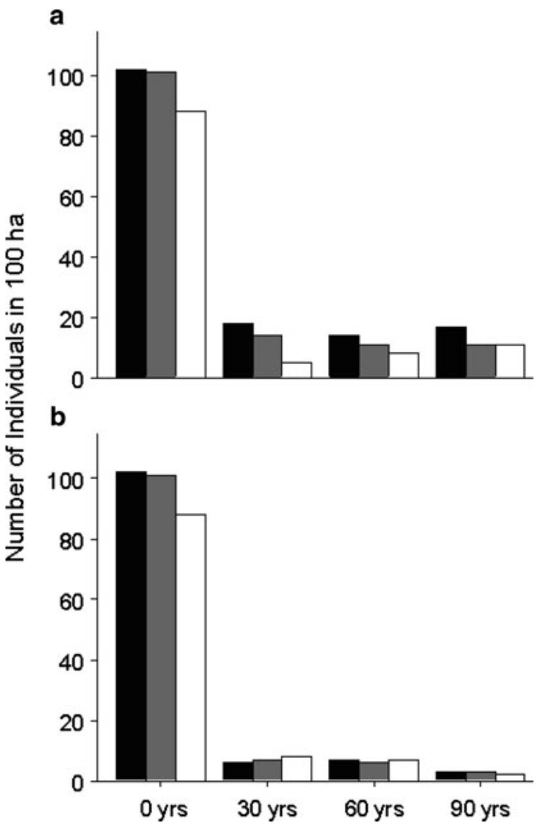

Fig. 14.1 Estimated maximum number of individual stems of *Entandrophragma cylindricum* exploitable by 30-year rotation on a 100-ha plot in the Central African Republic under three diameter-limit cutting scenarios – 80-cm diameter at breast height (DBH) (*black shading*), 90-cm DBH (*gray shading*), 100-cm DBH (*no shading*) – and where at least one seed tree is left per 10 ha in the first cutting cycle (data from Hall et al. 2003b). (**a**) The best natural conditions without silvicultural manipulations. (**b**) The average published values for growth and mortality (density can be calculated by dividing the numbers by 100). Even under the "best" scenario no trees from the 10-cm-DBH class recruit to the minimum exploitation diameter. "Best" calculated on the basis of 0.86% per year mortality and 0.64 cm year^{-1} diameter growth; "average" calculated on the basis of 1.5% per year mortality and 0.45 cm year^{-1} diameter growth (from Swaine et al. 1997; Durrieu de Madron et al. 2000)

the 10-cm-DBH size class is the result of insufficient crown exposure to light to allow for recruitment. In addition, these authors suggest that the paucity of individuals within intermediate size classes (20–60-cm DBH) is partially the result of accelerated recruitment resulting from natural release of these smaller individuals. Thus, liberation treatments where individuals within the 10-cm-DBH size class are released may be one way of improving stocking of African mahoganies in subsequent rotations.

14.4 Management Considerations

14.4.1 Best Management Practices

As with all sound forestry interventions, when designing silvicultural manipulations targeting African Meliaceae, one should follow best management practices with respect to road design and layout, reducing damage to the residual stand, reducing soil compaction, etc.

14.4.2 Soils

Recent research suggests that many African Meliaceae have some affinity to edaphic variables within the forest. Given that it is impractical to expect timber companies to complete detailed soil maps as part of their management plans, Hall (2002) proposed a "poor man's guide" to the regeneration of African mahoganies. Hall et al. (2004) found *E. cylindricum* to be associated with relatively high base cation sites within a very low soil fertility matrix. Because seedlings of this species exhibited a very positive growth response on forest soils with base cation concentrations consistent with termite mound soils (Hall 2002; Hall et al. 2003a) and trees were also found to be associated with termite mounds, they suggested targeting management interventions to favor seedling establishment on termite mounds. Data from Guillot (1981) also suggest that soil fertility should be considered when regenerating *K. anthotheca* because he found it to be among a group of species serving as an indicator of fertile soils. Thus, regeneration might be targeted for this species as for *E. cylindricum*. *E. candollei* and perhaps *L. trichilioides* regeneration might be targeted at sites with high acidity and low-percentage base saturation of the cation-exchange capacity (Hall et al. 2004; Hawthorne 1995), including within *Gilbertiodendron* stands. Alternatively, interventions might target natural regeneration or enrichment planting at the base of harvested trees.

14.4.3 Seed Dispersal

Given the fact that trees may not fruit abundantly in any given year and the relatively limited distance of dispersal in undisturbed mixed-species forest (see earlier), planning for regeneration of African mahoganies should take place up to 3 years before harvesting. Although dispersal distances in logged forest will likely exceed those of unlogged forest, it is unlikely that the generally accepted practice of leaving one seed tree per 10 ha after harvest will be sufficient to provide adequate regeneration to replace harvested individuals over time. A combination of prelogging treatments and/or leaving higher densities of seed trees (perhaps three to four per 10 ha) should be considered.

14.4.4 Light

African Meliaceae of the genera *Khaya* and *Entandrophragma* are classified as nonpioneer light demanders (Hawthorne 1995; Swaine et al. 1997). Although seedlings will germinate in deep shade, they need light to recruit. As noted earlier, *E. angolense*, *E. candollei, E. cylindricum*, and *K. anthotheca* have been shown to grow well in the light range between approximately 10 and 40% of full sunlight. Thus, gaps where one or two canopy trees are felled are optimal in terms of light for seedling recruitment of these species. *E. utile* appears to grow best at the seedling stage in sunlight levels slightly less than those for congeners, whereas both *L. trichilioides* and *Guarea* spp. appear to be more shade tolerant at this stage (Hawthorne 1995).

In forest inventories conducted throughout Ghana, Hawthorne (1995) found markedly more individuals *of Entandrophragma* spp. and *K. anthotheca* with crowns exposed to more than the average light than trees from all species throughout the forest within size classes greater than 10-cm DBH (Hall 2002; Hall et al. 2003b), indicating that crown exposure to full sunlight is essential for recruitment of these species. Therefore, silvicultural interventions targeting these species should be aimed at increasing light through liberation treatments. Individuals in these size classes of *L. trichilioides* appeared slightly more shade tolerant than those of *Entandrophragma* spp. and *K. anthotheca*, whereas *Guarea* spp. are true shade-bearers (Hawthorne 1995).

14.4.5 Species Selection and Harvest Intensity

The management of African Meliaceae should be considered within the local context of competing management objectives and constraints. Inventories in Central African Meliaceae-rich forest show that there is typically an abundance of other timber species already known to world markets within these forests (CTFT 1985; Hall et al. 2003b). More often than not these other timber species are also light-demanding through some stage of development (Hall et al. 2003b). A study comparing logging intensity in the Central African Republic found improved recruitment of timber species and projected faster overall stand recovery with increased canopy opening and silvicultural manipulation as compared with selective logging (Petrucci and Tandeau de Marsac 1994). As with Hall et al. (2003b), these authors recommend targeted interventions and basal area reduction by 33% (to approximately 20 m^2 ha^{-1}) to improve overall recruitment and forest stand recovery (Petrucci and Tandeau de Marsac 1994). Silvicultural interventions in Meliaceae-rich semideciduous forests should diversify species, reduce the reliance on African mahoganies during a given harvest, and employ techniques such as liberation treatments and targeted gap openings to improve the light environment for regeneration and recruitment. Given the clumped nature of their distribution

(Hall et al. 2004), the abundance of other timber species within the nonpioneer light demander guild associated with African mahoganies, and the potential protection from *H. robusta* attack, shelterwood systems should also be considered in forests where timber production is to be maximized.

14.5 Current Trends in Forest Management in Central Africa

Ironically, some of the current trends in protecting and managing Central African forests may produce a "perfect storm" scenario leading to long-term degradation and forest conversion. There is an increasing trend in forest certification where RIL practices are emphasized. In addition to advocating best management practices, RIL is being translated into highly selective logging where logging gaps are minimized and no further silvicultural manipulations to either enhance regeneration or improve recruitment are undertaken (J.S. Hall personal observation). Recent discussions revolving around Reducing Emissions from Deforestation and Forest Degradation as part of the international effort to control further additions of carbon dioxide into the atmosphere due to logging and forest conversion seem to support the idea of highly selective logging. At face value, this seems entirely reasonable as less cutting or damage leads to lower carbon emissions.

Managing forests for timber production requires taking the long view. Data from Hall et al. (2003b) and Petrucci and Tandeau de Marsac (1994) indicate that highly selective logging for African mahogany and other shade-intolerant species will likely lead to both stand impoverishment with respect to timber species and long-term forest degradation. There is little incentive to manage an impoverished or highly degraded forest for timber production as revenue streams will not come on line for decades. Thus, it is easy to conceive a scenario whereby timber companies abandon concessions over time. Given that Central African countries are likely to rely on their forests for economic development and revenue streams far into the future and that the amount of money generated by carbon offsets is small compared with that which can be generated from some other potential land uses, it is possible to conceive a scenario where degraded forests are converted to other land uses. One land use popular in the tropics today is the establishment of palm oil plantations using the African oil palm for the production of biofuels (Butler and Laurance 2009). A future use of a degraded forest could be biomass production for third-generation biofuels.

The above perfect storm scenario cites forces and processes that operate on different timescales. Nevertheless, because long-term management for timber production begins with the first harvest cycle, these forces must be considered prior to exploitation. Forest managers must diversify their species selection and experiment with regeneration and release treatments. Otherwise, without a massive transformation of Central African economies or without very large cash inputs from some as yet unknown collection of donors, it is hard to imagine a scenario where African Meliaceae will continue to play a significant role in natural forest management over the long term.

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Chapter 15
Managing Natural Populations of Big-Leaf Mahogany

James Grogan, Marielos Peña-Claros, and Sven Günter

Abstract Big-leaf mahogany is an emergent tree that occurs at low densities in seasonally dry forests from Mexico to Bolivia. Managing natural populations of mahogany for sustainable timber production requires matching harvest levels to population recovery rates. We describe the basic components of mahogany population dynamics observed from field studies – the distribution of stem size classes and ages from seedling to senescent adult, and mortality, growth, and reproductive rates – and silvicultural practices for reducing mortality and enhancing growth. Population structures vary predictably according to annual rainfall totals and dry season length, with important implications for management planning. For mahogany and heavily exploited high-value tropical species like it, silviculture based on thorough understanding of life history offers both management tools for ensuring future harvests and conservation tools for protecting natural populations.

Keywords Caoba · Mara · Mogno · Natural Forest Management · Population Ecology · *Swietenia macrophylla*

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15.1 Introduction

Big-leaf mahogany (*Swietenia macrophylla* King, Meliaceae) is a tree of superlatives. It is the most valuable widely traded neotropical timber species, commercially exploited over the longest period of time, and probably the most "branded" luxury wood in the world. Consequently, mahogany's conservation status has been the most controversial of all tropical timbers during recent decades. Although it is among the most studied tree species in the tropics, mahogany remains one of the most elusive to manage for timber production because natural populations tend to vanish on contact with people and markets. The issues and challenges associated with sustainable management of mahogany in natural forests are similar for many other high-value neotropical species gradually replacing it in the global timber marketplace whose distribution patterns, life histories, and regeneration syndromes resemble mahogany's, including *Cedrela odorata* L., *Tabebuia* spp., and *Hymenaea courbaril* L. (Schulze et al. 2008; Ward et al. 2008 and references therein).

In this chapter, we draw upon recent research findings and field experience in South and Central America to discuss the key issues that forest managers should consider when developing management plans for mahogany in natural forests. We consider strategies for managing both intact mahogany populations and for enhancing the recovery of previously logged populations. See Mayhew and Newton (1998) for details on managing mahogany in plantations.

15.2 Ecology and the Logging Conundrum

Mahogany is an emergent tree that occurs mainly in seasonally dry semi-evergreen tropical forests from Mexico to Bolivia, across a wide range of climatic, hydrologic, edaphic, and competitive circumstances. It is often found in groups along watercourses or in highly disturbed transition zones between forest types. Mahogany is deciduous during part of the dry season, dispersing large winged seeds by wind during the leafless period. Most seeds land within 50 m of parent trees, but longer dispersal distances (>150 m) occasionally occur. Germination follows water imbibition at the onset of the wet season; about 10–68% of seeds germinate under natural conditions depending on rainfall patterns and forest type. Seedlings and saplings are light demanding and capable of rapid height growth (up to 3 m year^{-1}) under ideal soil and light conditions; they may tolerate partial shade but cannot survive extended periods in the forest understory. This means that background densities of natural regeneration are typically very low in primary forests (Lamb 1966; Veríssimo et al. 1995; Gullison et al. 1996; Günter 2001; Grogan et al. 2002; Grogan and Galvão 2006; Lopes et al. 2008).

Mahogany requires canopy disturbance occurring shortly before or after seedling establishment for successful recruitment to adult size. While recruitment in small treefall gaps may occur under natural conditions, especially in drier forests with more open canopies, mahogany is generally thought to require large-scale disturbances such as hurricanes, fires, or floods occurring at long return intervals for successful population renewal (Gullison et al. 1996; Snook 1996; Brown et al. 2003; Grogan et al. 2003). Frequently observed post-logging regeneration failures have been attributed to the mismatch between selective logging practices creating small-scale disturbances compared with natural "catastrophic" disturbance regimes opening persistent growing space (Snook 1996).

15.3 Managing Populations: Density, Structure, and Dynamics

Mahogany typically occurs at low densities in species-rich forests. Population densities tend to be higher in Mexico and Central America than in the Amazon, where published inventories report 0.014–1.18 trees >20 cm diameter per hectare where mahogany occurs, or generally well below one commercial tree per hectare (Grogan et al. 2008). As annual rainfall totals increase and dry seasons shorten, population densities decline because taller, more closed forest canopies reduce ground-level light intensity necessary for seedling survival and growth. Under these circumstances, population structures tend to be weighted toward large stem size classes, with rare sapling and pole-sized regeneration because disturbances creating growing space at sufficient spatial scales for recruitment occur infrequently. Where rainfall totals are lower and dry seasons lengthen, mahogany population densities may increase as less dense and more disturbed forest canopies provide more light at ground level for regeneration and recruitment. However, at sites with prolonged dry season, seedling establishment may be restricted by drought effects (Fig. 15.1; Günter 2001; Brown et al. 2003; Grogan et al. 2008).

Observable population structure – the frequency distribution by size class of sub-commercial and commercial trees – is the main determinant of short-term (first and second cutting cycle) timber yields. Where large commercial trees dominate population structures, future harvests will yield only a fraction of first-harvest volumes if minimum exploitable diameter (MED) limits and commercial tree retention densities are set too low. Where size classes are more evenly distributed or weighted toward sub-commercial stems, lower MED and retention density may be possible without severely compromising future harvests. The current one-size-fits-all regulatory approach to MED and retention density prevalent in mahogany producing nations, especially in Brazil, Bolivia, and Peru, is thus ill-suited to accommodate natural variation in forest conditions and consequent mahogany population structures (Günter 2001; Grogan et al. 2008; Verwer et al. 2008).

Population dynamics shape changes in population structure through time and determine a population's response to perturbations such as large-scale natural disturbance or industrial logging. Under persistent ideal conditions in gaps where sunshine and soil water are readily available, seedlings and saplings can be expected to grow 0.5–1.5 m or more per year in height until reaching pole size (>10 cm diameter), and 0.25–1 cm year^{-1} or more in diameter after attaining the

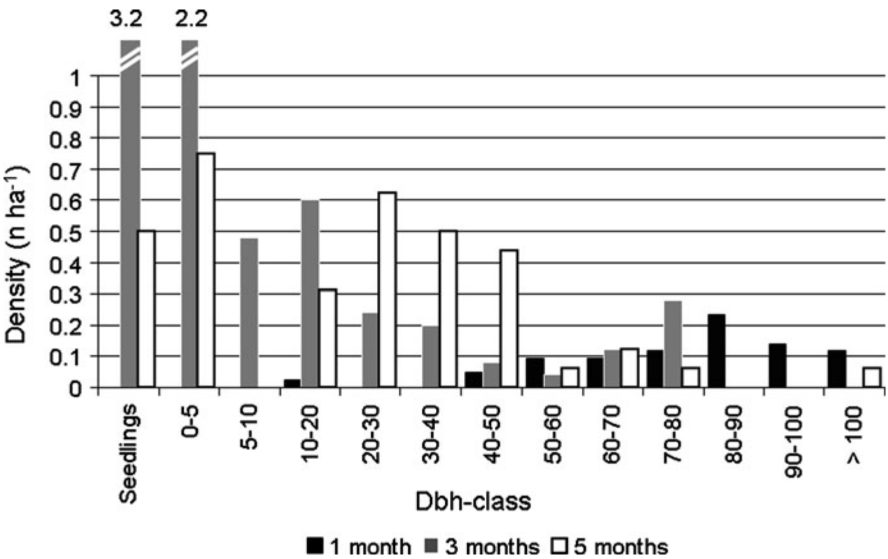

Fig. 15.1 Pre-logging population structures of mahogany in three Bolivian natural forests along a gradient of increasing dry season length (1–5 months). Site data (dry season length, annual rainfall, forest basal area, understory light as % of full sun) as follows: Mataracú (1 month, 2,693 mm, 41.5 m^2 ha^{-1}, 3.2%); La Chonta (3 months, 1,420 mm, 27.4 m^2 ha^{-1}, 6.5%); Lago Rey (5 months, 1,565 mm, 21.2 m^2 ha^{-1}, 12.0%). Low light intensities at wet sites (1 month dry season) and intensive drought (5 months dry season) provide suboptimal regeneration conditions compared to sites with intermediate light conditions and humidity (3 months dry season). Adapted from Günter (2001)

forest canopy (Günter 2001; Snook and Negreros-Castillo 2004; Grogan et al. 2005, 2008; Shono and Snook 2006; Lopes et al. 2008). Growth rates among similar-sized individual trees in a natural population may vary dramatically, depending on whether competing vines and surrounding vegetation reduce the amount of sunlight available to developing mahogany crowns, on local soil and drainage conditions, and on genetic differences. As trees grow very large, exceeding 100 cm diameter, diameter growth rates may begin to slow (Verwer et al. 2008; Grogan and Landis 2009). However, volume increment rates may continue to increase as a given linear increase in diameter leads to a much larger wood volume gain on a large tree when compared with a small one.

Mean individual volume increment ranges from 0.015 m^3 $year^{-1}$ at sites with a pronounced dry season up to 0.054 m^3 $year^{-1}$ in moist forests. Tree size and age at which volume growth culminates can vary extremely with forest type, from 41 cm diameter (corresponding to 80 years of growth) in drier forests to 102 cm diameter (more than 170 years of growth) in moist forests (Günter 2001). This variability highlights the importance of selecting suitable site conditions and suitable provenances for enrichment plantings (see Sect. 15.5).

Even under ideal conditions, mortality rates by seedlings and saplings far exceed those by sub-adult and adult trees. For trees >20 cm diameter, background mortality rates in primary forests are approximately 1% per year, and may double during the decade after selective logging due to damages and stress associated with forest structural changes caused by logging (Grogan et al. 2008; Verwer et al. 2008).

Although trees as small as 20 cm diameter may produce fruit containing approximately 50 seeds each, annual fruit production only begins as trees exceed 30–40 cm diameter depending on forest structure. Peak fecundity in moist forests occurs at large stem sizes (~130 cm diameter). A single tree may produce up to 800 fruit (Gullison et al. 1996; Snook et al. 2005; Grogan and Galvão 2006).

These vital rates are site-specific and ideally should be known for a given population or region if planning for sustainable management is to be successful (Verwer et al. 2008). The best predictors of future growth and reproductive rates by sub-commercial and retained commercial trees are growth and fruit production during the year or years previous to the first harvest, followed by the degree of crown vine coverage (Grogan and Landis 2009). With this information, and remembering that population losses due to natural mortality must also be accounted for, forest managers can estimate future timber yields based on simple simulation or modeling procedures projecting residual stand response during expected cutting cycles (usually 25–30 years).

In forests where mahogany populations have already been logged using “predatory” practices that prevailed across most of mahogany’s range until its listing on CITES Appendix II (Grogan and Barreto 2005), less than 10% of trees >60 cm diameter can be expected to survive, and trees as small as 20 cm diameter may have been harvested, depending on costs associated with log transport (Grogan et al. 2008). Under these circumstances, management must focus on population recovery through silvicultural practices aimed at enhancing growth rates and the establishment and stewardship of both natural and artificial seedling regeneration, as described below for management of natural populations.

15.4 Trees: Reducing Mortality and Enhancing Growth

Successful management of sub-commercial future crop trees (FCTs) and commercial-sized trees depends on reducing mortality and enhancing diameter (and hence merchantable volume) growth rates (Verwer et al. 2008). Residual tree mortality associated with timber extraction can be mitigated through planned harvest operations commonly known as reduced-impact logging (RIL) (Uhl et al. 1997).

Over 50% of retained seed trees and FCTs can be expected to suffer some degree of crown vine coverage, especially in semi-evergreen forests. Vines reduce stem diameter growth rates and fruit production, and may contribute directly or indirectly to 50% or more of natural tree mortality. Cutting vines, either pre- or post-logging, is the single most effective silvicultural treatment available for reducing mortality and enhancing growth and fruit production rates. Mahogany trees may respond gradually to vine cutting, requiring 4–10 years to accelerate diameter growth rates, but released trees can eventually recover above-average growth

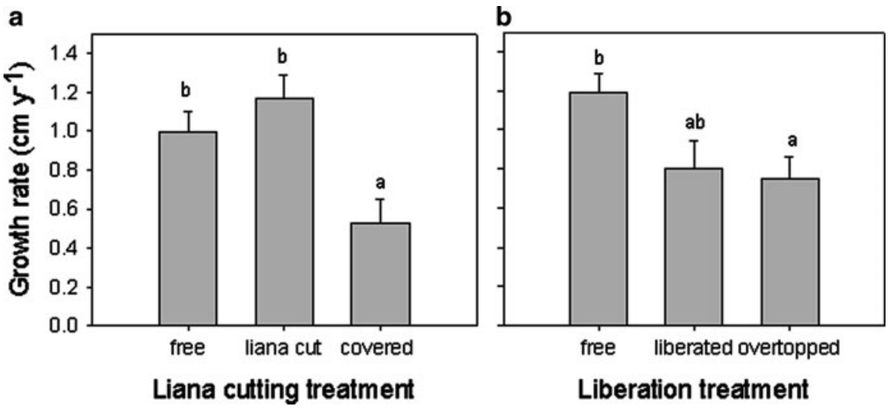

Fig. 15.2 The effect of (**a**) liana cutting and (**b**) liberation (girdling and felling neighboring trees) on growth rates of future crop trees of mahogany with 10–70 cm diameter. Data show mean values + SE; *small script letters* indicate significant difference (a < b). From Verwer et al. (2008)

capacity, and gains can be expected to persist over many years if vine re-infestation can be avoided (Fig. 15.2a; Verwer et al. 2008; Grogan and Landis 2009).

A second strategy for accelerating growth rates is liberation thinning, eliminating neighboring trees whose crowns compete with mahogany for overhead sunlight. The smaller the mahogany tree, the more likely it will face neighborhood crown competition. Crowding and overtopping trees can be girdled for gradual removal through attrition or directionally felled away from FCTs. This practice, while significantly increasing growth rates in field trials in Bolivia, does not provide as strong a performance boost as vine cutting, and tends to be more labor (and therefore cost) intensive (Fig. 15.2b; Verwer et al. 2008). Liberation thinning may be more effective in moist forests, where interspecific competition is higher, than in semi-evergreen forests.

15.5 Regeneration: Establishment and Enhancing Growth

Timber production beyond initial and second harvests can only come from trees that were seedlings when forest management efforts began. Because natural densities of mahogany seedling regeneration are very low, establishing natural regeneration where few or none exist requires the following: (1) delaying harvests until after seed dispersal by adult trees, that is, until the mid to late dry season; and (2) felling trees where possible in the direction of prevailing dry season winds, to open canopy gaps where recently dispersed seed densities are highest (Grogan and Galvão 2006). Only by establishing in logging gaps or other open areas can seedlings grow vigorously enough to recruit to pole and adult size.

Where seedlings establish at high density in the forest understory during the early wet season, survival and growth can be promoted by understory cleaning or canopy thinning. Shelterwood establishment of advance regeneration was tried with promising results in Belize in the early twentieth century, but this method is labor intensive and requires delaying overstory harvests by several years (Stevenson 1927). Clumps of vigorous advance regeneration occasionally occur near adult trees associated with recent canopy disturbances. These naturally established seedling and sapling cohorts respond to overhead canopy thinning or removal according to two general rules of thumb: (1) the larger the plant, the more vigorous the response; (2) the longer the period of suppression, the less vigorous the response. Seedlings establishing in understory shade lose their ability to respond to canopy release after 1 or 2 years of suppression (Grogan et al. 2005). Growth by natural regeneration can therefore be best enhanced by opening overhead canopy gaps as soon after establishment as possible.

Because nearly all seeds disperse to within 100 m of parent trees, seeds produced by trees retained during logging must be collected and dispersed artificially if population recovery will depend on natural regeneration. Direct seeding into log landings and logging gaps is the cheapest option but seeds experience high mortality and low establishment rates compared with outplanted nursery-grown seedlings (Gullison et al. 1996; Grogan and Galvão 2006). Experimental trials with direct seeding in artificial gaps in semi-evergreen forests in Bolivia showed a significant negative correlation of survival and light during the first year after sowing, but a positive correlation in the second year (Günter 2001).

Enrichment planting of 2- to 3-month-old nursery-grown seedlings into gaps requires fresh seeds, preferably collected from the managed population, and planting bags (10 cm diameter × 30 cm deep) filled with soil similar to soils where seedlings will be outplanted. Nursery seedlings grown under 35–50% shade tolerate outplanting into bright gap conditions with minimal mortality, and grow faster than seedlings established from direct seeding. Seedlings should be outplanted at relatively low density (5 m × 5 m spacing or more) to reduce attack rates by the mahogany shootborer (*Hypsipyla grandella*). Seedlings grow especially well when planted near stumps of tree and palm species that do not resprout, gaining a nutrient boost from decomposing roots (Grogan et al. 2003).

Log landings and logging gaps should be at least 350 m^2 at ground level in size and preferably much larger, up to 5,000 m^2 (Negreros-Castillo et al. 2003; Snook and Negreros-Castillo 2004; Lopes et al. 2008). Multiple treefall gaps in close proximity can often be emended with minimal additional tree felling to increase open space available for enrichment planting. Planting seeds or seedlings in the forest understory or even beneath thinned canopy cover will waste germplasm and labor unless overstory removal is planned within a year or two (Negreros-Castillo and Mize 1993). Avoid outplanting on the landscape where mahogany densities are low or non-existent, such as slightly elevated forest areas (higher ground on gently undulating terrain) where adults rarely occur (Grogan et al. 2003).

Competing vegetation can be cut to ground level by machete at minimal expense, increasing light availability to growing seedlings and slowing recovery by secondary vegetation. Regrowth by competing gap vegetation may quickly overtake and overwhelm outplanted mahogany seedlings. Particularly dangerous are vines, which can smother crowns and mechanically damage stems. Tending operations will thus be necessary at 1- to 2-year intervals during the first decade after outplanting, depending on site conditions. Cleaning operations should only remove vines and competing trees that directly overtop seedling and sapling crowns (Negreros-Castillo et al. 2003; Snook and Negreros-Castillo 2004; Lopes et al. 2008). Mahogany grows most rapidly in height, and escapes the shootborer at highest rates, when growing amidst "training" secondary vegetation. It can tolerate moderate shading by fast-growing short-lived pioneers such as *Cecropia* spp. and *Trema micrantha*.

15.6 The Big Picture: Implications for Management

We can be certain that mahogany and species like it face commercial extinction under current regulatory frameworks unless improved management practices address biological facts on the ground and regional ecological differences. In particular, emerging paradigms for mahogany's silviculture based on improved understanding of life history – especially population dynamics – must be reconciled with current best-practices RIL techniques aimed mainly at mitigating forest structural damages rather than enhancing timber yields (Fredericksen and Putz 2003).

We know that current legal logging intensities in Brazil, Bolivia, and Peru, ranging from 80 to 90% of commercial-sized trees, are too high for sustained yield over multiple cutting cycles. Given these retention densities, minimum exploitable diameters (MEDs) ranging from 60 to 75 cm are too low. Since these practices leave mahogany heavily overexploited in most natural forest stands, the common felling cycles of 20–30 years are far too short for sustained production. Population recovery can be accelerated through a combination of proper seed tree management and the application of silvicultural treatments as discussed in this chapter. Until these silvicultural practices are required and enforced by government regulatory agencies, management of natural mahogany populations cannot be called sustainable (Grogan et al. 2008; Verwer et al. 2008; Grogan and Landis 2009).

For these reasons, silvicultural practices reviewed here can be regarded as both management tools for ensuring future harvests, and conservation tools for protecting natural populations of this most precious and renewable resource.

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Part V
Silviculture in (Semi-)Natural Dry Forests

Chapter 16
Review
Silviculture in Seasonally Dry Tropical Forests

Todd S. Fredericksen

Abstract Because tropical dry forests have been disturbed by humans more than wetter tropical forests, there is an urgent need for conservation of the remaining tropical dry forests, and thus a large amount of research has also been directed toward their restoration. The development of cost-efficient silvicultural treatments will be critical to achieve sustained use of forest products from the remaining natural dry forests, as well as for restoration efforts. Tropical dry forests present special problems for silviculturists because prolonged dry seasons reduce growth rates and can cause significant tree mortality, especially for seedlings. Coppice systems are particularly important in the regeneration of these forests. Management of both timber and nontimber products is confined largely to managing their extraction, with relatively little application of silviculture. Although reduced-impact logging operations are increasingly applied in tropical forests, postharvest silvicultural treatments have not been integrated in the management of tropical forests, particularly in dry forests. Management in any form is often complex in areas where dry forests have been fragmented and harvested for subsistence products, such as firewood, and are affected by the damaging effects of livestock grazing and wildfire.

Keywords Forest ecology · Forest management · Reduced impact logging · Seasonally dry forest · Silviculture · Tropical dry forest · Tropical forest ecology

16.1 Introduction

Tropical dry forests occur in tropical regions with a climate characterized by several months of severe drought (Mooney et al. 1995). The Holdridge life zone classification system (Holdridge 1967) defines these forests as those receiving annual rainfall from 250 to 2,000 mm, with an annual mean temperature >17°C,

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and with a potential evapotranspiration to precipitation ratio >1 (Murphy and Lugo 1986). Tropical dry forests are tree-dominated ecosystems with a relatively continuous and closed canopy (Mooney et al. 1995), which separates them from savanna systems.

Tropical dry forests occur throughout the world, most commonly occurring in Africa, Central and South America, India, Southeast Asia, and Australia (Gerhardt and Hytteborn 1992). More than half (54%) of the remaining tropical dry forests occur in South America (Miles et al. 2006). Tropical dry forests cover more land area than tropical moist forest or rain forest (Murphy and Lugo 1986), but their original extent is probably much greater than their current distribution because of their susceptibility to conversion to agricultural lands (Murphy and Lugo 1986; Swaine 1992).

This chapter will review the silviculture of natural seasonally dry tropical forests. Silviculture refers to the theory and practice of controlling forest establishment, composition, structure, and growth (Smith et al. 1996). Silvicultural practices may be directed toward the production of timber or other forest products or for meeting conservation, restoration, recreation, or aesthetic objectives. Only existing silviculture in natural or seminatural dry forests (FAO 2000) will be addressed in this chapter. Please refer to Chap. 27 for information on planted forests. In addition, restoration and conservation of dry forests will also not be featured here, but will be addressed in other chapters (Chaps. 25, 30, 34).

As with the research on the conservation of dry forests (Sánchez-Azofeifa et al. 2005), the literature on silviculture in natural dry forests is rather limited when compared with that in other wetter tropical forests. The reasons for the scarcity of studies are not clear, but perhaps may be related to the slower growth of dry forest tree species and the smaller basal area and canopy stature of natural dry forests (Murphy and Lugo 1986), which may reduce the potential economic return of investments in silviculture. In some cases, there may be more appealing economic opportunities for planting selected fast-growing tree species, such as teak (*Tectona grandis*), on abandoned agricultural lands than for tending natural dry forests.

16.2 Dry Forest Ecology

16.2.1 Structure and Diversity

Dry forests tend to be shorter in stature and less structurally complex than wetter tropical forests. Species richness and diversity of dry forests (30–90 species) averages approximately half that of the wetter forests (50–200) (Murphy and Lugo 1986), but there can be a relatively high number of endemic species (Trejo and Dirzo 2002). Canopy height averages approximately 50% of that of wet forests. The understory tends to be better developed because of lower percentage canopy cover (Murphy and Lugo 1986). In the neotropics, there is a relatively high abundance of species in Leguminosae and Bignoniaceae in dry forests (Pennington et al. 2000). Tropical dry forests may be patchy in their distribution with respect to species composition (Jha and Singh 1990). As in other tropical forests, lianas are an important part of the structure and diversity of many dry forests (Gentry and Dodson 1987; Schnitzer and Bongers 2002). Lianas may also play an important role in water and nutrient cycling in tropical forests (Ewers et al. 1991). Lianas may have higher densities in dry forests when compared with wetter forests perhaps due to more efficient vascular systems and deeper rooting habits (Schnitzer 2005), but van der Heijden and Phillips (2008) found that liana density and basal area were not significantly related to mean annual precipitation or dry season length, at least in neotropical forests.

16.2.2 Disturbance

In addition to human impacts, natural disturbance at frequent intervals occurs in tropical dry forests through the creation of individual canopy gaps (Brokaw 1985). Larger natural disturbances also occur, such as with hurricanes in Central America and the Caribbean (Lamb 1966; Brokaw and Walker 1991; Snook 1996). In one respect, succession is a slower process in dry forests than in wetter forests because of slower growth rates of trees brought on by the strong dry season, although coppicing may accelerate recovery in some forests (Ewel 1977; Murphy and Lugo 1986; Quesada et al. 2009). Tree growth rates increase with increasing mean annual precipitation. For example, in dry forests in eastern Bolivia, tree growth rates increased from 0.08–0.35 cm to 0.24–1.34 cm along a gradient of mean annual precipitation from 1,000 to 1,700 cm (Dauber et al. 2005).

Despite a prolonged dry season with an abundance of dry fuels, it has been suggested that fire may not be an important natural disturbance agent in tropical dry forests (Murphy and Lugo 1986), although long-term influence by humans may have caused these forests to be more fire-adapted (Saha and Howe 2003). Fire is thought to provide conditions for regeneration of *Swietenia macrophylla* in Mexico (Brokaw et al. 1998; Snook 1998). Although some natural fires may occur in dry forests, if the fire frequency is high, it may negatively affect regeneration (Miles et al. 2006). Mostacedo et al. (2001) found that wildfire dramatically modified the structure and species composition of dry forests in Bolivia, but not as dramatically as in more humid forests, perhaps indicating some degree of tolerance to fire. However, Pinard and Huffman (1997) examined the characteristics in a Chiquitano dry forest and concluded that the species did not develop under a disturbance regime with frequent fire with likely changes in structure and composition with increasing fire frequencies. Schoonenberg et al. (2003) also found that high intensity fires result in higher incidences of decay in tropical dry forests trees than mechanical wounding or low intensity fires. Anthropogenic fires have long-shaped African forests and recurrent fires threaten their restoration (Swaine 1992). Human use of fire has a long history in parts of India where short fire intervals (10 years) are altering forest structure and composition (Kodandapani et al. 2008). Anthropogenic wildfires are having an increasingly large impact on seasonally dry forests in South America (Cochrane and Schulze 1999; Nepstad et al. 1999; Gould et al. 2002; Blate 2005). Control of fire and grazing promotes the passive regeneration of secondary dry forests in northwestern Costa Rica (Janzen 1988; Powers et al. 2009).

16.2.3 Water as a Limiting Factor

The seasonality of precipitation is a dominant ecological factor in tropical dry forests. It shapes the species richness, species composition, phenology, and structure of these forests (Reich and Borchert 1984; Gentry 1988; Swaine et al. 1990; Singh and Singh 1991; Bullock 1995; Swaine 1996; Eamus and Prior 2001; Poorter et al. 2004; Engelbrecht et al. 2007). Most tropical forests have seasonal variability in rainfall, but dry forests typically have one or more dry seasons totaling 2–9 months, with the length of the dry season generally increasing with distance from the equator (Murphy and Lugo 1986). Adaptations to drought include deciduousness, reduced leaf area, water storage in stems, and more efficient exploitation of soil water reserves (Reich and Borchert 1984; Borchert 1998). Most of the forest canopy becomes leafless during the dry season and foliage is restored in advance of the rainy season (Reich and Borchert 1984). Water stress is the single largest factor related to seedling mortality during the dry season (Khurana and Singh 2001). Other environmental factors that ameliorate water stress are thus important in these forests. For example, McLaren and McDonald (2003) found that shading can prevent seedling mortality because of water stress at the end of the dry season.

16.2.4 Regeneration, Growth, and Yield

Because of seasonal water stress, competition for water between tree seedlings and other vegetation can be particularly intense. Variation in soil water availability is thought to have a stronger impact on tree growth in seasonal tropical forests, while light availability is more important in wetter aseasonal forests (Baker et al. 2003). Competition from herbaceous vegetation may be more important in the relatively open understory of tropical forests compared with wetter forests, as well as within canopy gaps (Park et al. 2005; Khurana and Singh 2001; Fredericksen and Mostacedo 2000; Fredericksen et al. 2000; Teketay 1997). To survive drought, seedlings of dry forest tree species usually become deciduous and tend to have compound leaves, a higher stem dry matter content and low leaf area ratio (Poorter and Markestiejn 2008; Poorter and Kitajima 2007). Perhaps because of the intense competition for water facing tree seedlings, as well as susceptibility to mortality by seed predation, seedling herbivory, fungal disease, or fire, sprouting is a common regeneration strategy of dry forest tree species (Miller and Kauffman 1998; Mwavu and Witkowski 2008). Most tropical dry forest species have some type of dormancy and many species require some type of scarification of the seed coat to break dormancy (Khurana and Singh 2001). Seed dispersal and germination typically occurs in synchrony with the beginning of the rainy season, and wind dispersal is the primary mechanism for seed dispersal in dry forests (Bullock 1995).

Although they are often present on fertile soils relative to wetter tropical forests, dry forests are less productive because growth is limited by a long dry season (Pennington et al. 2002). Basal area and growth rates tend to decrease with decreasing mean annual precipitation and increasing dry season length (Table 16.1). For deciduous species in a Venezuelan forest, Worbes (1999) found a strong relationship between annual growth and amount of precipitation outside the dry season. Growth rates for some valuable timber species are extremely slow (e.g., *Tabebuia impetiginosa*, 0.09 cm year^{-1}) (Dauber et al. 2005). Fuelwood from coppice harvesting can be rapid. Estimated fuelwood yields in African dry forests approached 2 m^3 ha^{-1} year^{-1} or 2 t ha^{-1} year^{-1} (Chidumayo 1988a, 1990).

16.3 Dry Forest Resources

16.3.1 Uses and Users

Dry forests are less protected than wet forests (Murphy and Lugo 1986; Janzen 1988; Gerhardt 1993; Miles et al. 2006; Portillo-Quintero and Sánchez-Azofeifa 2010) perhaps because they are less diverse or because there are fewer intact dry forests worthy of protection. Nearly all of the remaining tropical dry forests in the world are threatened to some extent by human activity (Miles et al. 2006). Threats differ by region, including climate change, selective logging, and conversion to pasture in Central and South America; forest fragmentation, grazing, and fire degradation in Africa; and agricultural conversion and human population density in Asia (Cabin et al. 2002; Miles et al. 2006; Wassie et al. 2009; Espírito-Santo et al. 2009). Only about 15% of the original dry forests in India remain and are increasingly becoming degraded into savannas and grasslands (Chap. 18). The tropical dry forests of South America are perhaps the best protected, but few forests in Central America have some type of protected status (Janzen 1988; Portillo-Quintero and Sánchez-Azofeifa 2010). The dry forest regions most threatened include Madagascar, Indochina, Mexico, the Chhota-Nagpur forests of India, and the Chiquitano forests of Bolivia (Miles et al. 2006). Coping with conversion pressures on tropical dry forests include strict protection, sustainable management for timber or other resource uses such as recreation and ecotourism, and restoration of degraded stands (Sabogal 1992).

Dry forests have been exploited for their valuable timber species including teak (*T. grandis*), Sal (*Shorea robusta*), African mahogany (*Entandrophragma* spp.), big-leaf mahogany (*S. macrophylla*), and Spanish cedar (*Cedrela* spp.). The slow

Table 16.1 Comparison of growth and structure of selected seasonally dry tropical forest sites across a gradient of rainfall and dry season length

Location	Mean annual rainfall (mm)	Dry season length (mo)	Tree density (stems ha^{-1})	Canopy height (m)	Basal area (m^2 ha^{-1})	Average annual growth rate (cm)	Notes	References
Pacific Coast, Mexico	700–800	6–8	–	7–15	–	–	Most trees leafless during dry season. Transitions to thorn scrub with lower rainfall	Alvarez-Yepíz et al. (2008), Rendón-Carmona et al. (2009)
Southeastern Ghana	745–1,000	6–8	633	11	–	0.1–0.2	Few evergreen tree species	Swaine et al. (1990)
Brazilian Atlantic Forest	1,000	4	524	20–25	25	–	Mostly secondary semi-deciduous forest	Villela et al. (2006)
Chiquitanía, Bolivia	1,000–1,200	5–7	350–450	15–20	20–25	0.20–0.28[a]	Nearly all trees away from gallery forests are deciduous. Ground bromeliads in understory	Dauber et al. (2005) Chapter 19 Villegas et al. (2009)
Yucatan, Mexico	1,000–1,500	3–5	609	20–25	36	–	Many tree species leafless during dry season	Negreros-Castillo and Mize (1993), Snook (1996)
Southwestern Central African Republic	1,365	4	450	–	30.5	–	Semi-broken canopy	Hall (2008)
Pacific Coast Nicaragua	1,431	5	432–534	5–30	15–21	–	Trees are deciduous outside of gallery forest	Marín et al. (2005)
Northeastern Bolivia	1,500–1,700	4–5	300–400	20–30	20–25	0.38–0.41[a]	Many forests with very high liana densities	Dauber et al. (2005), Peña-Claros et al. (2008a, b)
Guanacaste, Costa Rica	1,575–1,717	5–6	428	15–30	30.1	–	Forests range from mostly evergreen to 80% deciduous	Powers et al. (2009), Kalacska et al. (2004)
Paragominas, Brazil	1,700–1,800	6	480	15–40	28	0.13–0.21[b]	Forests differ widely in height and liana load	Sist and Ferreira (2007), Gerwing (2001), Vidal et al. (1997)
Pará, Brazil	1,900–2,185	2–4	323	25–27	30.7	0.20–0.30[b]	Borderline seasonal forest	Parrotta et al. (2002), Silva et al. (1995)

Data are collected from control plots and not from plots where significant logging or other silvicultural treatments were performed. Unless otherwise indicated, all data are for trees $\geq$10 cm diameter-at-breast-height

[a]Includes future crop trees only

[b]Includes trees $\geq$5 cm diameter-at-breast-height

growth of many native species has also stimulated plantation forestry using exotic species, particularly teak and *Eucalyptus* species. Firewood is also an important product from dry forests, particularly in Africa (Chap. 17). More open dry forests are used for grazing and/or fodder production (Sagar et al. 2003). Exclosure systems are particularly important in Africa because of grazing and trampling by livestock (Chap. 17). A wide variety of other products are also exploited from dry forests for commercial or subsistence uses including culinary products, perfumes, medicines, resins, and building products (Bellefontaine et al. 2000).

16.4 Silvicultural Systems and Treatments

16.4.1 Harvesting

Except in the cases of some nontimber forest products, harvesting of trees is the logical starting point for a silvicultural system since the method of harvest determines the conditions for the subsequent regeneration of a stand or cutting unit. In areas with a large concentration of a valuable species, such as teak in India, some detailed silvicultural systems have been developed in natural forests (Chap. 18), which are sometimes converted into hybrids systems that rely on both natural regeneration and plantation forestry. In most natural forests in the tropics, however, harvesting has often been the only silvicultural treatment applied in tropical dry forests, and, indeed, this harvesting has often occurred without any thought of subsequent regeneration or sustainable forest management (Uhl et al. 1991; Fredericksen 1998; Sist 2000; Putz et al. 2000; Parrotta et al. 2002; Fredericksen and Putz 2003; Hall et al. 2003a; Quesada et al. 2009). It is debatable whether such selective logging could be categorized as silviculture, since it does not consider the subsequent regeneration, growth, and composition of the forest. Selective logging followed by wildfire along the frontier of agricultural development has had devastating consequences in the seasonally dry forests in the Amazon Basin (Holdsworth and Uhl 1997; Pinard et al. 1999b; Cochrane and Schulze 1999; Nepstad et al. 1999). Proactive forest fire prevention and protection have been largely neglected in tropical forest management (Gould et al. 2002; Blate 2005).

Harvesting of timber species in tropical forests is usually restricted to species with well-developed markets and, therefore, selective cutting has typically been employed. As a first step from unregulated exploitation to management, minimum cutting diameters (MCDs) have often been established for timber species in tropical forests with the idea that smaller trees would be retained for future harvest. Unfortunately, systems with MCDs have usually been subjectively defined and far too optimistic with respect to achieving sustained yield (Dawkins and Philip 1998). For example, using data from permanent sampling plots in the Chiquitano dry forests of Bolivia, Dauber et al. (2005) estimated that the second harvest (assuming a 25-year cutting cycle) would provide only 12.7% (2.40 m^3 ha^{-1}) of the volume of the initial cut (18.87 m^3 ha^{-1}) indicating that MCDs and/or cutting cycle length would have to be significantly increased to achieve sustained yield. Similarly, under the most optimistic scenario, Sist and Ferreira (2007) estimated a recovery of only 50% of commercial timber volume (only 10.5 of 21.0 m^3 ha^{-1} after 30 years) after reduced-impact logging in a seasonally dry forest in Brazil. The lack of sustainable harvesting also applies to nontimber forest products. In the Pacific Coast dry forests of Mexico, for example, (Rendón-Carmona et al. 2009) similarly found that the harvest of small trees for horticultural plant stakes was not sustainable.

As in other forests, selective harvesting in tropical dry forests tends to increase the percentage of early successional species (Villela et al. 2006; Peña-Claros et al. 2008a, b). African mahogany species may germinate in deep shade but need partial overstory release to advance beyond the seedling stage (Hall et al. 2003b). Parrotta et al. (2002) found that basal area growth increased with harvesting intensity in a seasonally dry forest in Brazil, particularly for early successional species. Intensively harvested areas were dominated by short-lived early successional species, which would be important for areas of dry forest managed for energy production. However, they also suggested that large-scale intensive harvesting may compromise long-term productivity and floristic diversity. Villela et al. (2006) found that selective harvesting in a dry Brazilian Atlantic forest reduced litter input and nutrients compared with control stands after approximately 5 years with potential negative implications for nutrient cycling. For example, input from litterfall was 154.1 and 139.6 kg ha^{-1} $year^{-1}$ for N and Ca, respectively, in logged stands compared with 189.3 and 171.4 for unlogged stands.

Another major focus of harvesting in tropical forests has been to reduce damage to the residual stand. Reduced impact logging (RIL) has been promoted both to improve sustained yield, but also to reduce incidental environmental damage, to conserve biodiversity, and as a carbon-offset method (Putz et al. 2008a, b). RIL involves such operational practices as planned layout of logging roads and skid trails, directional felling, flagging of future crop trees (FCTs), and preharvest liana cutting (Putz et al. 2008a). In tropical dry forests, RIL has perhaps received less attention, perhaps in part due to the perception of a reduced threat to soil damage in areas where much of the logging occurs during a prolonged dry season (Fredericksen and Putz 2003; Peña-Claros et al. 2008a). Still, logging damage can still be severe in seasonally dry tropical forests (Uhl et al. 1991; Verissimo et al. 1992; Johns et al.1996; Jackson et al. 2002).

In Africa and Asia, coppice systems have been developed, mostly for the harvesting of firewood. These systems include both the coppice systems using clearcutting, as well as coppice with standards. Coppice systems have generally produced good regeneration from sprouts and, in some cases, additional regeneration from seedlings (Renes 1991). In Africa, the understory tree species are often harvested for firewood on a coppice system, while the overstory species consist of timber species that are harvested less regularly (Bellefontaine et al. 2000). Coppice systems have generally produced good regeneration from sprouts and, in some cases, additional regeneration from seedlings (Renes 1991). Pruning and lopping of trees is employed by small farmers in many tropical dry forests to provide animal fodder (Bellefontaine et al. 2000).

16.4.2 Regeneration Treatments

Harvest intensity may affect the regeneration of tree species because light availability and soil disturbance increases with timber harvest intensity. However, increased light and soil disturbance may have positive or negative effects on regeneration depending on species and site conditions. Control of competing vegetation is particularly important in dry forests because of increased competition for water, but shading may also be important for small seedlings (McLaren and McDonald 2003). Having a more open canopy than other tropical forests, understory conditions in seasonally dry forests may be less impacted by logging than other forests. However, species vary greatly in ecological requirements within dry forests, which is an important consideration in the application of silvicultural treatments (Mostacedo and Fredericksen 1999; Pinard et al. 1999a; Peña-Claros et al. 2008a, b). In tropical dry forests heavily impacted by livestock, such as those in Africa, protection from livestock is critical for securing regeneration (Chap. 17).

As in other tropical forests, dry forest canopy gaps are often rapidly colonized by lianas and other competing vegetation (Dickinson et al. 2000; Fredericksen and Mostacedo 2000; Schnitzer et al. 2000; Khurana and Singh 2001; Gerwing and Uhl 2002). The dense cover of competing vegetation can result in conditions that are not favorable for regeneration of timber species because of competition for light, water, and nutrients, although shading can offset such competition in some situations (McLaren and McDonald 2003). In Ethiopia, Teketay (1997) found better regeneration of tree seedlings in the forest understory compared with gaps suggesting strong competition for water with other plants in gaps. Negreros-Castillo and Mize (1993) found that the regeneration of shade-intolerant commercial tree species increased with increasing overstory removal in a Mexican dry forest, while the regeneration of shade-tolerant species was not affected. Fredericksen and Mostacedo (2000) found a lack of regeneration of both shade-tolerant and shade-intolerant species in logging gaps in a Bolivian dry forest because of rapid development of competing vegetation. Control of competing vegetation is important because the regeneration of many commercial species is dependent to some degree on microsites with low levels of competing vegetation and/or soil scarification (Fredericksen and Putz 2003). Sist and Brown (2004) also note the need to protect advance regeneration during logging and maintain seed sources.

Silvicultural treatments that control competing vegetation or enhance regeneration have been tested in tropical dry forests including mechanical cleaning, herbicide application, prescribed fire, and mechanical scarification using logging machinery. The rationale for these treatments is based on observations of successful regeneration following the removal of competing vegetation after fires and after soil scarification by logging equipment in dry forests. Fredericksen and Licona (2000) found that without control of competing vegetation, only 31% of logging gaps in a Bolivian dry forest were likely to be refilled by commercial timber species.

Several studies have indicated an increase in regeneration of commercial species in dry forests with increasing disturbance of the canopy and soil suggesting that many of these species may require disturbance for successful regeneration, such as that provided by slash and burn agriculture, hurricanes, or fire (Brokaw et al. 1998; Snook 1998). Kennard (2002) noted an abundance of commercial species, such as *Cedrela fissilis*, *Anadenanthera colbrina*, and *Centrolobium microchaete*, in old slash and burn plots in dry forests of Bolivia and Fredericksen and Mostacedo (2000) noted increased regeneration of many Bolivian timber species by seed and by sprout in logging roads and skid trails compared with areas without soil disturbance. Fredericksen and Pariona (2002) observed significantly increased height growth of commercial seedlings, especially *Schizolobium amazonicum*, in areas of logging gaps disturbed by logging machinery compared with areas in the gap without soil disturbance. Snook (1996) and Dickinson et al. (2000) observed similar results for timber species in logging disturbed areas in Mexico, such as *S. macrophylla*, *Dendropanax arboreus*, *Bursera simaruba*, *Metopium brownei*, and *Lysiloma bahamensis*. In Bolivia, Nabe-Nielsen et al. (2007) found a significantly higher density of many light-seeded timber species, such as *Ficus glabrata* and *Terminalia oblonga*, on abandoned logging roads than in adjacent undisturbed forest. Peña-Claros et al. (2008a) found that more intensive silviculture, including more intensive harvesting, liberation of FCTs, and intentional soil scarification of logging gaps increased the density of seedlings, saplings, and pole-sized trees of shade-intolerant timber species and the growth of all commercial tree species 3 years after treatment application. However, an uncertainty with scarification treatments is the extent to which the higher recruitment of seedlings on these sites may be offset by higher mortality and/or reduced growth due to soil compaction. These results suggest that although it is important to reduce the damaging effects of logging, simply reducing disturbance may be counterproductive to regeneration of commercial tree species. Silvicultural prescriptions should be made with an understanding of the ecological conditions necessary for the regeneration of desired tree species.

Prescribed fire has been used experimentally both for fire protection and site preparation. Chidumayo (1988b, 1997) found that prescribed fire reduced the risk of more severe fires and also stimulated regeneration in coppice forest stands in a Zambian dry forest. For example, burned plots recovered 95% of prefelling density after 10 years compared with 86% for unburned plots. Kennard et al. (2002) compared high-intensity and low-intensity controlled burns in logging gaps with vegetation removal and a control treatment in a Bolivian dry forest. High and low-intensity burns reduced densities of viable seeds by 94 and 50%, respectively, but high-intensity burns led to the dominance of commercial tree species in gaps by providing good germination conditions for surviving seeds as well as the improved growth of these species by reducing competing vegetation. Heuberger et al. (2002) compared the effects of mechanical cleaning with machetes, prescribed fire, and their combination, along with a control treatment in logging gaps in a Bolivian dry forest. Although some species benefitted from the treatments, particularly in combination, there was no overall statistical increase in the density of tree seedlings or their growth following the treatments. They concluded that the high cost and potential danger of controlled burns escaping from the gaps did not justify the use of these treatments. However, the timing of treatments may be critical, since immediate establishment following gap formation may allow tree seedlings a head start over competing vegetation (Brokaw 1985; Uhl et al. 1988). Indeed, in the experiment described earlier, Heuberger et al. (2002) noted a failure of seed production during the year of their experiment, which may have accounted for the lack of successful regeneration following cleaning and burning treatments. Application of treatments should perhaps be timed to immediately precede the rainy season to minimize the effects of seed predation and establishment of competing vegetation.

Treatments directed at reducing competing vegetation in logging gaps other than through site preparation include more intensive prelogging liana cutting around trees to be harvested and individual release of advance regeneration in logging gaps. Both Gerwing and Uhl (2002) in Brazil and Alvira et al. (2004) in Bolivia found that liana cutting significantly reduced the proliferation of lianas in logging gaps, although this treatment did not necessarily keep lianas from dominating the regeneration of tree species in gaps. Release of individual seedlings in gaps with herbicide was also investigated in Bolivian seasonally dry forests (Pariona et al. 2003a). Advance regeneration of commercial species in new logging gaps was identified and competing vegetation within 1 m was sprayed with either 2,4-D or glyphosate. The growth of advance regeneration was improved, but competing vegetation rapidly encroached upon these seedlings after 1–2 years. The cost-effectiveness of the treatment was considered to be low.

Successful regeneration after disturbances in tropical dry forests is often dominated by resprouts from stumps or roots (Vieira and Scariot 2006; Mwavu and Witkowski 2008). Indeed, coppice systems are one of the most commonly employed regeneration system in Africa and India (Bellefontaine et al. 2000; Chap. 18). Sprouting is considered a viable strategy for woody plants in response to drought and other stresses (Castellani and Stubblebine 1993; Bellingham 2000; Bond and Midgley 2001). Many studies have reported abundant regeneration of trees in seasonally dry forests following logging from cut stumps or roots damaged by logging machinery (Kauffman 1991; Miller and Kauffman 1998; Kammesheidt 1999; Dickinson et al. 2000; Negreros-Castillo and Hall 2000; Mostacedo et al. 2009). Mostacedo et al. (2009) found that partially shade-tolerant and shade-tolerant species were more likely to sprout than light-demanding species. Similar findings were reported by Paciorek et al. (2000) in seasonally dry Panamanian forests. Growth rates of sprouts initially tend to grow more rapidly than seedlings (Miller and Kauffman 1998; Khurana and Singh 2001; Gould et al. 2002; Kennard et al. 2002; Kennard and Putz 2005), but trends may vary by species (Mostacedo et al. 2009). For example, Mostacedo et al. (2009) found that root sprouts of partially shade-tolerant dry forest tree species in Bolivia grew an average of 81 cm during the first year compared with 41 cm for seedlings, but root sprouts of long-lived pioneer species grew an average of 104 cm compared with 95 cm for seedlings. Coppice regeneration in tropical dry forest may reduce the dependence on seed tree retention and the often unpredictable production of seed crops in tropical forests, but long-term studies of the growth and development of trees from sprouts is needed to determine to what extent that dry forest species can be successfully established using this system. Sprouts are often subject to failure from fungal infections and mechanically weak connections with stumps or root systems, and coppice systems would likely require sprout clump thinning as an additional intermediate stand treatment. Reducing the height of residual stumps is important for ensuring mechanically stable stump–sprout connections (Bellefontaine et al. 2000).

Seed tree retention is also important for maintaining regeneration in tropical dry forests, particularly because many tree species are wind-dispersed. In Panama, Augspurger (1984) found that seeds were rarely dispersed more than 100 m from parent trees. In addition, Makana and Thomas (2004) found that dispersal strongly limited the establishment of *Entandrophragma utile* and *Khaya anthotheca* in the Congo. Logging may exacerbate seed dispersal problems by removing the largest and most vigorous reproductive individuals (Plumptre 1995; Guariguata and Pinard 1998). Seed tree retention guidelines have been commonly employed in selectively logged tropical forests, but they rarely incorporate ecological knowledge of commercial tree species (de Freitas and Pinard 2008). For example, under the Bolivian forestry law, 20% of all potentially harvestable trees must be left as seed trees, which may serve to provide seed, but also provide a reserve of volume for future harvest. Such retention of seed trees may help provide for regeneration in tropical dry forests, but their placement to areas containing microsites for regeneration is more critical than their absolute abundance and leaving a large number of such trees may be less important in dry forests where resprouting is common (Fredericksen et al. 2001).

An alternative to natural regeneration of timber species is seeding or enrichment planting in the understory or in forest openings. Enrichment planting has been attempted in many parts of the tropics, but has often resulted in failure due to poor choice of planting locations, high treatment costs, and low rates of net present value of forests originating from enrichment planting (Lamb 1969; Lamprecht 1989; Dawkins and Philip 1998; Schulze 2008). Enrichment planting in dry forests has typically been prescribed for high-value species such as big-leaf mahogany, Spanish cedar, African mahogany, and teak (Gerhardt 1996; Negreros-Castillo and Mize 2003; Snook and Negreros-Castillo 2004; Hall 2008; Chap. 18) and in forest restoration efforts (e.g., Gerhardt 1993; Sampaio et al. 2007). High light environments, such as logging gaps, or areas with reduced competing vegetation, such as skid trails or logging decks, are logical sites for artificial seeding and enrichment planting (d'Oliveira 2000), especially when following site preparation by burning or mechanical treatments (Snook and Negreros-Castillo 2004).

16.4.3 Stand Tending Treatments

Stand tending treatments are those that normally occur between harvests of forest products that promote the survival and growth of existing trees. In selectively logged forests, however, these treatments could be conducted in conjunction with harvesting to reduce the cost associated with additional stand entries. Although rarely applied, stand tending treatments that have been carried out in tropical dry forests include release of tree seedlings or liberation of larger FCTs from lianas and neighboring noncommercial trees. Unregulated high grading has resulted in the scarcity or even commercial extinction of valuable tree species in many tropical forests (Hutchinson 1988; Wadsworth 1997; Fredericksen 1998).

Increasing the growth rates and survival of FCTs can be achieved by liberating them from competition with neighboring trees and lianas (Wadsworth and Zweede 2006). Models developed from growth and yield plots in the tropical dry forests of Bolivia indicate that yield recovery rates could be increased from 13 to 16% by releasing FCTs from competing trees and lianas (Dauber et al. 2005). Growth responses of FCTs liberated from vines and competing trees increased compared with untreated trees in Bolivian Chiquitano dry forests (~0.3 and 0.2 cm year^{-1}, respectively) (Villegas et al. 2009). A similar experiment by Peña-Claros et al. (2008b) in another seasonally dry forest found that liberated FCTs grew 50–60% more than those in treatments without liberation (~0.8 and 0.4 cm year^{-1}, respectively).

Methods for liberation of FCTs from competing trees include felling, girdling, and girdling followed by herbicide application. Removal of competing trees by felling can result in damage to the residual stand and create drastic environmental changes that may be undesirable for FCTs. Girdling, which consists of the removal of cambial tissue in a complete ring around the tree stem with a hatchet or chainsaw, causes less damage and less drastic environmental change for the released trees (De Graaf 1986; Lamprecht 1989; Dawkins and Philip 1998). Wider girdling bands may increase the efficacy of treatments (Negreros-Castillo and Hall 1994) but will increase the time and cost of treatment. The use of herbicides following girdling, often called "poison girdling," may increase the efficacy of treatments (Lamprecht 1990). Pariona et al. (2003b) found that chainsaw girdling followed by application of the herbicide 2,4-D resulted in 75% crown mortality in 68–82% of treated trees in two seasonally dry Bolivian forests within 1–2 years. Girdling followed by application of glyphosate resulted in similar levels of mortality in 55–67% trees. Crown mortality of 75% or greater for trees girdled without herbicide application occurred for only 14% of treated trees. Treatment costs were relatively low, ranging from \$0.16 to 0.32 for girdling with herbicide and only \$0.09 for girdling alone. Ohlson-Kiehn et al. (2006) found that double-ring girdling followed by herbicide application was more effective than single ring girdling with herbicide, but probably not as cost-effective, except for species that are difficult to control with simple girdling. They also found that the effect of season of application was not as important in drier forests than more humid forests.

Liana control is often important for enhancing the seed production (Nabe-Nielsen et al. 2009) and growth (Stevens 1987; Gerwing 2001) of trees in dry forests. They may also cause stem deformation and tree mortality (Putz 1991). As mentioned previously, liana cutting is an important practice for reducing logging damage and minimizing the risk of injury to loggers (Johns et al. 1996; Vidal et al. 1997; Putz et al. 2008a). Liana cutting increases the availability of light to trees

(Gerwing 2001), and it may also reduce the competition for water (Pérez-Salicrup and Barker 2000).

The benefits of liana cutting to tree growth may last as long as for 4 years (Peña-Claros et al. 2008b), and Gerwing and Vidal (2002) found a 55% reduction in stem density of climbing lianas 8 years after cutting in a seasonally dry forest in Brazil. Searching for lianas only around targeted trees to release may not provide effective release because lianas in the crowns of these trees often originate from neighboring trees (Vidal et al. 1997; Alvira et al. 2004). For reducing logging damage, Vidal et al. (1997) suggested that this problem could be addressed by restricting cutting to large-diameter trees or those trees within a defined radius of trees selected for harvesting.

Gerwing (2001) tested liana cutting and controlled burning in patches forests with high liana density in seasonally dry forests in Brazil. Although burning treatments increased tree growth, sprouting of burned lianas combined with lianas from seed origin resulted in liana densities returning to 70% of pretreatment values in only 2 years. Burning treatments may also increase the vulnerability of stands to repeated burns. However, liana cutting without burning improved the growth of trees, and after 2 years, 66% of trees in treated plots carried no lianas. Sprouting of lianas after cutting can be problematic because of their rapid regrowth. Since they may sprout from both the cut ends, there is possibility for increased liana stem densities. Spraying cut liana ends with concentrated solutions of herbicides is relatively inexpensive, but had mixed results in controlling resprouting of liana species (Fredericksen 2000).

16.5 Conclusions

Tropical dry forests have long been exploited for their valuable timber species, but without much consideration of sustainable forest management. Reduced-impact logging is increasingly used to reduce the damage to residual stands in natural forests, but much less attention is paid to provisions for regeneration or intermediate stand tending treatments to increase the growth or to improve tree form or species composition. Liana cutting and enrichment planting of the most valuable commercial species are the most widely implemented treatments. Complete silvicultural systems developed to achieve sustainable forestry goals are rare. Collection of firewood, fodder, foods, and other nontimber products is often carried out on a subsistence basis without the intentional use of silviculture. Implementing planned silvicultural systems in dry forests is particularly challenging because annual tree growth is impeded by a prolonged dry season.

Silvicultural research in tropical dry forests is still mostly based on natural experiments and manipulative studies are relatively rare, particularly at the operational level. In addition, silvicultural research methods are extremely heterogeneous, making it difficult to compare the results from different studies. Cost data on treatments are rarely included in silvicultural studies in the tropics. Although these data may be difficult to obtain and problematic to interpret without a net present value analysis, the lack of any data on cost-effectiveness in most studies is a serious barrier to operational implementation of treatments.

The development of silvicultural systems in tropical dry forests will require an increased knowledge of the silvics of commercial tree species and increased emphasis on harvesting systems that not only reduce stand damage, but also stimulate regeneration. Modern tropical forest managers are often well-trained with respect to harvesting operations, but they are poorly-trained with respect to silviculture. An increased emphasis on silviculture is particularly important in dry forests because poorly-managed, degraded forests are more likely to be converted to other uses. Another challenge is promoting silviculture for the large amount of tropical forests that are not suitable for timber harvesting, but are used for subsistence products, such as firewood. These forests are often overexploited and are subjected to degradation and conversion.

Because they are often imbedded within human-populated landscapes, silviculture and conservation in dry forests may need to be developed with the participatory management of communities and indigenous groups (Sagar and Singh 2006; Walters et al. 2005), the members of which often have intimate knowledge with the forest resources. Within these landscapes, it will be important to protect dry forests from wildfire and livestock grazing. Logged forests are also more vulnerable to wildfire, and hence wildfire prevention and suppression techniques should be considered by forest managers. For tropical dry forests that have been severely degraded, forest restoration efforts will also require silvicultural knowledge. For example, natural restoration of tropical forests could be facilitated by the establishment of framework trees to attract seed dispersers (Wydhayagarn et al. 2009).

Protection of all types of dry forests will require increased ecological knowledge of these diverse ecosystems. It is particularly important that these measures be cost-effective because of the long time required to recover investments in slower-growing tropical dry forests. Integrated adaptive and sustainable management of both timber and nontimber forest products will increase the value of these forests. In addition to extraction of forest products, the potential value of conserving tropical dry forest could be enhanced by promoting their added value for carbon sequestration, biodiversity conservation, and ecotourism. Whatever the management goals, there will be a need to control the establishment, stand composition, and growth of tropical dry forests based on an enhanced understanding of their ecology.

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Chapter 17
Dry Forests of Ethiopia and Their Silviculture

Mulugeta Lemenih and Frans Bongers

Abstract An estimated 55 million ha of various kinds of dry forests exist in Ethiopia. Productivity of these dry forests is generally low, and intensive logging for timber and fuelwood has significantly altered their species composition toward the dominance of pioneers and shrubs. A number of efforts to improve dry forest management is underway, and some of these include establishment of plantations as buffer and substitute for wood supply, area exclosure to rehabilitate degraded dry forests, and introduction of participatory forest management to regulate forest access. Traditional community based forest management (TCBFM) is common in Ethiopia, and has contributed to the conservation of considerable parts of dry forests. TCBFM practices comprise various forms of traditional agroforestry systems and communal forest management using traditional institutions such as the Gada and Kobo systems. Dry forests are utilized for various purposes and by various users. Primarily, they provide frontiers for agricultural expansion and fuelwood for household energy. They are also used as natural rangelands and to provide diverse non-wood products such as gum-incenses for subsistence and cash income generation.

17.1 Introduction

Dry forests cover some 55 million ha in Ethiopia (WBISPP 2004), and are the largest vegetation resources in the country. In this chapter, the phrase "dry forests" is used in a general manner to refer to the different vegetation types found in the

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drylands[1] of Ethiopia, which comprise all vegetation types (dense and sparse) but the tropical rain forests of the southwest and the Afro-alpine and sub-Afroalpine vegetations in the central highlands. Structure and composition of the dry forests of Ethiopia are diverse, reflecting their wide distribution in diverse climatic types and over a large altitudinal range, covering from below sea level in the salt marshes of the Afar depression to the dry cool sub-afroalpine mountains. Structurally, dry forests cover the range from a high forest (closed canopy with tall trees) to desert scrubs. Compositionally, these forests are rich in endemic plant and animal species, especially the lowlands in the southeast. Perhaps one in four Ethiopia's plant species is found only in this part of the country, which is characterized by its high diversity of *Acacia* and *Commiphora* species. The latter are particularly important since about half of the 150–200 species of the genus are endemic to the small area of southeast Ethiopia, northeast Kenya, and Somalia. These dry forests are important natural endowments of drylands that have been and are still contributing to human welfare and environmental health. They provide diverse goods and services and thus play considerable ecological and socio-economic roles. This chapter provides the major attributes of Ethiopian dry forests in terms of their formation, composition, productivity, regeneration status, management, and their uses and users.

17.2 Forests and Their Structures

17.2.1 Dry Forest Types and Species Composition

Dry forests of Ethiopia are diverse and complex, and comprise vegetations from the very dry *Acacia* and *Commiphora* scrublands in the deserts of Afar and Ogaden to forests in the dry subhumid Afromontane ecosystems in the central highlands. Nine broad vegetation types are recognized in Ethiopia on the basis of climate, vegetation formation (physiognomic and habitat groupings), and associations (species composition/structure) (Demissew 1996; Anonymous 1997). Out of the nine, seven are typical of drylands and can be designated as dry forests: (1) dry evergreen Afromontane forests; (2) lowland dry forests; (3) lowland wetland (riparian) vegetations; (4) evergreen scrubs; (5) *Combretum–Terminalia* (broadleaved) deciduous woodlands; (6) *Acacia–Commiphora* (small-leaved) deciduous woodlands; and (7) Desert and semidesert scrubs. The major attributes of these vegetation types are summarized in Table 17.1.

[1]Drylands are areas characterized by seasonal climate having several months of drought (Murphy and Lugo 1986) or areas with the aridity index of ≤ 0.65 (Middleton and Thomas 1997), and this includes areas traditionally labeled as dry subhumid, semiarid, and arid.

Table 17.1 Vegetation types, distribution, species composition, and state of human disturbance of dry forests in Ethiopia

Vegetation type	Locations/distributions	Major species	Extent of human disturbance
Dry evergreen montane forest	1,900 and up to 3,400 m asl Central, eastern, south-eastern and northern highlands	*Juniperus procera*, *Podocarpus falcatus*, *Prunus africana*, *Ekebergia capensis*, *Olea* spp. and *Apodyttes dimidiate*; *Allophylus abyssinica*, *Euphorbia ampliphylla*, *Olinia rochetiana*, *Myrsine melanophloeos*, *Dovyalis abyssinica*, *Myrsine africana* and *Calpurnia aurea*	The most inhabited dry forest zone in Ethiopia, where extensive crop cultivation and grazing is widespread; Forests have significantly diminished
Lowland dry forest	450–600 m asl. This is a special type of forest in Ethiopia Only in Gambella region	*Acalpha neptunica*, *Alstonia boonei*, *Baphia abyssinica*, *Celtis gomphophylla*, *C. toka*, *Milicia excelsa*, *Mimulopsis solmsii*, *Xylopia parviflora*, *Acacia mellifera*, *Combretum* spp. *Terminalia* spp.	Little affected and the existing threats are mostly for hosting refugees, and due to dams and large scale farming
Low land wetlands (riparian) vegetation	Along the Rift Valley and major river courses crossing the lowlands such as the Awash	*Celtis africana*, *Ficus sycamorus*, *Mimusops kummel*, *Maytenus senegalensis*, spp., *Syzgium guineense*, *A. falcatus*	Significantly affected by fuelwood gathering and over grazing
Evergreen scrubs	On undulating and steep slopes of the highland plateaus (>1,500 m asl)	*Euclea schimoeri*, *Dodonaea angustifolia*, *Carissa edulis*, *Scolopia theifolia*, *Rhamnus staddo*, *Myrsine africana*, *Alpurnia aurea*, *Jasminum abyssinicum*	Are expanding at the expenses of other forest land degradation
Combretum–Terminalia (broad-leaved) deciduous woodlands	500–1,800 m asl Western, north-western and parts of south-western lowlands	*Boswellia papyrifera*, *Terminalia glaucescens*, *Grewia* spp. *Terospermum kunthianum*, *Stericulia setigera*, *Oxytenanthera abyssinica*, *Balanites aegyptiaca*, *Annona senegalensis*, *Acacia polyacantha*, *A. senegal*, *A. seyal*, *Combretum adenogonium*, *C. collinum* and *C. molle*	Human influence is growing in recent years; Fire, crop cultivation (particularly sesame) and over grazing are becoming threats to the vegetation
Acacia–Commiphora (small-leaved) deciduous woodlands	900–1,900 m asl Southern, central (Rift Valley) and eastern and south-eastern lowlands	*Acacia seyal*, *A. albida*, *A. senegal*, *A. etbaica*, *A. mellifera*, *A. drepanolobium*, *Balanites aegyptiaca*, *Commiphora africana*, *C. myrrha*, *C. fluviflor*, *C. paolii*, *C. crenulata*, *C. guidotti*, *C. erythraea*, *C. schimperi*, *C. ogadensis*, *C. rostrata*, *C. serrulata*, *C. gileadensis*, *C. hildebrandtii*, *C. cyclophylla*, *C. corrugata*, *Boswellia microphylla*, *B. ogadensis*, *B. neglecta*, *B. rivae*	Traditionally occupied by nomadic and agro-pastoralists However, those in the Rift Valley are being affected by cropland expansion, grazing, drought and unsustainable fuelwood harvest
Desert and semidesert scrubs	Below 900 m asl North-eastern and eastern (Ogaden) lowlands	The vegetation consists of deciduous shrubs, mostly *Acacia* spp., together with sparse evergreen shrubs and succulents. *Commiphora* and *Boswellia* species also exist	Grazing and refuge camping are affecting the vegetation considerably

17.2.2 Productivity and Regeneration Status

Growth and productivity of dry forests of Ethiopia is generally low, although this varies with forest types (composition), their location, climate, soil, and altitude. Most of the remnant stands of dry forests are selectively over-logged by illegal pit-sawing and fuelwood collectors. Densities of stems over 10 cm in diameter are low and range from less than 100 stems ha^{-1} in the degraded *Acacia–Commiphora* woodlands such as those in the central Rift Valley, to 1,100 stems ha^{-1} in the dry evergreen Afromontane forests (Mengesha 1996; Yebeyen 2006). Species richness reported on a hectare base also varies from 5 in the degraded *Acacia–Commiphora* woodlands to about 80 in the dry evergreen Afromontane forests. Incremental yield for the dry evergreen Afromontane forest and lowland dry forest is estimated to be 1 m^3 ha^{-1} $year^{-1}$ with the standing volume of about 100 m^3 ha^{-1} (e.g., Ameha 2002). The incremental yield for the *Acacia–Commiphora* woodlands is very low (ca. 0.0015 m^3 ha^{-1} $year^{-1}$) with the standing volume of about 6.5 m^3 ha^{-1}. The desert and semidesert scrubs and evergreen scrubs provide a much less incremental yield.

The quality of dry forests of Ethiopia is also poor. They are characterized by the presence of few good quality mature trees and dominance of deformed, pioneer, shrub, and over-aged trees. Selective illegal pit-sawing of important timber species such as *Cordia africana*, *Hagenia abyssinica*, and *Podocarpus falcatus* has degraded the quality of the dry evergreen Afromontane forests and changed their structure. For instance, in Tiro Boter Becho dry evergreen Afromontane forest, illegal pit-sawing has declined the stocking of important commercial species (e.g., *Poutria adolfi-friederici*, *C. africana*, *Syzygium guineense*, *Ekebergia capensis*), which are supposed to predominate the forest composition, to make up only 25% of the stand density (Mengesha 1996). In this forest, 75% of the stock is dominated by species that are commercially less attractive and common to disturbed forests such as *Bersema abyssinica*, *Maytenus senegalensis*, and *Teclea nobilis*. Similarly, in Munessa dry evergreen Afromontane forest, the density of *Croton macrostachyus*, a pioneer species common to disturbed sites, is more than twice the density of *P. falcatus*, a species that should have dominated the upper story of the forest (Abate 2004). Other dry forests such as the *Acacia–commiphora* woodlands are also similarly affected but from unregulated fuelwood harvest.

Dry forests of Ethiopia are characterized by poor regeneration, which is reflected (e.g., Yebeyen 2006; Tesfaye 2008) in the patterns of species-level size-class distributions: (1) species with an inverse J-shaped curve, generally indicating healthy regeneration; (2) species with high proportion of individuals in the lower (<10 cm DBH) size class but absence of individuals in the subsequent middle height classes and relatively high mature trees in the upper size classes (>100 cm DBH for the dry evergreen Afromontane forests or > 40 cm DBH for the rest of the dry forests), representing species with fluctuating patterns of regeneration due to periodic disturbances, and thus interrupted regeneration; and (3) species with bell-shaped or J-shaped distribution patterns that indicate species suffering from absence of regeneration and risk of local extermination.

Human-caused disturbances such as forest grazing, fire, timber/fire wood harvest, and farming are widespread and significant, and in most cases it negatively affects their regeneration. However, regeneration responses of plant communities and species vary considerably in response to disturbances and to environmental variables such as altitude, slope, aspect, canopy light (for highland dry forests), and edaphic conditions (e.g., Tesfaye 2008). In the same plant community, some species may show a healthy regeneration and normal inverted J-shaped population structure, while others show critical regeneration problems. For instance, in the dry evergreen Afromontane forests of Munessa (central Ethiopia), species such as *Celtis africana*, *Prunus africana*, *C. macrostachyus*, and *P. falcatus* produce reasonably large saplings (160–7,180 individuals ha^{-1}; individuals with less than 150 cm height or <10 cm DBH), while species such as *Polyscias fulva* were without saplings (Tesfaye 2008). Similarly, in the *Combretum–Terminalia* deciduous woodlands in Metema (northwest Ethiopia), species such as *Boswellia papyrifera* and *Acacia polyacantha* (Asfaw 2006) and in the *Acacia–Commiphora* deciduous woodlands in central Rift Valley species such as *A. senegal* and *A. tortilis* produce high seedling population (>500 individuals ha^{-1}), while species such as *A. etbaica* are without seedlings (Yebeyen 2006). Various disturbance regimes also influence seedling emergence and survival. For instance, in the dry evergreen Afromontane forests of Munessa, *P. adolfi-friederici*, *P. africana*, *C. africana*, and *S. guineense* lost nearly 10–70% of their seedlings due to herbivory, while *C. macrostachyus* and *A. falcatus* experience no seedling mortality under the same disturbance (Tesfaye 2008). These latter species possess tough unpalatable leaves as an adaptive defense to herbivory, contributing to low seedling mortality (Tesfaye 2008). Yebeyen (2006) found that disturbance-stimulated regeneration of *A. senegal* in the *Acacia–Commiphora* deciduous woodland – seedling density in open grazing site (620 seedlings per ha) and farmlands (600) – was much higher than in controlled grazing land (170) and in undisturbed (protected) sites (62). However, at the community level, the undisturbed site had a balanced inverse J-shaped population structure, while the disturbed sites had extra large population in the lower DBH classes of <8 cm but very few individuals in the subsequent higher DBH classes. In *Combretum–Terminalia* deciduous woodland, however, grazing suppressed seedling emergence, while fire favored it (Asfaw 2006). Burnt-and-grazed woodland sites hosted 30% less seedling density than burnt-but-ungrazed woodlands, while unburnt-but-grazed woodlands had 50% less seedlings. However, despite a good seedling emergence, species such as *B. papyrifera* showed very high (almost 100%) seedling mortality, both in burnt and in grazed woodlands, and even when protected against fire, grazing, and human disturbances (Ogbazghi et al. 2006; Negussie et al. 2008). The causes for such seedling mortality have not been identified yet.

17.2.3 Managements of Dry Forests in Ethiopia

There have been wanes and waxes in forest management practices in Ethiopia, which are closely related to the sector's institutional arrangements and political reforms in the country. Overall, compared with other sectors such as agriculture, the forestry sector suffers large neglects and has low political profile. Consequently, except for limited and intermittent conservation and management efforts, there have never been well-organized and long-term planned national scale technical management practices such as planned logging, enrichment planting, assisted regeneration, tree improvement programs, protection against fire, and maintaining/improving the health and quality of the forests. Substantial portions of dry forests of Ethiopia are managed by local communities that do exercise diverse traditional forest management practices. Generally, measures and interventions are directed at preventing further degradation (e.g., area exclosure, protected area, and plantation forests as buffer), or at regulating access to the forest and harvest of products (e.g., participatory forest management and traditional institutions for forest management), or both. Below we describe some of the prominent dry forest management measures in the country. For details of other management efforts, reference is made (e.g., Chap. 3).

17.2.3.1 Plantation Developments as Buffer to Natural Forests

In Ethiopia, reforestation and afforestation activities began over a century ago, and plantation forests now cover about 230,000 ha (Teklu 2003; Lemenih and Bongers 2010), excluding the small scale tree plantations by local people. The most planted species are the exotics *Eucalyptus* spp. (59.3% of industrially planted area) and *Cupressus lusitanica* (20.6%), followed by the indigenous *Juniperus procera* (5.7%). Plantations are confined to a narrow altitudinal range in the highlands (1,800–2,800 m asl). Productivity of the exotics is high: up to 40–55 m^3 ha^{-1} $year^{-1}$ for *Eucalyptus* on 5–10 years rotation (Örlander 1986; Kebebew 2002), and 30 m^3 ha^{-1} $year^{-1}$ for older ages (Pohjonen and Pukkala 1990). As expected, productivity varies considerably with site quality, even within species. The national scale average mean annual wood increment (MAI) of 30 m^3 ha^{-1} $year^{-1}$ for *E. globulus* is considered fair when managed on short rotation (e.g., Pohjonen and Pukkala 1990). For *Pinus patula* and *C. lusitanica,* productivity is 18–25 m^3 ha^{-1} $year^{-1}$ depending on site factors, intensity of silvicultural treatments (e.g., thinning) applied, and when managed on 20–25 years rotation. The total clear cut yield is usually between 350 and 560 m^3 ha^{-1} (Örlander 1986). *Eucalyptus* is managed almost entirely through coppice method, while other species plantations are clear felled and replanted using seedlings.

Throughout rural Ethiopia, farmers plant trees either in rows or patches as woodlots, scattered on farmlands, pasturelands, or other open areas nearby homes and farms. Farmers try to satisfy the demand of households for wood products, and the cash income is relatively large (Jagger and Ponder 2000; Mekonnen et al. 2007). Farmers prefer planting *Eucalyptus* over other species for its rapid wood production and coppice-based management (Jagger and Ponder 2000; Abebe 2005; Mekonnen et al. 2007). Their market value is thus high. In the Sidama traditional agroforestry systems (TAS), out of 116 tree species *E. camaldulensis* alone accounted for up to 61% of tree population (Abebe 2005). In woodlots, farmers manage the species usually on short rotation basis (5–10 years), and plant at densities of 10,000–25,000 trees ha^{-1} (Kebebew 2002; Abebe 2005; Mekonnen et al. 2007), four to seven times denser than stem density used in large scale industrial plantations.

Plantation forests, and especially *Eucalyptus*, are key in wood production in Ethiopia. For instance, in the Arsi highlands, 92% of rural households use *Eucalyptus* for poles, 74.7% for timber, 85% for firewood, 40% for charcoal, 93% for posts, 91% for farm implements, and 25% for traditional medicine (Mekonnen et al. 2007). In cash income, Eucalyptus is second (average 28% of household income) after crop farming, and contributes 5% of annual income to wealthy households, 20% to medium, 72% to poor households (Mekonnen et al. 2007). In this way, plantation forests help avoiding degradation of the biologically rich natural forests and woodlands. Plantation forests have a catalyzing and fostering role in rehabilitation of degraded forests and their ecosystems (Yirdaw 2002), and these and other ecosystem services are increasingly acknowledged. Plantations thus contribute to conservation of indigenous plant species. Additionally, plantation development is offering high employment opportunities, both in plantation management as in product processing industries.

17.2.3.2 Area Exclosure

Area exclosure refers to the practice of land management that involves the exclusion of livestock and humans with the aim to prevent further degradation and to stimulate restoration of the overall ecological conditions of the areas. In most cases, natural regeneration is predominant, but sometimes the area is enriched with native and/or exotic species. Soil and water conservation activities as supplementary rehabilitation efforts to foster the restoration processes are sometimes conducted as well. The technique is employed from dry woodlands to subhumid Afromontane forests in Tigray (northern Ethiopia), area exclosures cover 8% (about 400,629 ha) of the land area (Nyssen et al. 2004). Area exclosures rehabilitate vegetation, restore wild life population, and improve soil properties of degraded lands. Abebe et al. (2006) recorded 16 woody species per ha in exclosures and only 9 in nearby open grazed land. Mengistu et al. (2005) found 3,705 woody plant stems per ha in exclosures and 3,048 in nonexclosed plots. Assessment in a chronosequence of closely located exclosures (Fig. 17.1) shows an increase in species richness and density of plants and animals with exclosure age. Area exclosures improve soil chemical and physical properties, increase litter production, and reduce soil erosion (Mekuria et al. 2007). Exclosures improve long-term access of local people to wood and nonwood forest products such as firewood, fodder, and construction materials and provide greener landscapes (Nedessa et al. 2005; Yami 2005).

Community support and attitude toward exclosures were weak in the 1970s and 1980s, mostly because of the top-down approach used in establishment, but is gradually improving since. More and more local communities are engaged in exclosure establishment decision-making processes and benefit sharing (Nedessa et al. 2005).

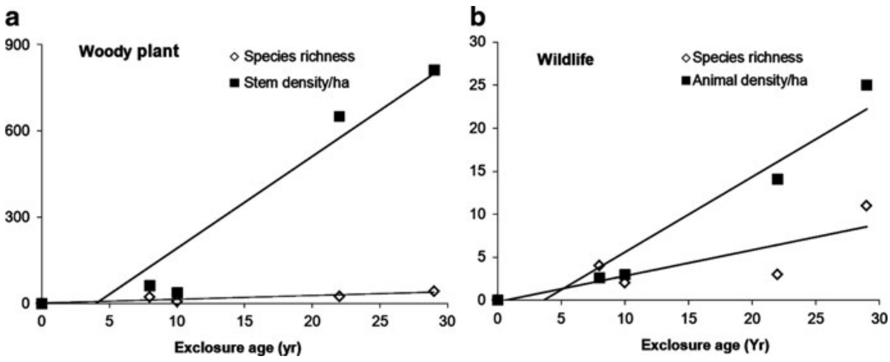

Fig. 17.1 A general trend indicating increases in woody plant (**a**) and wildlife (**b**) species richness (*triangular marker* and *broken line*) and density (*solid line with circle marker*) along a closely located chronosequence of exclosure areas (*X*-axis is in log scale; the decline in species richness and density on the 10 year exclosure compared with the 8 year might show spatial variability between the exclosure sites; data source: Yami 2005)

Still, unclear institutional organization, irregularities in management responsibility, unclear exclosure ownership, and right of benefit sharing are persistent challenges. Management plans and time frames for closure hardly exist, nor do indicators and criteria to measure successes of restoration.

17.2.3.3 Traditional Forest Management and Conservation

In many places, local people have developed a relatively advanced indigenous knowledge and well-organized indigenous institutions to manage their forest resources. Some of these traditional community based forest management (TCBFM) systems involve communal efforts such as the *Acacia–Commiphora* woodlands managed as rangelands by the Borana people with the Gada institution (Watson 2003), and the management of afromontane forests in the southwest for nontimber forest products extraction by Kobo system (Wakjira and Gole 2007). Others are smaller and private efforts involving various forms of traditional agro-forestry. The Kobo system is a forest (tree) tenure institution that grants first claimers an exclusive use right of a block of forest usually for collection of forest coffee, hanging beehives, and other NTFPs. Once claimed, the forest block is de facto individual property, respected by fellow citizens of the area, and the owner has the right to exclude others. This way, the system has resolved what could have been an open access system with threat of degradation (Wakjira and Gole 2007). The Borana Gada system imbeds different hierarchical rangeland management institutions within it. The most important part of the rangeland management institution is the obligation for animal movement, according to the patterns outlined by elders based on range availability, rangeland condition, and seasonal carrying capacity of the Borana plateau to avoid degradation (Watson 2003). This way the institution has managed the rangeland for generations.

TAS are common as well as cover large areas. The typical homegarden agroforestry system in the drylands of south and southwest Ethiopia is estimated to cover 576,000 ha (Abebe 2005). Most of the homegardens evolved from forests, where farmers maintain the upperstory trees and clear the understory vegetation to open up space for planting enset, coffee, and other food and cash crops. Abebe (2005) found 120 tree and shrub species in 144 homegardens with 83% indigenous species, an average number of tree species per farm of 20.7, an average of wood standing volume of 50.4 m^3 per farm (24 m^3 ha^{-1}), and an average density of 855 trees per farm. Stem density in the TAS varies from area to area and according to the size of land holding. In Sidama, stem density ranged from 13 to 64 per ha, with species richness from 3 to 35 per ha. Parkland agroforestry is almost the rule throughout the country (Tolera et al. 2008), and households mostly preserve selected tree species during their transformations of forest lands into agriculture land. Farmers also maintain diversity and density of woody plants in their TAS through enrichment planting using indigenous as well as exotic species, with *Eucalyptus* predominating. Some of TAS even host higher diversity of woody species than their nearby natural woodlands or forest lands (Tolera et al. 2008).

Significant forest patches are conserved and managed as sacred grooves in and around Churches, Monasteries, graveyards, Mosque compounds, and other sacred sites in several parts of Ethiopia. Particularly, the northern highlands are almost devoid of forests, and in other areas, these have been converted into farms and grazing lands leaving few patchy remnants confined mainly around Churches (Aerts et al. 2006; Wassie 2007). Wassie et al. (2009) studied 28 Orthodox Churches in Northern Ethiopia and found a total of 500.8 ha of remnant forests around them (average of 17.9 ha per church), and 160 indigenous and eight exotic woody species (100 tree species, 51 shrubs, and 17 lianas). The total number of species per Church ranges from 15 to 78. These church forests are dominated by *J. procera*, *Olea africana*, and *C. africana*. The 35,000 churches throughout Ethiopia (Wassie 2007) are likely to contribute to the conservation of considerable areas of remnant dry forests in Ethiopia. These forests are not only remnants of old-growth vegetations but also provide diverse forest products and services, and may act as sources of genetic materials for restoration of degraded dry afromontane forests. Linked through appropriate vegetation corridors they may form a unique landscape matrix for large scale landscape restoration. Recent studies on management interventions (e.g., seed sowing, seedling planting, soil scarification, exclosing) in and around these forests (Wassie et al. 2009) show promising results in this respect.

17.2.4 Uses and Users

The Ethiopian economy still largely depends on subsistence agriculture. The sector employs over 85% of the population. In traditional agriculture extensification (horizontal expansion) rules, rather than intensification, dry forests have provided much fertile cropland for millennia. When unconverted to croplands, these forests serve as natural rangeland for one of the largest livestock population (over 70 million heads) in sub-Saharan Africa (FAO 2004). Clearance for subsistence agriculture is the leading deforester of dry forests in Ethiopia (Lemenih et al. 2008; Teketay et al. 2010), causing the loss of 65,540 ha of high forest, 91,400 ha of woodlands, and 76,400 ha of shrublands annually (WBISPP 2004).

After agriculture, dry forests are mainly used for fuelwood (firewood and charcoal) supply. Biomass is the major source of energy in the country, accounting for 97% of the total domestic energy consumption, out of which woody biomass covers 78% (WBISPP 2004). Over 90% of the population is dependent on biomass fuel. The volume of fuelwood demand at national level is nearly 20-fold greater than the demand for other forest products combined. The demand has increased from 80 million m^3 $year^{-1}$ in the 1990s to 109 million m^3 $year^{-1}$ after 2000 (FAO 2005). Freely collected fuelwood from public (State) forests is used by the majority of the households, and is especially important for poor and female-headed households as they depend most on fuelwood business to generate cash income. Since fuelwood (firewood and charcoal) harvest is unregulated, the practice can be described as "forest mining."

Ethiopian dry forests, specifically those in the lowlands, are endowed with diverse *Acacia*, *Boswellia*, and *Commiphora* species, which are producing globally traded products such as gum arabic, frankincense, myrrh, and opoponax. Gum-resin production has significant socio-economic importance in the country. It provides foreign currency at the national scale, employments, and cash income to producing households. Over the last decade, average annual gum-resin production in Ethiopia was about 4,107 tons, while the export quantity stands at 2,667 tons per year. The amount of foreign currency earned from the export sale is about US$ 3.4M per year (S. Melaku unplub.). Average annual frankincense yield from *B. papyrifera* species is about 250 g per tree, with the range from 70 to 450 g, based on tree size and tapping spots (Tadesse et al. 2004). For gum arabic from *Acacia* species, average annual yield per tree is estimated to be 250 g (Yebeyen 2006).

17.3 Conclusions

The dry forests of Ethiopia, despite their important socio-economic and ecological benefits, are poorly managed and received no proper silvicultural treatments and attention. They are highly fragmented, poor in regeneration, and degraded in species composition and productivity. Efforts of dryland forest resources restoration are needed for the degraded parts and for conservation and regulated use of the existing forest. Above all, a solution is needed for the open access and unregulated harvest of the dry forest, warranting appropriate institutional arrangements that benefit both the local people and the State.

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Chapter 18
Silviculture of Dry Deciduous Forests, India

J.S. Singh and K.D. Singh

Abstract The dry deciduous forest is the most extensive forest type of India and exhibits a wide range in structural and functional attributes in response to marked spatial variation in soil and climatic conditions. Two subgroups, southern and northern, are clearly distinguished. The climax types under the southern subgroup are teak (*Tectona grandis*) forests, red sander (*Pterocarpus santalinus*) forests, and mixed forests without teak, and those in the northern subgroup are sal (*Shorea robusta*)-bearing forests, and mixed forests without sal. The most common species in the southern types are *Tectona grandis*, *Anogeissus latifolia*, *Diospyros melanoxylon*, *Boswellia serrata*, *Emblica officinalis*, *Acacia leucophloea*, *Bridelia retusa*, *Wrightia tinctoria*, *Pterocarpus marsupium,* etc. In the northern subgroup, main associates of sal are *Anogeissus latifolia*, *Buchanania lanzan*, *Terminalia tomentosa*, *Emblica officinalis,* and *Lannea coromandelica*. These forests are under-stocked and lack natural regeneration on account of excessive grazing, trampling, firewood removals and recurrent fire.

Keywords Basal area · Biomass · Biotic interferences · Coppice · Coppice rotation · Degradation stages · Dry forest · Felling cycle · Fodder trees · Grazing · Growing stock · Increment · Natural regeneration · Nontimber forest products · Nutrient conservation · *Shorea robusta* · *Tectona grandis* · Working circle

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18.1 Introduction

The dry deciduous forests in India occur in the rainfall zone between 750 and 1,500 mm with a long dry season extending over 6–8 months. The greater part of the rain falls during July to September. The maximum summer temperature may be as high as 48°C and the minimum −2.2 to 6.1°C. This type of forest occurs in Utter Pradesh, Bihar, Orissa, Punjab, Haryana, Madhya Pradesh, Gujarat, Maharashtra, Madhya Pradesh, Karnataka, and Tamil Nadu (Fig. 18.1). Dry forest once covered

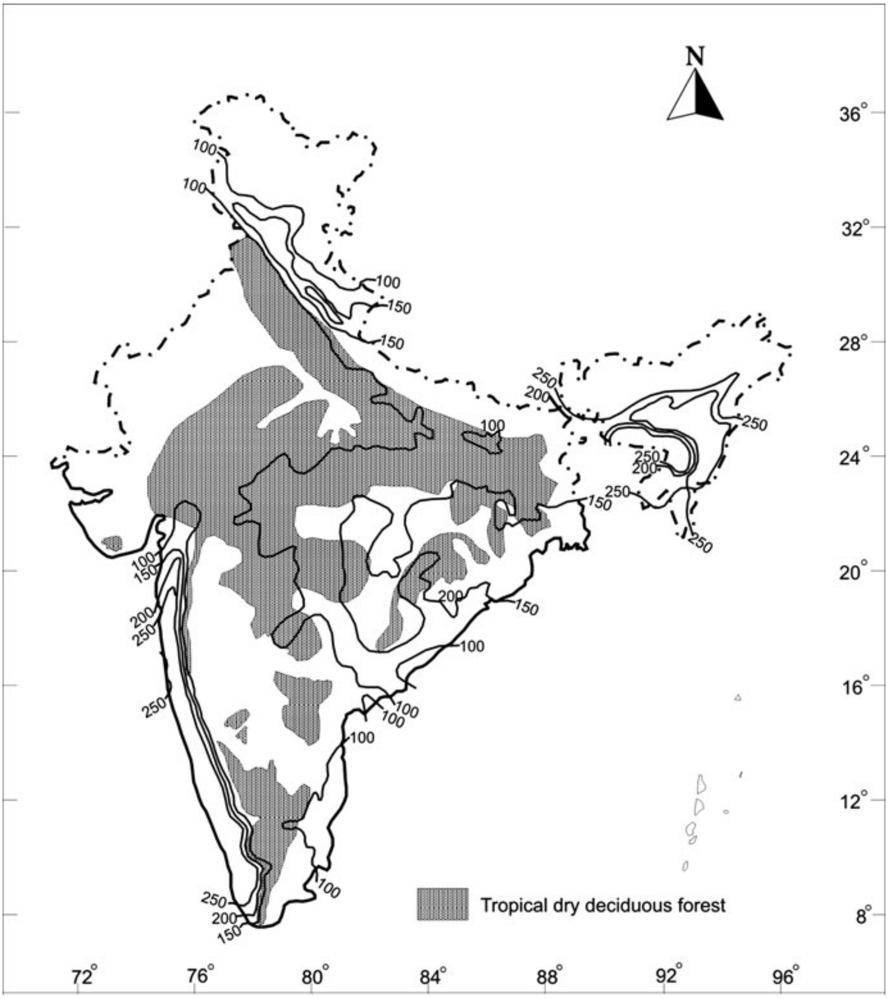

Fig. 18.1 Potential distribution of tropical dry deciduous forest in India. The *contour lines* represent annual rainfall in centimeter

more than half of the area (about 170 million ha) of the country. However, on account of population pressure, the dry forests now cover only around 26–27 million ha.

Two subgroups, southern and northern, are clearly distinguished under this broad formation (Champion and Seth 1968a). The climax types under the southern subgroup are (1) teak (*Tectona grandis*) forests, (2) red sander (*Pterocarpus santalinus*) forests, and (3) mixed forests. The most common species in the southern types are *Tectona grandis*, *Anogeissus latifolia*, *Diospyros melanoxylon*, *Boswellia serrata*, *Emblica officinalis*, *Acacia leucophloea*, *Bridelia retusa*, *Wrightia tinctoria*, *Pterocarpus marsupium,* etc. Apart from teak (*Tectona grandis*), some of the valuable species like *Santalum album* and *Pterocarpus santalinus* are found in these forests. The most common bamboo is *Dendrocalamus strictus*. The northern subgroup is divided into two types (1) sal (*Shorea robusta*)-bearing forests; and (2) mixed forests without sal. Main associates of sal are *Anogeissus latifolia*, *Buchanania lanzan, Terminalia tomentosa, Emblica officinalis* and *Lannea coromandelica.* In the mixed type, species composition is essentially the same, except that sal is absent. Several edaphic climax types and seral stages have been recognized within the two subgroups (Table 18.1) and these are described in detail by Champion and Seth (1968a).

The forests are subjected to severe biotic interferences, particularly fire and grazing. These forests supply about 80–95% of fodder needs of the livestock

Table 18.1 Tropical dry deciduous forest types of India (based on Champion and Seth 1968a)

Southern tropical dry deciduous forests		Northern tropical dry deciduous forests
Dry teak bearing forest		Dry sal-bearing forest
Dry teak forest		Dry Siwalik sal forest
Very dry teak forest		Dry plains sal forest
Dry red sanders bearing forest		Dry peninsular sal forest
Southern dry mixed deciduous forest		Northern dry mixed deciduous forest
Degradation stages	**Edaphic types**	**Seral types**
Dry deciduous scrub	Anogeissus pendula forest	Dry tropical riverain forest
Dry savannah forest	Anogeissus pendula scrub	Khair (Acacia catechu)-sissoo (Dalbergia sissoo) forest
Dry grassland	Boswellia forest	Inundation babul (Acacia nilotica) forest
Euphorbia scrub	Babul (Acacia nilotica) forest	Secondary dry deciduous forest
	Hardwickia forest	
	Butea forest	
	Aegle forest	
	Laterite thorn forest	
	Saline/alkaline scrub savannah	
	Phoenix savannah	
	Babul (Acacia nilotica) savannah	
	Salvadora-Tamarix scrub	
	Dry bamboo brake	

Degradation stages, and edaphic and seral types occur in both southern and northern areas and 81–100% of fuelwood needs (Singh and Singh 1992a). A total of 76–89% of herbage is consumed by animals through free-range grazing (Pandey and Singh 1992). Productivity is low and far exceeded by removals, causing degradation. Degradation stages of dry deciduous forest can be compared with Savanna woodland (see Table 18.1). Here vegetation consists of broken soil cover, trees 3–6 m high including some trees having many stems from the base. The grasses occur throughout. Fire-resistant trees persist, either slowly establishing themselves as trees or sending up annual shoots from woody root stock. Thorn forests where Acacias predominate can be compared with "Tree savanna." Here locally consociations of certain species are predominant, notably *Acacia planiforns*, *Acacia Senegal,* and *Prosopis spicigera*. Thorn forests on degradation due to biotic factors resemble "Scrub savanna." *Zizyphus* scrub is common throughout intermingled with Euphorbias which are more conspicuous in rocky areas due to intense biotic pressure. On semiarid or arid saline/alkaline soils, *Salvadora* scrub is met with. At places, *Cassia auriculata,* which coppices very well, is left as residual vegetation almost in pure patches.

Small leaf size, medium leaf texture, rough bark texture, and medium deciduousness characterize the dry deciduous forest vegetation (Sagar and Singh 2003). These forests occur on nutrient-poor soils and show several strategies of nutrient conservation, e.g., nutrient immobilization by litter microbes (Singh et al. 1989), and substantial withdrawal of nutrients from the senescing leaves (sufficient to meet the demand of next generation of leaves) (Lal et al. 2001a, b) which enable them to cope with nutrient poverty. The canopy is renovated toward the end of the dry hot period enabling the plants to take full advantage of the ensuing short rainy season for biomass generation and growth (Singh and Singh 1992b). Because of the great variety of community types, soil and climatic conditions within the dry forest zone, these forests exhibit a wide range in structural and functional attributes. Basal area (trees + shrubs) ranged between 9 and 15 m^2 ha^{-1}, total biomass (trees + shrubs + herbs) from 53 to 94 ton ha^{-1}, and total net production (trees + shrubs + herbs) from 11 to 19 ton ha^{-1} $year^{-1}$ among three sites in an area experiencing 800 mm annual rainfall (Singh and Singh 1991).

Production of fuelwood, small timber, and other nontimber forest products (NTFPs) is the main objective of management and these forests primarily cater to the local demand. Firewood, bamboo, tendu (*Diospyros melanoxylon*) leaves, sandal (*Santalum album*) wood, red sanders, fruits and seeds, medicinal plants, and spices, etc. are important products. According to one estimate, NTFPs from these forests contribute 77% of the net value of forests, with the net present value of revenues from NTFPs being US $1,016–1,348 ha^{-1}, significantly higher than the returns from alternate land uses (Mahapatra and Tewari 2005). If use, option, and existence value are all taken into account, the total present value of nontimber goods and services available from a tropical deciduous forest in India varies from a minimum of US $4,034 to a maximum of US $6,662 ha^{-1} (Chopra 1993).

18.2 Silvicultural Systems

Coppice system is the most prevalent management tool for the dry deciduous forests of India (FAO 1989). The coppice rotation varies between 30 and 40 years. Simple coppice (complete felling followed by natural and coppice regeneration), coppice with standards (complete felling retaining specified number of standards to provide seedling crop and to meet larger size timber demand), and coppice with reserves (retaining individuals of specified species and diameter limits, mainly saplings and poles, felling the rest) are the usual practices. In some states, suitable areas are also worked under clear felling system with artificial plantings quite often by teak (*Tectona grandis*). Sal (*Shorea robusta*) and teak have reliable yield functions (tables), available with the forest department, to guide site classification, stocking level, thinning and harvesting operations. Best management practice results in forest stocking and growth in accordance with the yield tables. When the same forests are utilized for producing more than one product, incompatibilities arise and operations intended to benefit one become detrimental to the other. For example, burning which is practiced to induce production of coppice shoots in tendu (*Diospyros melanoxylon*) increases the availability of tendu leaves (an important source of revenue to forest departments) but adversely affects natural regeneration of other species. Similarly collection of seeds adversely affects regeneration of sal, especially during poor seed years (Verma and Sharma 1978). Most of the degraded areas are restocked artificially.

18.3 Teak-Bearing Forest

Economically, teak is the most valuable timber species. It occurs in the dry forest but grows best in areas having 1,250–3,750 mm annual rainfall (see Seth and Kaul 1978 for review). Teak begins flowering and seeding about 20 years from seedling and about 10 years from coppice, and produces abundant seeds almost every year (Seth and Kaul 1978). Teak forest is best managed under "coppice with standards system," in which 25–50 trees per hectare are selected as standards, on the basis of their larger diameter, and are retained as seed bearers. The remainder trees are clear-felled to produce coppice shoots. The rotation varies between 30 and 60 years; in rare instances it is 80 years (Pandey and Brown 2000). In some areas "coppice with reserves" system is followed wherein advance growth and pole size trees are retained as reserves which provide large size timber in the next rotation (Pandey and Brown 2000). Site classification, stocking level, thinning, and harvesting operations are guided by the yield tables available with the forest managers. For effective management, existing forests are stratified into working circles. Working circle refers to a forest area (forming the whole or part of a working plan area) organized with a particular object, and under one silvicultural system and one set of working plan prescriptions. The best teak forests (well stocked with fair quantity

Table 18.2 Density of teak for fully stocked teak forest (based on Nigam 1968)

GBH (cm)	Spacing (m)	Number of plants per hectare
30.5	3.66	741
38.1	4.27	544
45.7	4.88	430
61.0	6.10	272

GBH girth at breast height

Table 18.3 Exploitable girth for species under selection-cum-improvement management of relatively inferior teak forest (based on Nigam 1968)

Species	Exploitable girth (cm) at breast height
Tectona grandis, *Mytragyna parvifiora*	120
Adina cordifolia, *Terminalia tomentosa*, *Pterocarpus marsupium*, *Terminalia bellirica*	125
Dalbergia sissoo, *Gmelina arborea*, *Diosyoros melanoxyion*, *Soymida febrifuga*	105
Anogeissus latifolia, *Lagerstroemia parviflora*, *Ougeinia oojinensis*	90

of advance growth) are managed under "Teak conversion working circle" (Nigam 1968). Objective is to obtain well stocked/even-aged crop. Four treatment types are recognized (1) no felling (leave the forest as it is), (2) felling of limited number of trees, (3) clear felling with reservation of advance growth, and (4) plantation if the crop is very degraded. For full stocking, a spacing of 3.6 m $\times$ 3.6 m or 740 plants per hectare in the form of seedling, sapling, or coppice shoots up to 22.5 cm girth at breast height is provided. In the case of saplings or poles above this girth, wider spacing is provided (Table 18.2). A conversion period of 80 years is adopted. Within this time, the expectation is that crop will become even aged.

Relatively inferior teak forests, growing in poor sites (stocking not very good and regeneration and advance growth not adequate), are worked under "Selection-cum-improvement working circle." The felling cycle is kept at 40 years. Only dead, wind fallen, and top broken trees are marked for felling in areas which are steep or understocked or lie along water courses. In other areas, 50% of sound and well-grown exploitable trees (between 90 and 125 cm girth) are marked for felling (Table 18.3). To provide optimal growing space to young regeneration, cleaning in the 6th year and thinning in the 21st year are also prescribed (Nigam 1968).

Inferior teak and mixed forests (poorly stocked and inadequate regeneration), occurring in sites poorer than the above, are kept under "Improvement felling working circle." Here only improvement felling (removal of inferior growing stock of all ages, primarily dead and dying trees, and tending the better elements of the crop) is prescribed. The felling cycle is fixed at 30 years and the prescriptions are similar to preceding working circle with reduced exploitable girths on account of poor growth (Table 18.4).

Generally, mixed forests near human habitation are allotted to "Coppice with reserves working circle." The rotation is kept at 50 years. All growth up to 30.5 cm

Table 18.4 Exploitable girth for species under improvement felling management of inferior teak and mixed forest (based on Nigam 1968)

Species	Exploitable girth (cm) at breast height
Tectona grandis, *Soymida*, *Mitragyna*, *Chloroxylon swietenia*	105
Adina cordifolia, *Terminalia tomentosa*, *Pterocarpus*, *Terminalia bellirica*, *Hardwickia*	120
Dalbergia sissoo, *Diospyros melanoxylon*, *Gmelina arborea*, *Anogeissus latifolia*, *Lagerstroemia parviflora*	90
Grewia tiliaefolia, *Ougeinia oojeinensis*	60

Table 18.5 Growing stock and increment in even-aged mature crops of *Shorea robusta* (125 year) and *Tectona grandis* (80 year) forests (after Champion and Seth 1968b; Seth and Kaul 1978)

	Shorea robusta	*Tectona grandis*
Number of trees (>20 cm diameter) per hectare	138	111
Basal area (m^2 ha^{-1})	15.6	11.9
Growing stock, stem timber (m^3 ha^{-1})	85	46
Mean annual increment of stem timber (m^3 ha^{-1} a^{-1})	1.5	1.0
Mean annual increment of stem timber plus small wood (m^3 ha^{-1} a^{-1})	3.9	5.0

Mean annual increment is calculated from total yield (including all thinning plus final yield) of a plot/divided by harvesting age

girth (for teak 45.7 cm) is reserved and groups of poles 30.5–61 cm girth (for teak 45.7–61 cm) are thinned in well-stocked patches to produce high value trees.

Growing stock and increment in even-aged mature crop of teak of coppice origin is given in Table 18.5.

18.4 Sal-Bearing Forest

These forests are typical of the Vindhyan tract which extends over northern Madhya Pradesh and southern areas of Uttar Pradesh. Here sal is commonly found instead of teak. Sal is economically important but grows best in higher rainfall areas that support moist deciduous forest (Troup 1921; Tewari 1995). Flowering in sal occurs at the age of nearly 15 years and the maximum life span of the tree exceeds 160 years (Champion and Seth 1968b). All sal bearing forests except those which are degraded and excessively grazed are worked under “Coppice with reserves working circle” (Singh 1980). These forests have been traditionally managed through selection felling, i.e., harvesting of individuals above a certain diameter and leaving a few mother trees for regeneration (Upadhyay and Srivastava 1980; Harikant and Ghildiyal 1982). The interval and diameter for harvest varies according to species. No tree less than 10 cm diameter is felled. The diameter considered suitable for felling is >30 cm for *Boswellia serrata*, >50 cm for *Shorea robusta*, >70 cm for *Sterculia urens,* and >60 cm for other species. The rotation period for felling is 30 years except for the fast growing *Holarrhena antidysenterica* for which it is 10 years. Leaving 15–40 mother trees per hectare, beyond recommended diameter for felling of *Shorea robusta, Terminalia tomentosa, Anogeissus latifolia, Lagerstroemia parviflora, Adina cordifolia, Acacia catechu, Hardwickia binata, Miliusa tomentosa,* and *Chloroxylon swietenia*, is practiced. The purpose of reservation is to leave an adequate number of trees of important species to serve as seed bearer and to produce some large sized timber at the next felling. The rotation is fixed at 30 years when the crop is expected to attain 16.4 cm diameter. General principles by which felling is controlled are given in Table 18.6. Degraded and excessively grazed areas are managed under "Protection working circle." Here only improvement felling, consisting of removal of the dead trees is practiced.

Table 18.6 Control of felling under "coppice with reserves" system for sal bearing forest for natural regeneration by coppice and seed (based on Singh 1980)

Species	Reason for retaining
Buchanania lanzan	Fruit tree
Emblica officinalis	Fruit tree
Madhuca latifolia	Fruit tree
Mangifera indica	Fruit tree
Syzygium cumini	Fruit tree
Tamarindus indica	Fruit tree
Terminalia bellirica	Fruit tree
Terminalia chebula	Fruit tree
Bridelia species	Bird nesting
Ficus species	Bird nesting
Sterculia urens	Gum tapping
Bombax ceiba	Uncommon valuable species[a]
Dalbergia sissoo	Uncommon valuable species[b]
Gmelina arborea	Uncommon valuable species[b]
Hymenodictyon excelsum	Uncommon valuable species[a]
Tectona grandis	Uncommon valuable species[b]
Acacia catechu	Advance growth[c]
Adina cordifolia	Advance growth[c]
Anogeissus latifolia	Advance growth[c]
Ougeinia oojeinensis	Advance growth[c]
Shorea robusta	Advance growth[c]
Terminalia tomentosa	Advance growth[c]
Acacia catechu	Seed bearers and standards[d]
Adina cordifolia	Seed bearers and standards[d]
Albizia species	Seed bearers and standards[d]
Anogeissus latifolia	Seed bearers and standards[d]
Dalbergia latifolia	Seed bearers and standards[d]
Diospyros melanoxylon	Seed bearers and standards[d]
Lagerstroemia parviflora	Seed bearers and standards[d]
Shorea robusta	Seed bearers and standards[d]
Terminal1ia tomentosa	Seed bearers and standards[d]

The list could vary from region to region
[a]Wood for artifacts
[b]Timber
[c]<10 cm diameter trees are retained
[d]25–50 straight, healthy and vigorously growing young stems per hectare are retained

Sal is well adapted for the production of successive even-aged crops but for doing this a system of concentrated regeneration (i.e., complete felling followed by regeneration through coppice supplemented with, if needed, artificial regeneration by planting) is needed (Troup 1921).

Growing stock and increment in even-aged mature sal forest of coppice origin is given in Table 18.5.

The dry forest is facing problem of natural regeneration, particularly because of illegal harvest of saplings, free-range grazing, and fuelwood and timber extraction (Jha and Singh 1990; Sagar et al. 2003; Sagar and Singh 2005). As a result, >50% of recorded species exhibit demographic instability (Sagar and Singh 2004). The success of seedling growth to saplings and of saplings to adult trees has been relatively low in many species (Sagar and Singh 2006). It has been reported that the intensity of anthropogenic disturbance and soil water holding capacity control the understory (seedlings and saplings) composition through effects on soil organic matter (Sagar et al. 2008). The immediate priority, therefore, should be to control disturbance, particularly cattle grazing and fuelwood collection, as they are the major cause of site quality deterioration in the area. Systematic fuelwood plantations of fast growing trees on the village commons, setting aside selected forest compartments for raising high density short rotation energy plantations, and developing village pastures with a mixture of grasses and legumes with scattered native fodder trees (such as *Hardwickia binata*, *Dalbergia sissoo* and *Holoptelea integrifolia*), could be a viable strategy for easing the anthropogenic pressure on these forests. In addition, the existing vegetation on various sites may be enriched by seeding and planting of field-collected or nursery-raised seedlings of desired native species through aggressive forestry. Further, there is a need to integrate the livelihood of local human populations with conservation measures through participatory forest management such that the local inhabitants are able to appropriate a large share of benefits from conservation of these forests (Sagar and Singh 2005, 2006).

18.5 Conclusions

The dry deciduous forest is the most extensive forest type of India and comprises a variety of plant communities in response to marked spatial variation in soil and climatic conditions. Two subgroups, southern and northern, are clearly distinguished. The climax types under the southern subgroup are teak (*Tectona grandis*) forests, red sander (*Pterocarpus santalinus*) forests, and mixed forests without teak. The northern subgroup is divided into two types: sal (*Shorea robusta*) bearing forests and mixed forests without sal. These forests primarily cater to the local demand and production of fuelwood, small timber, and other NTFPs is the main objective of management. “Simple coppice,” “coppice with standards,” and “coppice with reserves” are the most prevalent silvicultural systems. The forests, however, are under-stocked and lack natural regeneration on account of excessive grazing, trampling, firewood removals and recurrent fire.

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Chapter 19
Silviculture of Tropical Dry Forests: Bolivian Case Study

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Abstract Tropical dry forests in Bolivia are considered to be the best conserved in Latin America and provide many forest resources, particularly wood products and environmental services. The objectives of this chapter are to demonstrate the limitations of current forest management, considering the characteristics of the tree species, and to suggest the use of silvicultural treatments to improve production and sustainability. There are various problems with natural regeneration in these forests, especially with regeneration from seed origin, which is principally caused by the long dry season. However, many species do not have regeneration problems because of their ability to resprout. The overall growth rate shown (mean = 3 mm year^{-1}) is another limitation to the recovery of stand volume after harvesting, which is exacerbated by a high rate of liana infestation. The majority of the commercial species are long-lived pioneers or partially shade-tolerant species, and the abundance of trees varies greatly by species. The majority of commercial species have wind-dispersed seeds with an annual seed production cycle. Approximately half of the species resprout from cut stems or root systems. In Bolivia, including dry forests, reduced impact logging and silvicultural treatments are employed to increase growth rates and reduce stand damage. The most effective reduced impact logging practices include directional felling, planning of skid trails, and marking of future crop trees. For silvicultural treatments, prescribed burning increases the abundance of seedlings and saplings as well as the abundance and growth rates of resprouts. Liana cutting and liberation of crop trees increases diameter growth with relatively low investment. Despite limitations to management in Bolivian dry forests, the sustainability of harvesting can be enhanced by employing reduced impact logging practices combined with silvicultural treatments that increase the regeneration and growth rate of commercial tree species.

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19.1 Introduction

Tropical dry forests are considered to be endangered because of continuing threats from conversion to agriculture and cattle ranching, the illegal extraction of timber, and exploitation for minerals and energy sources, and the slow recuperation from disturbances of these ecosystems (Dinerstein et al. 1995). In Bolivia, a relatively large amount of dry forests remain largely intact (Parker et al. 1993). In fact, along with Mexico, it contains one of the largest areas of tropical dry forests in the world, wherein a large amount of plant and animal diversity is protected (Dinerstein et al. 1995). From a forestry and conservation perspective, Bolivian dry forests are important for their large number of forest products, their potential productivity (Dauber et al. 2001), and also they represent a complex mosaic of ecosystems generated by their geological, climatic, and altitudinal variation (Ibisch and Mérida 2003; Killeen et al. 2006).

The lowland dry forests in Bolivia, locally known as the Chiquitano dry forest, is part of the Brazilian Shield and is characterized by rolling hills with superficial thin oxisols, mostly derived from gneiss or other granitic rocks (Cochrane 1973; Navarro and Maldonado 2002). There is a N–S rainfall gradient from 1,500 to 2,000 mm year^{-1} in the subhumid forest in the Amazon Basin to 500 mm year^{-1} in the Chaco forest. Tropical dry forests are part of a complex mosaic that includes Cerrado vegetation and outcrop rocks (Killeen et al. 2006). Tropical dry forests in Bolivia are suffering a high rate of deforestation because of the pressure to convert forests to cattle pastures and agricultural areas. These forests are increasingly threatened by development pressure radiating out from the city of Santa Cruz (Camacho et al. 2001).

In Bolivia, forest management has experienced rapid development in the past few decades including the promulgation of a new forestry law and a rapid increase in forest area under third-party certification (Nittler and Nash 1999). A number of neighboring countries have adopted the Bolivian forestry example, which was initiated in the dry forests of the country. One of the major features of the model is the implementation of forest management practices designed to reduce logging damage and help achieve sustained timber yield. The model includes designation of cutting cycles and minimum cutting diameters, maintenance of seed trees, harvest planning to reduce stand damage and increase harvest efficiency, and promotion of the application of silvicultural treatments.

A large investment has been made in silvicultural research with a large portion of it carried out in dry forests. Silvicultural treatments have been developed to resolve problems with achieving sustainable forest harvesting for dry forest tree species. In this chapter, I review the experiences with the development and application of silvicultural treatments. I first review ecological limitations and problems caused by traditional forest harvesting in these forests and then discuss the development of different silvicultural treatments.

19.2 Ecological Limitations to the Management of Bolivian Dry Forests

There are a number of obstacles to forest management in dry forests including problems with regeneration, relatively slow growth rates of trees, and competition from lianas. In this section, these obstacles are described as well as efforts to overcome them using silvicultural treatments.

19.2.1 Natural Regeneration

Approximately 60–70% of the timber species in Bolivia have problems with natural regeneration (Mostacedo and Fredericksen 1999). In dry forests, resource limitations, as well as the lack of appropriate conditions with germination and establishment present problems for forest management. Regeneration in dry forests is impeded by a long dry season, which may result in mortality of as much as 90% of tree seedlings (Khurana and Singh 2001). For example, *Copaifera chodatiana* and *Tabebuia impetiginosa* have a high germination percentage, but because of the long dry period, nearly all seedlings die. Only a few species have the capability to withstand 5–6 months of drought, and survival is often dependent on resprouting from root systems after the dieback of stems (Mostacedo et al. 2008). In this group, *Amburana cearensis*, *Aspidosperma rigidum*, and *Centrolobium microchaete* tolerate long periods of drought. Canopy openings created by logging gaps, roads, and skid trails can increase sunlight to the forest floor and exacerbate the effects of drought on tree seedlings. Athough the quantity of light reaching the forest floor relative to the forest canopy is only 2–5% during the growing season, leaf drop in the dry season may increase relative light levels to 30%. In the dry season, seed predation is high in tropical forests due to lack of food resources for animals and further reduces the availability of seeds for germination (Janzen 1970). In addition, herbivory can eliminate 90% of germinated seedlings (Grol 2005; Mostacedo 2007). Following forest harvesting, even with the retention of 20% of potentially harvestable stems as seed trees, there is often insufficient seeds to provide for the regeneration of many timber species, although some species may regenerate from resprouts (Mostacedo et al. 2008).

19.2.2 Tree Growth Rates

Because of the limitation of rainfall (1,100 mm and a 5-month dry season) and their intrinsic growth characteristics, tropical dry forest trees grow much more slowly than trees in other tropical forests. The slow growth presents a problem for forest management because it increases the length of cutting cycles. Trees in the Chiquitano dry forest of Bolivia grow an average of 3 mm year^{-1} (Dauber et al. 2003), which is approximately 50% less than the growth rates in humid forests. Worse yet, some shade tolerant species or those with poor crown condition or infested with vines may only grow 1 mm year^{-1}. During the dry season, trees go dormant and stems may even shrink due to water loss (Dauber et al. 2003).

19.2.3 Liana Infestations

Although lianas are important structural components of tropical forests and their biodiversity (Schnitzer and Bongers 2002), they are considered a problem for forest management (Fredericksen and Peralta 2001). Lianas compete with trees for light, nutrients, and water (Putz 1980), can reduce timber yields, and cause injuries to workers during forest harvesting (Alvira et al. 2004). The abundance and diversity of lianas seems to be related to the degree of disturbance to forests, with more disturbed areas containing more stems and species of lianas (Vroomans and Toledo 2008). In many cases, the abundance of lianas in Bolivian forests is much higher than that reported in other tropical forests (Perez-Salicrup et al. 2001). Fire and illegal logging, and in some cases strong winds, are the most frequent disturbances to these forests. In general, Bolivian dry forest tree species have higher rates of liana infestations. In forests of Lomerio, 77% of trees had at least one vine and 35% of trees had crowns totally covered by lianas (Carse et al. 2000). In two other subhumid forests, liana infestation rates were higher than 70%, while more than 80% of trees with a diameter-at-breast-height >50 cm were infested by lianas (Alvira et al. 2004; Perez-Salicrup et al. 2001).

19.3 Commercial Species and Ecological Requirements

In the Chiquitano dry forests, there are fewer than 12 species with national or international markets. Each species differs with respect to ecological characteristics including natural abundance, natural regeneration, and growth. One of the most abundant commercial species is *Anadenanthera macrocarpa* (Table 19.1), which, in some areas, represents more than 30–40% of all commercial tree species. This dense hardwood species has a regular diameter distribution and could sustain a high rate of extraction. *Caesalpinia pluviosa* and *Centrolobium microchaete* are species with an abundance of mature individuals. The other species are less abundant, and some such as *Cedrela fissilis*, *Cordia alliodora*, and *Zeyheria tuberculosa* are relatively rare, although they exist in quantities that are potentially harvestable.

The diameter growth increments of tree species in the Chiquitano dry forest are among the smallest in Bolivia with an average increment of 0.19 cm year^{-1} (Dauber et al. 2003). These increments are similar to those reported in other tropical dry forests (Murphy and Lugo 1986; Uslar et al. 2004), and much lower than increments reported in tropical humid forests. *Centrolobium microchaete*, *Anadenanthera macrocarpa*, *Cordia alliodora*, and *Hymenaea courbaril* have the largest

Table 19.1 Ecological characteristics of the main commercial tree species in the Chiquitano tropical dry forests of Bolivia

Species	Family	Density (# ha^{-1})[a,b]	Diameter increment (cm $year^{-1}$)[c]	Ecological guild[d,e]	Natural regeneration (# ha^{-1})[f]	Mode of regeneration[g]	Fruiting frequency[g]	Mode of dispersion[g]	Germination (%)[h]
Amburana cearensis	Fabace.	0.9	0.22–0.31	LLP	10	Se	Supra-annual	Wind	80
Anadenanthera macrocarpa	Mimosa.	10.4–25.3	0.27	PST	2,895	Se	Annual	Auto	48
Aspidosperma rigidum	Apocyn.	1.9–4.3	0.05–0.07	PST	400	Se, Re	Annual	Wind	70
Astronium urundeuva	Anacar.	0.5–3.3	0.13–0.15		21	Se	Annual	Wind	80
Caesalpinia pluviosa	Mimosa.	8.2–10.1	0.23–0.24		891	Se	Annual	Auto	90
Cedrela fissilis	Meliac.	0.4	0.27–0.30	LLP	7	Se	Annual	Wind	85
Centrolobium microchaete	Fabace.	5.4	0.30–0.50	LLP	68	Re, Se	Annual	Wind	8
Copaifera chodatiana	Caesal.	1.3–2.4	0.11–0.16	PST	1,327	Se	Annual	Animal	65–80
Cordia alliodora	Boragi.	0.2	0.46	LLP		Se	Annual	Wind	90
Hymenaea courbaril	Caesal.	0.3–0.4	0.45	TST	23	Se	Annual	Animal	20
Machaerium scleroxylon	Fabace.	1.1–2.0	0.20–0.24		150	Se	Supra-annual	Wind	10–30
Platimiscium ulei	Fabace.	–	0.30–0.37	LLP	–	Re, Se	Annual	Wind	–
Schinopsis brasiliensis	Anacar.	0.4	0.15–0.17		56	Se	Supra-annual	Wind	–
Swettia fruticosa	Fabace.	0.9	0.13–0.26	LLP	522	Se	Annual	Wind	–
Tabebuia impetiginosa	Bignon.	0.8–5.0	0.09–0.14		6	Se, Re	Annual	Wind	28
Zeyheria tuberculosa	Bignon.	0.2–0.4	–	LLP	17.5	Re	Supra-annual	Wind	

LLP long-lived pioneer, *PST* partial shade-tolerant, *TST* total shade-tolerant

Family: The name of families is represented only for the first six letters. Ecological guilds: The density of individuals shows the range of variation between two different sites: INPA and Lomerio. Natural regeneration: <3 m tall. Mode of regeneration: *Se* seed; *Re* resprout

References: [a]IBIF, unpublished; [b]WWF (2009); [c]Dauber et al. (2003); [d]Poorter and Kitajima (2007); [e]Poorter (2007); [f]van Andel (2005); [g]Mostacedo (2007); [h]Mostacedo and Fredericksen (2001) growth increments, while *Aspidosperma rigidum* and *Tabebuia impetiginosa* have the lowest.

The majority of these commercial dry forest species are either long-lived pioneers or partially tolerant of shade. Only *Hymenaea courbaril* is considered to be shade-tolerant.

On the one hand, the species with the most abundant seedling regeneration are *Anadenanthera macrocarpa* and *Copaifera chodatiana*. On the other hand, *Cedrela fissilis* has poor regeneration due to a low abundance of seed trees, and *Tabebuia impetiginosa* regeneration is probably severely limited by drought. Although the regeneration of most species comes primarily from seed, resprouting is important for others (Mostacedo et al. 2008). The seeds of approximately 70% of commercial tree species are wind-dispersed, although secondary seed dispersal may be carried out by insects or small mammals. *Hymenaea courbaril* is the only species principally dispersed by animals. The majority of species have an annual seed production cycle, but some have masting cycles of 2–3 years. Seed production frequency is important for defining the percentage of seed trees to leave after harvesting. For example, *Machaerium scleroxylon* produces seeds on a 2-year cycle, during which nearly all trees produce an abundance of seeds. Greenhouse germination tests have shown that nearly half of Bolivian dry forest species have germination percentages >70% (Table 19.1). Some species, such as *C. microchaete*, *H. courbaril*, and *T. impetiginosa*, have percentages <30% (Mostacedo and Fredericksen 2001). Although *C. microchaete* has a germination percentage <10%, it regenerates prolifically by resprouting.

19.4 Results from Research on Silviculture Treatments in the Bolivian Dry Forests

Numerous studies on silvicultural treatments have been conducted in the Bolivian dry forests. Although research on silviculture is not unique to Bolivia, this research has helped to convince many Bolivian forest managers to use them to improve forest production.

19.4.1 Harvesting

Forestry is considered the most important silvicultural treatment, since it creates conditions necessary for the regeneration and accelerated growth of many tree species. During harvesting, some other practices that can increase the efficiency of management operations include the planning of skid trails, directional felling, and flagging of future crop trees. In a study in a subhumid forest in Bolivia, skid trail planning reduced the time needed to search for harvested trees by 90% (Krueger 2004). Flagging future crop trees reduced damage to them by 10–20%. Likewise, directional felling increased the efficiency of extraction and reduced damage to surrounding trees. Finally, a commercial-scale study indicated that harvesting damage did not increase with increasing harvest intensity with the use of reduced impact logging practices (Mostacedo et al. 2006). Tree growth rates increased with a doubling of harvest intensity (from 4.1 to 8 m^3 ha^{-1}) when combined with silvicultural treatments to liberate future crop trees from lianas and overtopping trees (Villegas et al. 2008b).

19.4.2 Treatments to Improve Natural Regeneration

The use of fire, mechanical thinning, and herbicides are treatments that can increase light availability and promote natural regeneration of forest trees. The availability of nutrients, water, and light are perhaps the most important factors for increasing regeneration (Markesteijn et al. 2008; Oussoren 2008; Poorter and Markesteijn 2008). Measures of regeneration success include both the abundance of seedlings and saplings as well as their growth rates.

Prescribed fire as a site preparation treatment has been tested in Bolivian dry forests. One study demonstrated that intense burns in logging gaps enhance the abundance of saplings $\geq$2.5 m tall by 3$\times$ and of seedlings by 2$\times$, although the regeneration of shade-tolerant species is not necessarily improved (Kennard 2004; Oussoren 2008). High-intensity burning significantly increased the tree density of several species including *Anadenanthera macrocarpa*, *Astronium urundeuva*, *Copaifera chodatiana*, and *Acosmiun cardenasii* (Kennard 2004). For other species, such as *Centrolobium microchaete* and *Caesalpinia pluviosa*, fire did not increase regeneration density. Regeneration by resprouting improved seed origin regeneration by 2$\times$ (Kennard 2004), and resprouts for all commercial (mean 162 cm) grew 2$\times$ more than seedlings (68 cm) in burned gap sites (Oussoren 2008). However, other studies indicate that the cost of conducting controlled burns in logging gaps does not justify the use of this treatment (Heuberger et al. 2002).

Mechanical removal of vegetation with machetes or logging machinery did not significantly increase the density or growth of seedlings in a Bolivian dry forest (Heuberger et al. 2002), but it was noted that the timing of treatments is important with respect to seed dispersal. Without a viable seed source or seed bank, cleaned sites are quickly invaded by competing vegetation. In addition, competing species grow much more rapidly than the regeneration of commercial tree species, except for resprouts (Mostacedo et al. 2008).

19.4.3 Liberation Treatments

Liberation of future crop trees is one of the most utilized silvicultural treatments in Bolivian dry forests. Liberation treatments include liana cutting and liberation of future crop trees from neighboring competitors. Once cut, lianas can resprout and increase the effectiveness of treatments, herbicide application to cut stems is recommended, although some liana species may still resprout following application (Fredericksen 1999). The cost of vine cutting averaged only \$1.64 and 1.85 ha^{-1} with and without the use of herbicide, respectively. Liana cutting increases the growth of liberated trees when compared with controls (Villegas et al. 2008b). In a subhumid forest in Bajo Paragua, liana cutting increased the growth of trees by 0.36–0.7 cm $year^{-1}$ (Villegas et al. 2008a). The internal rate of return in 5 years could amount to 9% simply by vine cutting (Evans et al. 2003).

The liberation of future crop trees from lianas or competing trees could be carried out in conjunction with timber harvesting. Liberation treatments are conducted using girdling followed by herbicide application. Double ring girdling is more effective than single ring girdling, and girdling during the dry season results in a mortality rate of 71% when compared with 66% mortality for girdling during the rainy season (Ohlson-Kiehn et al. 2006). The efficacy of these treatments is lower in dry forests compared to humid forests. Regardless of this variation, liberation treatments are highly effective in increasing the growth rates of future crop trees. In a Chiquitano dry forest, liberation treatments increased tree growth between 0.18 and 0.30 cm $year^{-1}$ (Villegas et al. 2008b), while in a subhumid forest in Bajo Paragua, growth was increased between 0.7 and 0.9 cm $year^{-1}$ (Villegas et al. 2008a). Future crop trees of *Centrolobium microchaete*, *Caesalpinia pluviosa*, and *Copaifera chodatian*, which are long-lived pioneers or partial shade-tolerant species, displayed an increase of 25–60% in diameter increment in areas with a higher intensity of logging and liberation treatments compared to normal harvesting and without liberation compared to normal(Villegas et al. 2008b). Also, a fully shade-tolerant species, *Acosmiun cardenasii*, had a 40% increase in diameter increment in higher-intensity logging areas. However, there are other species, such as *Machaerium scleroxylon*, *Zeyheria tuberculosa*, and *Hymenaea courbaril*, that did not respond significantly in diameter increment in areas with higher-intensity harvesting and liberation treatments (Villegas et al. 2008b).

19.5 Implementation of Silvicultural Treatments in Bolivian Dry Forests

Silvicultural treatments in Bolivia are encouraged under the best management practices described in Forestry Law 1700 (MDSP 1998). Various silvicultural practices are essentially mandated as part of best management practices related to reduced impact logging. Harvesting must include preharvest mapping of harvestable trees above established minimum diameter limits, designation, and protection of seed trees (20% of harvestable trees), planning of logging roads and major skid trails, preharvest liana cutting, and directional felling. For most commercial species, the minimum cutting diameter is 40 cm, but it is 45 cm for *Anadenanthera macrocarpa* and *Amburana cearensis* and 60 cm for *Cedrela fissilis*. Forest owners or concessionaires must also set a portion of the total property in protected areas and calculate cutting cycles based on survey of the timber inventory on the property.

The minimum cutting cycle mandated by law is 20 years, but several concessionaries have used a cutting cycle of 25–30 years. However, because tree growth rates in this forest are low, cutting cycles would probably need to be more than 30 years for sustained yield.

The application of other silvicultural treatments in Bolivian forests is increasing, although there is resistance to employ treatments that do not demonstrate significant short-term economic benefits. However, research results have demonstrated the benefits of treatments for reducing damage to the residual forests during harvesting (5–10% damage for trees $\geq$20 cm dbh), increases in regeneration and tree growth rates, and improvement in timber stand quality. Many industry- and community-owned forests are experimenting with the use of silvicultural treatments, and liana cutting has become a common practice in recent years, although it is most often employed only on trees to be harvested to reduce danger to sawyers and to improve directional felling. The use of liana cutting to improve the growth rates of future crop trees is still not a common practice in Bolivia. Another practices commonly carried out in Bolivia include marking future crop trees to protect them during felling and skidding operations. This practice is simple and inexpensive to carry out in conjunction with preharvest inventories using paint or flagging.

19.6 Conclusion

The experimentation with silviculture in Bolivian dry forests could be useful for other countries with similar forests, although these forests have their special characteristics that require specific silvicultural solutions. Research has been carried out on a wide variety of treatments and some have been implemented on an operational scale by forest managers. Others require more refinement or more extension before they will be adopted. There are major problems with securing natural regeneration, improving the slow growth rates of timber species, and freeing trees from infestations by lianas in Bolivian dry forests. Because of the slow recovery of volume after harvesting in dry forests, it is more difficult to realize the results of silvicultural treatments when compared with that of humid forests. The challenge in these forests is to better understand the ecological basis for silviculture in these forests and to minimize, as much as possible, the damage to the residual stand during harvesting.

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Part VI
Silviculture in Azonal (Semi-)Natural Forests

Chapter 20
Review
Mangroves and Mountains: Silviculture at Ecological Margins

Sven Günter

Abstract Humidity and light were the most dominant ecological factor for silviculture in the ecosystems presented by Ashton and Hall (this volume) and Fredericken (this volume). However, several other factors can also shape structure and function of forest ecosystems and hence have great impact on the corresponding silvicultural potential. At the example of mountain forests and mangroves, basic similarities and differences of forest ecosystems at ecological margins and their silvicultural potential are highlighted in this paper. As simliar to other ecosystems at ecological margins, mangroves and montane forests are of extraordinary importance for the provision of environmental services or are characterized by a relatively high proportion of endangered species. In turn, they are frequently highly endangered by human pressure and global change, converting them into high priority areas for conservation aspects. At ecological margins growth conditions are frequently poor, individual tree dimensions and timber yields are low. Forest management, based on timber alone can therefore hardly compete with alternative land use forms, which finally leads to conversion into agricultural land and deforestation. Even more than in other ecosystems, payments for environmental services, REDDplus- mechanisms, the use of non-wood forest products, or a combination of all these approaches may help to increase the economic attractivity for sustainable forest management and thus allow to combine conservation aspects with additional income for rural population.

20.1 Introduction

Manifold ecological factors can shape structure and dynamic and ecosystem functions of forest ecosystems: for example, temperature, salinity, nutrient availability and many others, but it is impossible to cover all tropical forest formations which can be formed by a combination of these factors in this book. While the chapter "humid

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forests" (Chap. 12) was attributed to ecosystems with almost no limitation by certain abiotic factors, the second chapter "dry forests" (Chap. 16) describes the influence of limited humidity on structure, dynamics and silviculture of tropical forests. Exemplarily for many other forest ecosystems of the world, in this chapter we will focus on the ecology of two forest ecosystems at ecological margins and the potential role of silviculture: montane forests (altitudinal gradient) and mangroves (coastal gradient).

Besides very obvious differences between all ecosystems at ecological margins, they also may have some aspects in common. Usually, smaller tree dimensions and growth limitation are effects of environmental stress. Reduced yield in turn is an awkward precondition for sustainable forestry. Does this generally indicate that forests at ecological margins are marginally profitable? Structural and floristic variability is typical for tropical forests. For ecosystems at ecological margins, these changes are superimposed by the corresponding ecological gradients. According to the intermediate disturbance hypothesis (IDH), this may create additional ecological niches with accumulation of species in the middle of the gradient. Accordingly to IDH, increasing environmental stress will cause a reduction of biodiversity, sometimes very abruptly close to the tree line. However, many organisms are able to adapt to amazingly harsh life conditions. The harsher the life conditions, the lower is the chance for survival of species with intermediate requirements of ecological conditions. Consequently, forests at ecological margins frequently harbour a high number of endemic species, especially when they were isolated for a long time. This indicates a rather higher importance of conservation issues for these ecosystems.

Finally, these forests are highly threatened by climate change and human pressure. Firstly, low growth rates can reduce economic profitability of forest management and hence increase the probability for conversion into agricultural land in many cases. Secondly, while other forests can partly compensate climate changes by spatial shifts (though accompanied by shifts in synecological barriers), forests at ecological margins face more pronounced autecological barriers and are consequently more susceptible to environmental changes. This may raise the question, what can be the role of silviculture in ecosystems with generally lower growth rates and high relevance for conservation (see Chap. 7)?

Complementary to the dry forest and humid forest chapters of this book, an overview will be given about ecological characteristics of mangroves and montane forest ecosystems, their structure, dynamics and silvicultural aspects. The final section of this paper will address the rationale of silviculture at ecological limits, considering conservation aspects.

20.2 Mangroves

20.2.1 Ecological Factors

Mangroves cover about 140,000–200,000 km^2 along tropical and subtropical coastlines from the mean sea level up to the highest spring tide level (Lüttge 2008; Spalding et al. 1997). Nearly 60–75% of all tropical coast lines are covered by mangroves (Lüttge 2008). Mangroves are formed by trees with main distribution in the intertidal zone between terrestrial and marine ecosystems (Nagelkerken et al. 2008). Three major abiotic categories of environmental factors influence the mangrove ecosystems: tidal regime, edaphic conditions (including salinity) and climate (Iftekhar and Saenger 2008).

Tidal inundation is an ideal proxy for the wide variety of environmental conditions that affect plant growth, including soil salinity, redox potential and waterlogging (Ellison 2002). As adaptation to frequent flooding, mangroves have developed an extended and unique root system (above the soil surface) to provide anchorage as well as aeration. The root systems can be classified into stilt-type, buttress-type and knee- or finger-like pneumatophores (Lüttge 2008). Pneumatophore aerenchyma can occupy considerable percentages of up to 70% of the total root volume system (Curran 1985). During high tide, root respiration reduces the O_2 concentration in the intercellular spaces of the root aerenchyma to a hypoxia of only 4–8% (Kitaya et al. 2002). According to the tidal amplitude, the forest can be classified into zones with different frequency of tidal inundation (Iftekhar and Saenger 2008; Watson 1928 cit. Siddiqi 1994; Jin-Eong 1995).

Mangroves have the unique capacity to tolerate large short-term changes of *salinity* (Lüttge 2008). In the Sunderbans in Bangladesh three salinity zones, oligohaline (salinity > 2 dsm^{-1}), mesohaline (2–4 dsm^{-1}) and polyhaline (>4 dsm^{-1}) are distinguished. Besides salinity, hypoxia is a prominent stress factor in mangrove ecosystems. Species with special adaptations thus have considerable competitive advantages.

Salinity is not only influenced by the tide but also by the *climate*, especially during times of low tide, i.e. humid climate with rainfall diluting and leaching salt and arid climate with concentration of salt (Lüttge 2008). Similar to other halophytes, mangroves can be subdivided into salt includers with intracellular salt dilution and compartmentation and salt excluders, the first ones resisting much higher salinity (Lüttge 2008). Salinity and irradiance stress may be additive. However, seedlings from most species require full sunlight without inference of salinity (Lüttge 2008). In contrast, *Bruguiera parviflora* and *Ceriops australis* required high salinity under full sunlight and established better under low salinity and 30% sunlight.

An alternative classification based on six physiographic types of mangroves is proposed by Lugo and Snedaker (1974): (1) riverine mangroves along rivers and streams are flooded by daily tides, (2) fringe mangroves are exposed to the open sea, periodically flooded by tides and very sensitive to erosion, (3) basin forests are located inland in depressions channelling terrestrial runoff to the sea; they are irregularly flushed by tides and are sensitive to flooding, (4) overwash mangroves typically occur on islands and are inundated in each tidal cycle with productivities similar to fringe mangroves, (5) scrub mangroves are found in extreme environments and tidal inundation is relatively infrequent and (6) hammock mangroves are restricted to elevated ground caused by accumulation of organic peat, and are infrequently flushed by tides.

20.2.2 Species and Diversity

The forests are strongly zoned in dependence from distance to the sea and elevation. For example, *Sonneratia alba* Sm was growing closest to the sealine along the Kenya coast, followed by *Rhizophora mucronata* Lam., *Bruguiera gymnorrhiza* (L.) Lam., *Ceriops tagal* (Perr.) C.B. Robinson, *Avicenna marina* (Forrsk.) Vierh., *Lumnitzera racemosa* Willd. and *Xylocarpus granatum* Koen. (Van Spreybroeck 1992). This zonation can partly be attributed to tolerance to salinity. For example, Wells (1982) found *Avicennia marina* growing in soils with salinity >65‰ in contrast to *Rhizophora mucronata* was restricted to salinity below 40‰.

Mangroves host very few tree species in comparison to other tropical forest ecosystem. For example, only 1–5 tree species occur in the Neotropics, 4 species in West Africa, 8 in East Africa, 25 in India and 30 species in Southeast Asia (Lüttge 2008). Ellison (2002) calculates a total of 50–75 species, a number which is easily exceeded by one single hectare of a typical lowland rain forest. The hot spot of mangrove tree diversity is the Indo-West Pacific region and species diversity declines smoothly from the equator towards higher latitudes (Ellison 2002). Interestingly, the classical assumption that the Indo-Malayan region is the origin of modern mangrove ecosystem is not supported by fossil records (Plaziat et al. 2001).

As homogeneity of species composition is relevant for forest management, it is important to note that single species can become very dominant in mangrove ecosystems. *Heritiera fomes*, for example, occupies between 20% and more than 45% area in the Sunderbans in monospecific stands and mixed with *Excoecaria agallachoa*, respectively (Iftekhar and Saenger 2008). *Xylocarpus granatum* represented more than 40% of the basal area in Malaysia (Ashton and Macintosh 2002). In East Timor, *Rhizophora apiculata* (mid-intertidal), *Avicennia marina* and *Ceriops tagal* (high-intertidal) can become very dominant (Alongi and de Carvalho 2008) too.

Despite their low diversity in tree species they are habitat for an amazingly broad range of organisms which do not occur in other forest ecosystems (Nagelkerken et al. 2008), for example, fish (400 species in Sunderbans; Iftekhar and Saenger 2008), brachyuran crabs, gastropods, bivalves, hermit crabs, barnacles, sponges, tunicates, polychaetes and spunculids, crocodiles, alligators, caimans, gharials, lizards, snakes, and birds (300 species in Sunderbans; Iftekhar and Saenger 2008). Many of these organisms use special ecological niches and exhibit a clear zonation within the mangrove habitat. Macrofaunal communities appear to be influenced by hydroperiod. In higher intertidal mangroves the soils are sandier and less frequently inundated, resulting in drier and more saline conditions, with leaf litter accumulation (Nagelkerken et al. 2008). Prawns and many other organisms apparently depend on tidal fluctuations too (Vance et al. 1996).

Several biotic–biotic and abiotic–biotic interactions make this ecosystem complex and unique. For example, wood-boring bivalves which are associated with nitrogen-fixing bacteria are considered to stimulate the decomposition of wood and thus, contribute to the fixation of nitrogen (Nagelkerken et al. 2008). Also, adult trees' roots leak oxygen into the soil and thus, could modify redox potential and sulphide concentration (McKee 1993).

20.2.3 Structure and Dynamics

20.2.3.1 Structure

In general, forest structure and biodiversity get more complex with increasing distance from the sea line and less influence of tide and salinity. Mangroves are structurally more simple than lowland rain forests, often lacking an understorey of ferns and shrubs and are less diverse (Alongi 2002). Mangrove trees have thicker leaves with lower turnover than those in upland forests (Ellison 2002). The densities (DBH > 2.5 cm) reach enormous values of more than 3,000 stems ha^{-1}, those of larger individuals in contrast are relatively low (ca. 165 stems ha^{-1} for DBH > 15 cm) in comparison to other tropical forest ecosystems (Iftekhar and Saenger 2008). The forests are vertically less structured with relatively low mean heights of about 10 m. Taller trees are formed in the fringe zone near the water's edge (Lüttge 2008). Some species can reach considerable mean heights of >25 m, for example, *Sonneratia apetala* (pioneer) or *Heriteria fomes* (climax) (Iftekhar and Saenger 2008). About 40–50 m of height can be reached by *Rhizophora mangle* along caribbean riverine forests (Cintrón et al. 1978). Basal areas usually range up to 33 m^2 ha^{-1}, for example, reported by Ashton and Macintosh (2002) for Sarawak, only in exceptional cases they can reach more than 50 m^2 ha^{-1}. However, scrub forms are frequently associated with high salinity during the dry season and present basal areas far below the mentioned values (Lüttge 2008). Above ground biomass can reach values of 220–350 tonnes ha^{-1} in Timor (Alongi and de Carvalho 2008; Alongi 2002). Exceptional high values (571 tonnes ha^{-1}) are reported for *Rhizophora mangle* in the Caribe (Cintrón et al. 1978). Senescence processes begin already approximately 70 years after the colonisation phase (Alongi 2002).

20.2.3.2 Regeneration

One of the major characteristics of mangrove regeneration is their vivipary and the ability to produce propagules making use of the tidal regime for a broader distribution of natural regeneration, e.g. *Rhizophora, Ceriops, Bruguiera, Kandelia* and *Nypa* (Hussain 1995). *Rhizophora mangle* propagules show the ability to survive 12 months of floating until they are exposed to shallow and untroubled waters, where they have to overcome a critical 2-week period of anchoring. But, while *Rhizophora* forests tend to regenerate via production of propagules, *Avicennia* forests regenerate with a combination of regenerative and vegetative growth. *Avicennia* may dominate over *Rhizophora* in areas of wind and hail damage and shows a high capability for coppicing (Duke 2001).

Large-scale disturbances tend to encourage regeneration of light-demanding species that are able to recover large openings by seed dispersal such as *Rhizophora apiculata, R. mucronata*, *R. stylosa* or *Bruguirea* spp. Smaller gaps, in contrast, favour species such as *Avicennia* spp. or *Sonneratia* spp. with the capability for resprouting from surviving stems (Walters et al. 2008). Two general regeneration pathways are discussed in the literature. Van Spreybroeck (1992) summarises the state of the art until the early 1990s: the so-called “stranding theory” stresses that all mangrove propagules are water dispersed before establishment. In contrast, the “self-planting-theory” emphasises that propagules are pointed to promote self-planting as they fall from the tree. The author found that self-planting is the major mechanism in undisturbed forests where the forests as such provides protection from waves. In overexploited or cleared forests waves can enter the forests and thus, the “stranding-strategy” becomes more important in these cases. These findings indicate that the usually pronounced zonation can be altered by logging activities. Also early natural succession stages should be less stable in zonation than advanced stages.

Hence, the small-scale gap dynamics are complemented by typical large-scale disturbances for mangroves such as hail, floodings or wave damages. In addition, also medium-scale disturbances are frequent, because mangrove trees generally die standing in small clusters (Duke 2001). The natural disturbance regime is considered as a reference for nature-based forest management. Thus, especially in regions with frequent large-scale disturbances and homogenous species, silvicultural instruments such as group selection or shelterwood methods are considered to be more important while selection systems in contrast fit better to regions with low disturbance regimes. These arguments focus mainly on species composition, but they largely disregard the protective functions of the forests.

20.2.3.3 Growth

Basal area growth in the interior zones tends to be about 1.5 times higher than in the fringe zone (Krauss et al. 2006). Information about volume increments is very rare: values of about 4–5 m^3 ha^{-1} $year^{-1}$ are reported for rather average conditions. Stands with largest standing stock tend to show highest volume increment with 9.7–17.9 m^3 ha^{-1} $year^{-1}$ (Devoe and Cole 1998). Table 20.1 shows some examples about species-specific growth data for Micronesia. The total above ground net primary production (NPP) averages up to 20 tonnes ha^{-1} $year^{-1}$ or more. Since

Table 20.1 Data for diameter growth of different mangrove species in Micronesia (adopted from Devoe and Cole 1998)

Species	Diameter growth (cm $year^{-1}$)
Rhizophora apiculata	0.25
Sonneratia alba	0.5
Rhizophora mucronata	0.37
Xylocarpus granatum	0.31
Bruguiera gymnorrhiza	0.35

30–50% corresponds to leaf production, the respective woody production amounts approximately10–14 tonnes ha^{-1} $year^{-1}$ (Amarasinghe and Balasubramaniam 1992). These values can only indicate approximations. The natural variability is enormous and single stands may differ greatly in these estimates.

20.2.4 Wood Production and Silviculture

20.2.4.1 Exploitation

Typically for most accessible natural forests worldwide, mangroves are frequently overexploited. For example, while in East Timor there has been a long tradition of harvesting sandalwood, nowadays these forests are mainly exploited for firewood (Alongi and de Carvalho 2008). The applied management systems are sometimes based on historical experience, which can lead to "almost" sustainable management in terms of yields and forest structure (see Chap. 22). However, in many cases, application of local experience does not prevent from overexploitation: scientific and societal conceptions of sustainability differ widely (López-Hoffman et al. 2006). Alongi and de Carvalho (2008) show that in slow-growing dry mangroves forests (<800 mm rain fall) biomass was reduced by 47–78% by small-scale logging for firewood. These drastical interventions were accompanied by significant increases in interstitial salinity, ammonium, sulphide, total sulphur, iron and manganese.

20.2.4.2 Clear-Felling Methods

Wood chip industry widely converted natural forests into plantations by applying clear felling systems with felling cycles of 20–30 years (Chong 2006). These systems are mainly based on technical and economical operability and do not consider very much ecological sustainability.

An alternative example of applying silvicultural practices is the Matang Mangrove Forest Reserve in Perak. The silvicultural system established in 1908 was based on retention of seed trees and cutting cycles of 20–40 years (Chong 2006). Two years after decomposition of the slash, the area is assessed for natural regeneration. Where seedlings are missing they are planted with wildings collected elsewhere. When trees have reached pole size (approximately 15 years) they receive a first thinning for achieving 1.3 m distances between remaining individuals, followed by additional thinnings after 20 and 25 years (2 and 2.6 m distances, respectively) (Jin-Eong 1995). Based on mortality data, Wooi Khoon and Jin Eong (1995) recommend to bring forward the thinnings in Matang for about 3 years (age 12/13 and 17/18 years instead of age 15 and 20) in order to avoid wastage. However, generally speaking, the non-interrupted and permanent management of the Matang Reserve, the application of 10-year working plans and proper implementation were the key activities for maintaining a forest cover of more than 97% during the last 100 years. Considering the enormous human pressure of alternative land uses such as aquaculture or agriculture, this is a success story. However, further research should look for economically feasible silvicultural techniques to avoid clear-felling systems, and thus contribute to higher ecological standards of sustainability. It is important to note that even this rather positive model can be called sustainable only in terms of stable land use, but not in terms of stable yields: While in the first rotation, yields of 299 tonnes ha^{-1} of greenwood (*Rhizophora apiculata*) were extracted, the yields declined in the following rotations to 158 and 136 tonnes ha^{-1} (Wooi Khoon and Jin Eong 1995).

A modification of the Matang system was applied in Thailand and Indonesia, where clear felling is carried out in strips of about 50 m perpendicular or 45° to the water line in only 15 years rotations (Jin-Eong 1995). He argues that the easy structure and the presence of monospecific stands justify clear-felling methods.

This may be true in some cases, but in cases where protective functions of the mangroves have high priority, silvicultural systems with continuous forest cover such as minimum cutting diameter systems and careful interventions should have priority. The decline of yields in the exemplary model Matang shows that clear felling systems should be used more carefully until all reasons for degradation of productive functions are scientifically understood. Jin-Eong (1995) shows at the example of Sabah and Sarawak, that it is possible to apply minimum cutting diameters for mangrove ecosystems, too.

20.2.4.3 Alternative Silvicultural Approaches

Wood cutting for fuel tends to focus on mid-size individuals because small trees are not worth the effort and large trees are too difficult to extract. In forests for sawwood in contrast, the larger individuals are harvested selectively (Alongi and de Carvalho 2008). Since tree diversity of mangroves is relatively low and the floristic composition is relatively uniform in comparison to other tropical ecosystems, the preconditions for selective systems are relatively good. Another advantage is the accessibility by boat and hence lower damages on the remaining stand resulting from extraction of logs. In the early twenteith century, selection systems with a fixed number of seed trees and rotation periods between 20 and 40 years were commonly applied in Malaysia. Early silvicultural thinnings result in better growth stimulation than later thinnings (Watson 1931 cit. Devoe and Cole 1998). Early thinnings provide stems with smaller dimensions for fuelwood and poles, while the harvesting cut at the end of the rotation provides saw-timber. The stand itself remains intact and thus maintains several environmental services.

The mangroves of the Sundarbans are managed under a selection-cum-improvement silvicultural system. All types of harvest for timber, fuelwood and pulpwood are carried out in a 20-year cycle (Hussain 1995). *Excoecaria agallocha* is exclusively used as pulpwood and matchwood. All trees above the exploitable diameter are harvested in a single operation. In the case of *Sonneratia apetala*, all trees that are more 30 cm in diameter are removed. However, in cases where *H. fomes*, *E. agallocha*, *Ceriops decandra* and other mangrove species have regenerated, all *Sonneratia apetala* trees are cut in order to assist the establishment of a more valuable crop (Hussain 1995). Because of its pioneering character, *S. apetala* seedlings do not establish under a mature crop of the same species. See also Chap. 22 for management of mangroves in the Sunderbans.

As modification of this system, buffer zones are maintained along coasts and river banks in Indonesia. Only 40 evenly distributed individuals ha^{-1} of *Rhizophora*, *Bruguiera* or *Ceriops* with DBH >20 cm in diameter are retained as seed trees, the rest is removed in a single cut at the end of a 30-year rotation. One single thinning at age 15 is applied (Hussain 1995).

Xylocarpus granatum, referred to as mangrove mahogany or rosewood and highly prized for its timber is a good example for seed tree methods. Despite its extraordinary dominance in the upper canopy, this species regenerates with relatively low seedling numbers of only 500 ha^{-1} (Ashton and Macintosh 2002). Some authors state that mangroves tend to regenerate close to their mother trees due to weak tidal movement and thus reduced dispersal of propagules (Ashton and Macintosh 2002; Van Spreybroeck 1992). For cases where successful regeneration depends on proximity of seed trees, management guidelines should quantify the number and distances of seed trees in logging areas for guaranteeing the establishment of future crop trees.

The ability of some species to coppice could also be interesting for testing coppice with standard systems, which are able to produce both fuelwood and timber very effectively. However, natural forest management systems are nowadays widely replaced by clear-felling systems (Chong 2006), especially for industrial production with a temporary loss of tree vegetation and many environmental services. This tendency towards clear-felling systems with short to medium rotation lengths may even increase when prizes for energy and oil substitutes rise. It is surprising how little scientifically based literature is available about the development of sustainable management systems for natural mangrove forests.

Numerous non-timber products, mainly for subsistence, are in use. For example, *Nypa fruticans* leaves are widely used in Southeast Asia for roofing, thatch in walls, floor mats. Mangroves are appreciated for their barks (providing tannins and dyes), wood fibre, animal fodder, medicines and habitat for bees and wildlife (Walters et al. 2008). A good example for a non-timber forest product (NTFP) which may be conflicting with management aims for timber or protective functions is water buffalo. Dahdouh-Guebas et al. (2006) approximate a carrying capacity of roughly 0.8–2.5 animals ha^{-1}. Browsing is species-specific with high preference to *Avicennia marina* over *Sonneratia apetala* and *Lumnitzera racemosa* with possibly negative effects on regeneration and future forest structures.

Coasts with high priority of protective functions require management methods with a continuous forest cover. Especially for these areas the use of NTFPs could provide added benefits to income from selective systems. However, an integrative approach is necessary to solve sometimes conflicting management objectives (Chap. 10).

20.2.5 Use or Conservation?

Mangroves have extraordinary importance for coastal protection. Hurricanes and inundations caused devasting damages in coastal areas in the last decades. Since IPCC predicted an increase for wind intensities of 5–10%, this importance may even rise in the next decades (McLeod and Salm 2006). Over half of the mangroves are located within only 25 km of urban centres with 100,000 inhabitants or more (McLeod and Salm 2006). Human pressure has led to a conversion of 30% mangrove area into coconut and oil palm plantations in Selangor, Malaysia. Rice planting, cattle and other uses may lead to similar rates in other regions (Chong 2006). Aquaculture for tiger prawns (*Penaus monodon*), white prawns (*P. merguiensis*), barramundi (*Lates calcalifer*) and mangrove snappers (*Lutjanus* sp.) provide much higher internal benefits than forestry. Thus, economical reasons frequently are behind the causes for deforestation. Even when forestry may be economically profitable, deforestation will continue as long as alternative land uses are more profitable.

The direct harvest of mangrove wood and products usually is related to subsistence need or is an important income supplement. Besides wood and NTFPs, the majority of people living in or near mangrove areas derive their principal income from fishing and related activities (Walters et al. 2008) and hence, depend on an intact mangrove ecosystem. Fishery and wood extraction are closely correlated in many cases. In South West Cameroon 63% of the wood extracted is used for smoking 91% of the fish landings. Thus, it can be concluded that sustainable land use and sustainable silviculture are interdependent. They depend on integrative solutions, considering fishery, NTFP uses and timber together. The role of payments for environmental services in this context will be discussed later.

20.3 Montane Forests

20.3.1 Definition and Area

Though there may be a common agreement of about "what is a mountain", numerous definitions for mountains exist using such different criteria as elevation, volume, relief, steepness, spacing or continuity. Kapos et al. (2004) state that the agreed components are relative altitude, absolute altitude and steep slopes, generating strong environmental gradients with high energetic variability and unstable environments. Additionally to heterogeneous environmental conditions, mountain areas are important refugia for migrating species in different time scales. Both, temporal and topographical heterogeneity have contributed that tropical montane forests are the terrestrial ecosystems with the highest biodiversity in vascular plants (Barthlott et al. 2007). A total of 24.3% of the total global land area are calculated to be mountains with a total forest area of 9.1×10^6 km^2. Mountain areas in the tropics comprise about 6.8% of the global land area. According to Kapos et al.

(2004), 55.5% of the global mountain areas are allotted to low elevations between 300 and 1,000 m a.s.l.

20.3.2 Ecological Factors

It is evident that one main ecological factor which influences the ecology of tropical montane forests is the enormous gradient of *temperature*, from hot tropical conditions at lower montane forests to cool temperate conditions at the tree line. However, many other environmental factors are closely linked to this altitudinal gradient (see Beck et al. 2008 for an example in Ecuador). The amount of *precipitation* usually increases and interception decreases with increasing altitude (Gomez-Peralta et al. 2008; Bendix et al. 2006). Total evapotranspiration decreases with increasing altitude, amounting from 155 to 1,380 mm year^{-1} for equatorial lower montane forest to 310–390 mm year^{-1} for upper montane cloud forest (Bruijnzel 2001). In ecosystems with high incidence of clouds, the canopies absorb precipitation from clouds which results as gain of water in contrast to interception which is usually considered as net loss of water for the ecosystem. Thus, as a result of interception losses and gains from clouds, Bruijnzel (2001) calculated an average relative net precipitation for lower montane forests not affected by clouds of 67–81%, in contrast to 80–101% for lower cloud forests and for upper montane cloud forests up to 81–179% in comparison to precipitation.

Decomposition in *soils* is generally considered to be slower than in lowland forests. The decomposition decreases with increasing altitude and from valleys to ridges which results in accumulation of organic matter. Kitayama and Aiba (2002) found *k* values of 1.7 at lower elevations to 0.4–0.9 at 3,100 m a.s.l. shoot/root ratio of tree biomass exhibits a fivefold increase along similar elevations in South Ecuador (Moser et al. 2008). The ratio of total organic matter in the soil to that in plants can rise from 0.2 in lowland forests to 4.0 in upper montane forests (Grubb 1977).

Montane forests receive lower amounts of *light* by sun flecks, but higher amounts of diffuse light (Grubb and Whitmore 1967) in comparison to humid low land forests. Leaf area index (LAI) usually decreases with increasing altitude from 2.9 to 5.6 at the lower montane level (700–1,050 m a.s.l.) to 1.6–2.9 at the upper montane level (3,060–3,100 m a.s.l.) (Gomez-Peralta et al. 2008; Kitayama and Aiba 2002; Moser et al. 2008). However, extreme high values of 7.7 LAI were reported for upper montane oak forests in Costa Rica (Hölscher et al. 2004).

20.3.3 Species and Diversity

Tropical montane areas are global hotspots for vascular plants (Mutke and Barthlott 2005; Barthlott et al. 2007). Richter (2009) and Richter et al. (2009) summarise several possible causes for the outstanding diversity in these ecosystems from global scale (paleo-ecological factors, IDH, rainfall-diversity hypothesis, plant– animal interactions and others) to site-specific reasons at the microscale (niche complexity, dynamic processes and others). Mountains are isolated ecosystems. Consequently they host high numbers of endemic species. For example in Sub-Saharan Africa, 80% of about 4,000 plant species of the Afromontane region are estimated to be endemic (Stuart et al. 1990). In Africa and Asia montane forests are the last refuges for a series of large mammals (*Gorilla beringei beringei* and others). They are also exclusive habitats for many seed dispersers as birds or bats. The diversity of tree species in montane forests is generally slightly lower than in lowland forests (Cao and Zhang 1997). While exceptional high tree diversity is reported for lowland forests (Valencia et al. 2004), several world records are registered for other prominent organism groups in montane ecosystems, in particular for bryophytes (Parolly et al. 2004) and geometrid moths (Brehm et al. 2005). Owing to high moisture and relative high light transmission, montane forests, especially cloud forests, are rich in epiphytes. For example, 300 species of vascular epiphytes were recorded for a 4-ha sample plot of in Monteverde, Costa Rica (Ingram and Nadkarni 1993). The total epiphyte biomass in oak forests of Costa Rica amounts 2,600 kg ha^{-1} (Hölscher et al. 2004). Single trees can be hosts for several hundreds of epiphyte species ranging from lichens, mosses, bromeliads to orchids. Especially mosses and tank bromeliads are considered to have high water storage capacities and release water slowly (Bruijnzel 2001). The so-called "cloud forest" belt, which is characterised by a prevalent cover of cloud or mist, usually occurs between 1,500 and 2,500 m a.s.l. at the lower limit and 2,400–3,300 m at the upper level (Bruijnzel and Veneklaas 1998). Due to the "Massenerhebung" effect the altitudinal level of these belts is usually correlated with the height and volume of the respective mountain range.

Characteristic tree families of montane areas comprise, e.g. Lauraceae and Fagaceae for the lower levels, Symplocaceae, Cyatheaceae and Myrsinaceae for intermediate levels and Ericaceae for the upper altitudes (Ohsawa 1991). Further important families are Podocarpaceae and Melastomataceae (Sri-Ngernyuang et al. 2003; Homeier 2004). Lauraceae are among the most species-rich families and comprise 18% of the basal area in a forest in Thailand (Sri-Ngernyuang et al. 2003). In the Neotropics many commercial species can be found in the genera: *Juglans*, *Cedrela*, *Podocarpus*, *Nectandra*, *Ocotea* (La Torre-Cuadros et al. 2007; Pinazo and Gasparri 2003; Günter et al. 2008). Topography has a very strong impact on the association of plant species (Sri-Ngernyuang et al. 2003; Homeier et al. 2010) and may play a higher role for building of ecological niches than in lowland forests.

20.3.4 Structure and Dynamics

20.3.4.1 Structure

Forest structures and regeneration patterns are very variable along altitudinal and latitudinal gradients. While at lower altitudes species with different diameter

Table 20.2 Comparison of structural data (ranges for minima and maxima) of mangroves and montane forests

	Mangroves	Montane forests
N ha^{-1} (>2.5 cm)	790–3,000 (1, 20[a], 21[a])	
N ha^{-1}(>10 cm)		400–1,400 (9, 10, 13)
N ha^{-1} (>15 cm)	165 (1)	
Above ground biomass (tonnes)	77–384 (2, 3, 19)	100–550 (6, 11)
Basal areas (m^2 ha^{-1})	9.3–49 (4, 19, 21)	8.7–56.7 (12, 13)
Diameter increment (cm $year^{-1}$)	0.05–0.64 (1)	0.1–12.3 (13, 16)
NPP (tonnes ha $year^{-1}$)	1–56 (5, 19, 20)	1–18 (6)
LAI	0.3–7.4 (17, 18, 19, 20)	1.6–7.7 (6, 7)
Min–max height	1–45 (14, 15, 19)	2–45 (8)

Note that main differences between the ecosystems could be revealed for NPP and tree densities. Authors are indicated in brackets

[a]No diameter limits were indicated

(1) Iftekhar and Saenger (2008); (2) Alongi and de Carvalho (2008); (3) Alongi (2002); (4) Ashton and Macintosh (2002); (5) Amarasinghe and Balasubramaniam (1992); (6) Kitayama and Aiba (2002); (7) Hölscher et al. (2004); (8) Grubb (1977); (9) Crome et al. (1992); (10) Günter and Mosandl (2003); (11) Moser et al. (2008); (12) Bellingham and Tanner (2000); (13) Homeier (2004); (14) Iftekhar and Saenger (2008); (15) Cintrón et al. (1978); (16) Herwitz and Young (1994); (17) Kovacs et al. (2009); (18) Lovelock (2008); (19) Sherman et al. (2003); (20) Clough et al. (1997); (21) Satyanarayana et al. (2010)

distributions (from inverse-J, to sporadic and emergent type distribution) can become dominant, only the inverse-J type tends to be predominant and upper altitudes. Ohsawa (1991) attributes altitudinal zonation in the tropics to vegetation units which are controlled by different thermal determinants, i.e. the −1°C-isotherm and the annual temperature sum of 85°C.

It is well established that tree heights decrease with increasing altitude. Tree heights reach from 45 m in lower montane forests to 2 m at the subalpine level (Grubb 1977). However, depending on region and latitude the absolute values can vary considerably. Wilcke et al. (2008) revealed slower growth rates at upper altitudes due to different soil properties, and increasing C:N, C:P, and C:S ratios. Also exceeding rainfall, radiation and wind may limit tree growth at high elevations (Peters 2009). The maximum dimensions of trees are usually smaller than in lowland forests and decrease with increasing altitude. The stem densities in turn can reach very high values of more than 1,000 ha^{-1} especially along ridges (Table 20.2).

20.3.4.2 Regeneration

Similar to in lowland ecosystems, regeneration is strongly influenced by disturbance regimes and gap-phase dynamics. For example, *Chusquea* spp. (bamboo) are abundant indicators for disturbance in neotropical montane forests. Some authors state that *Chusquea* in gaps outcompetes pioneer tree species like *Cecropia*, *Alchornea* spp., *Tibouchina* (Tabarelli and Mantovi 2000). However, they not always necessarily suppress regeneration of tree vegetation (Young 1991). Besides gap dynamics and natural disturbances which are typically considered as major trigger for plant species composition in forest ecosystem (Brokaw 1985), in montane humid forests, plant life benefits additionally from extreme heterogeneity of climatic and edaphic site conditions. Consequently, disturbances in montane forests are considered only as one of many driving forces for outstanding biodiversity and triggering factor for pathways of natural regeneration (Richter 2009).

Topography plays an important role in montane forest ecology (Oesker et al. 2008). However, it is not possible to generalise the influence of topography on local ecological factor combinations. While Bellingham and Tanner (2000) found that recruitment processes and growth did not differ between ridges and slopes in Jamaica, Herwitz and Young (1994) found higher growth and turnover rates on ridges of Australia. Homeier et al. (2010) and Oesker et al. (2008) in contrast found higher growth rates in valleys in comparison to ridges in Ecuador. Reasons behind these different patterns may be different soil properties. For example, while water logging in lower topographical positions may be a limiting factor for dipterocarp forests in Sri Lanka (Chap. 13), nutrient deficiencies as limiting factor could be revealed for upper topographical positions in Ecuador (Homeier et al. 2010).

High light and high litter accumulation seem to be universally good cues for moderation of temperature and moisture extremes, and for regeneration despite for species with small seeds like *Cecropia* spp. which cannot establish in high litter environments (Everham et al. 1996). However, the depth of the litter layer may be extremely variable, due to high topographical variability and influence of landslides or fires.

20.3.4.3 Growth

Growth, in general, is slow in montane forests with a continuous decline from low- to uplands. However, reflecting the extraordinary heterogeneity of site conditions in mountain areas, growth values are extremely variable. Bellingham and Tanner (2000) revealed that best predictors for tree growth were soil pH and percentage of basal area damaged by hurricanes, indicating that both soil conditions and light competition are limiting factors for growth in montane forests of Jamaica.

Tree biomass decreases with increasing altitude. Kitayama and Aiba (2002) and Moser et al. (2008) found pronounced differences: 280–550 tonnes ha^{-1} at lower altitudinal levels (700–1,050 m a.s.l.) vs. 100–308 tonnes ha^{-1} for upper levels (2,400–2,700 m a.s.l.). Basal areas frequently increase with increasing altitude up to a certain maximum, with strong declines at high elevations (Moser et al. 2008; Bellingham and Tanner 2000). However, it is notable that very high values of >60 m^2 ha^{-1} can frequently be attributed to small sampling areas.

NPP production is a linear function of altitude (Hazarika et al. 2005; Kitayama and Aiba 2002) with values reaching lowland levels at lower montane altitudes (for example, approximately 18 tonnes ha^{-1} year^{-1}) at Mt. Kinabalu, (Kitayama and

Aiba 2002), intermediate values at 2,000 m a.s.l. to almost zero at the upper tree line. But NPP is also influenced by precipitation: While sites with 2,000 mm precipitation produced >10 tonnes ha^{-1} of annual NPP, those with 5,000 mm precipitation produced only approximately 4 tonnes ha^{-1}. This can either be attributed to higher cloudiness and hence lower available light for photosynthesis (Bruijnzel and Veneklaas 1998) or to low redox potentials affecting N and P cycling (Schuur and Matson 2001). Consequently, at sites with high precipitation higher yield for trees should be expected at microsites with reduced waterlogging. In mature forests, NPP consists to more than 80% to investment in leaves (Schuur and Matson 2001). Thus, investment in growth is higher at earlier successional stages. Many authors state that below-ground biomass investment increases with increasing altitude (Bruijnzel and Veneklaas 1998; Moser et al. 2008).

20.3.5 Wood Production and Silviculture

Several studies show the close relationship between population and deforestation in mountain areas (Turner et al. 1993; Kaimowitz and Angelsen 1998; Mosandl et al. 2008). However, very few studies present economically feasible possibilities for avoiding deforestation. As stated in previous sections, tree growth is much slower than in lowland forests and final dimensions usually are smaller, so that preconditions for sustainable management for timber is rather unfavourable in comparison to lowland forests.

Forest operation techniques in montane areas are much more expensive and complex than in the lowland forests. Accessibility is the most critical factor influencing feasibility of operations in mountainous terrain, regarding remoteness and steepness (Heinimann 2004). Whereas cable-based techniques or even flight paths are applied in industrialised countries, animal-based or bio-mechanical extraction techniques have to be applied in developing countries. The latter ones usually imply timber losses due to manual preparation of logs in situ by chain saws. For economically feasible timber extraction, the higher costs in montane areas could be compensated by either higher economic value or higher densities of timber species in relation to lowland forests. Unfortunately, many accessible forests already have been exploited and remaining timber species exhibit relatively small dimensions (Günter and Mosandl 2003; Günter et al. 2008).

Two major silvicultural systems are applied in the tropics: shelterwood systems (monocyclic systems, resulting in relatively uniform stands with reference to species composition and structure, but higher average yields) and selective systems (polycyclic, resulting in species rich and strongly structured stands, but usually accompanied by high logging damages and lower yields). Montane forests containing many shade-tolerant timber species, e.g. in South America from the families Lauraceae, Podocarpaceae or others could consequently be managed by selective systems. As loss of vegetation cover in montane ecosystem will drastically increase the susceptibility to degradation, shelterwood systems must be considered as rather critical. Likewise, the key question for avoiding degradation in selective systems is, up to which intensity can interventions still be considered as reversible within defined operational time scales?

In Boxes 1 and 2, I present two examples from different continents and from opposite positions along the altitudinal gradient: The Queensland system in Australia (600–1,000 m) and improvement fellings in Southern Ecuador (2,000 m a.s.l.). Both experiments distinguish between ridge forests and gullies (respectively, gorges). While the Ecuador example represents relatively difficult silvicultural preconditions (high altitudinal level, previous overexploitation, low growth rates, socio economic conditions of a developing country), the Queensland example represents better preconditions in all aspects.

Box 1: Queensland System

In Queensland, commercial stems and higher basal areas are concentrated on ridges. However, this can be attributed rather to higher stem densities for smaller individuals (smaller than 60 cm DBH) than to larger individual dimensions. Logging reduced the average basal areas on ridges by 25% and in gullies by 17.1%. Only 42.4% of these values were attributed directly to harvested trees, the rest were felling damages (Crome et al. 1992). The felling activities lead to a reduction of canopy cover from 0 to 13% on ridges and from 0 to 15% in gullies. Logging tracks (4.2% of logging area) and skidder trails (0.8%) caused further disturbances. The authors suppose that about half of the regeneration was lost on ridges and fewer in gullies. Eighteen months after intervention, casual observations showed a dense growth of the Stinging Tree and rattans in larger gaps. In this period, no loss of any tree species from the sample sites could be detected. Erosion was not quantified or measured but apparently channelling was already noticeable. Thus, detailed planning of logging and extraction operations is essential for reducing of environmental damages in mountain areas. North Queensland's forestry can be considered as advanced in comparison to most other countries in the tropics. Vanclay et al. (1991) cite several studies from the 1980s, where effects on fauna, flora, hydrology, soils and timber production have been studied intensively and did not reveal indication that harvesting is not sustainable. A key element in the Queensland system is the creation of mosaics of logged and unlogged areas for mitigating logging effects. Besides the high operational costs, applying well-trained foresters and expensive machinery, the Queensland system is applied profitably for industrial proposes for more than four decades. However, it is notable that even in this well-designed selective system, many species are used in the second rotation which were considered as not desirable in the first cut or from areas which were missed in the first cut or which were not accessible (Vanclay et al. 1991). This is a strong indicator for non-sustainability of the system, because more valuable species were possibly overused in the first rotation.

Box 2: Experimental Improvement Fellings in Ecuador

Thirteen hectares of permanent plots have been established in 2003 in a tropical montane rain forest of the Reserva Biologica San Francisco, South Ecuador at an altitude of approximately 2,000 m a.s.l. Experimental improvement fellings in these permanent plots showed that felling intensities of 32 dominant or co-dominant trees per hectare (approximately 10% basal area) can be considered as sustainable with respect to nutrient cycles. It is notable that ridges in Ecuador were nutrient limited and hence showed lower heights, basal areas and growth rates, but higher stem densities than in valleys in contrast to the Queensland example (Günter and Mosandl 2003; Homeier 2004). Due to limited growth rates on ridges, logging was restricted to valleys only. Degradation risks decreases with soil quality and increases with slope angle (Schulte 1998). Given the extremely high steepness (up to >45°) and the poor soil quality in the Ecuador case study, it is surprising that no changes in nutrients fluxes on the catchment level could be revealed. As already mentioned, the eastern slopes of the Ecuadorian Andes are considered as an important hotspot biodiversity regarding epiphytes and insects. Thus, it is surprising that there were no changes in alpha diversity of moths and epiphytes, only changes in beta-diversity of moths in the first year after intervention (Günter et al. 2008, Wilcke et al. 2009). However, the mortality of the remnant stand tends to be higher in logged areas in relation to reference areas, possibly due to reduced resilience and higher susceptibility for windthrow.

The key elements for these promising results are (a) the fact that no extraction of the stems was carried out and all effects are attributed to felling only, (b) relatively small crowns in comparison to lowland forests, with (c) correspondingly slight damages and slight changes in light and microclimate, and (d) large buffer zones around the logging areas for buffering of diversity and water cycles. With US \$31 ha^{-1} year^{-1}, the annualised net revenues of this forest is very low (Chap. 11) in comparison to dipterocarp forests of Asia (see Chap. 13). The most important timber species for a potential silvicultural system is *Cedrela montana* with acceptable growth rates for this altitude (3–4 mm diameter growth per year) and excellent timber qualities. Therefore, domestication efforts for this species are highly recommended (Chap. 27).

Of course, these two case studies may not be representative for all montane areas worldwide. However, some general aspects can be demonstrated: Both cases show that ecologically sustainable management can be possible even in steep areas, with high susceptibility to erosion and biodiversity degradation. Both experiments applied careful felling/logging intensities of approximately 10–15% basal areas and were working with buffer zones around the intervention areas, which may be the key elements for these promising results. Hence, as general recommendation, and for montane areas in particular: any intervention in a given area should be surrounded by non-intervention buffer zones within the same catchment, preferably several times larger than the logged area, for mitigating nutrient and biodiversity losses. Especially in steep areas, careful extraction techniques should be applied such as winching, cable techniques, chutes or animal extraction. While cable techniques and chutes require higher density of commercial stems, winching and animal extraction are feasible also for lower densities. However, there is evidence that economic aspects apparently are major problems for both cases.

Since NPP, in general, is lower in montane forests than in lowland forests and steepness and inaccessibility of the landscape is higher, the management of NTFPs in montane forests can be of higher importance than in lowland forests and could possibly contribute for solving the economic problems. A good example for NTFP is the extraction of *Aquilaria* spp. for productions of aromatics in Indonesia (Paoli et al. 2001). The proportion of trees with presence of the corresponding fungi (resulting in the appreciated gaharu wood) was higher in the upper montane forests (73%) than in the lowland forests (27%). The average pre-harvesting density was very low (0.32 ha^{-1}). Thus, the net revenue for sustainable production amounts only \$10.83 ha^{-1}. A value which is much lower than (presumably unsustainable) commercial logging in the same area. However, it is a highly profitable activity on a per time basis, reaching almost the triple value of local wages. A further important NTFP product of worldwide importance is coffee. In the last century this crop moved consecutively more and more from production under shade within the forests to the higher productive coffee systems outside the forests. Similarly to a multitude of NTFPs such as honey, *Annona* spp. and others, the classical coffee production under shade could become more attractive in the future if consumers would compensate biodiversity losses by accepting higher prices or if coffee cultivators could benefit from carbon markets.

Growth acceleration by improvement fellings, or combining fast-growing long-living pioneers with short rotations together with slow growers such as many Lauraceae and Podocarpaceae managed in longer rotations may be further options for increasing annual net revenues, helping to reduce the attractivity of alternative land-use options (Marin Velez 1998; Homeier 2004; Günter et al. 2008). But even under best application of all silvicultural options, natural forest management may not always be competitive with (ecologically unsustainable) alternative land uses such as cattle raising, in particular, for forests with low volume growth or timber value. In these cases, silvicultural options may meet their limits.

Besides silvicultural options, further promising tools are proposed to contribute to sustainable forest management, i.e. payments for environmental services, REDD plus, risk compensation by land-use diversification or a combination of several aspects (Knoke et al. 2009a, b; Chap. 11; Redford and Adams 2009). These aspects will be discussed in the next section.

20.4 Ecosystem Services Versus Logging in Ecosystems at Ecological Margins?

The importance of tropical montane forests and cloud forests for protection of water sheds against erosion and providing fresh water for human settlements and irrigation is generally recognised (Hostettler 2002; Doumenge et al. 1993). The water-retaining capacity of these forests is of extraordinary importance not only for the local population and their land-use systems, but also for the people of lower altitudinal belts whose existence frequently depends on careful management of the water resources in the mountain areas. Ecosystem products such as timber, charcoal, NTFPs or agricultural products from anthropogenic replacement systems such as cattle from pastures contribute directly to the livelihood of local land users. In contrast, environmental services like water, carbon sequestration, biodiversity, protection from erosion, recreation and education are benefits for societies which are located remotely from mountains, sometimes many thousands of kilometres away. Local land users only share these benefits to a very small percentage. This is a classical dilemma addressed by the Convention on Biological Diversity and the efforts to access and benefit sharing (see Chap. 4).

Similarly to montane forests, the whole economic value of mangroves for fishery including all support functions are considered to be enourmous. Rönnbäck (1999), for example, estimated a value of about US \$1,000–10,000 ha^{-1} $year^{-1}$ in developing countries. Mangroves also provide special and valuable environmental services which cannot be provided by other ecosystems. Mangroves are effective sewage-filters for peri-urban coastal regions. According to Walters et al. (2008) this function is worth US \$1,200–5,800 ha^{-1} $year^{-1}$. The coastal protection services from storms, coastal erosion and floodings are estimated to value US \$1,600–4,700 ha^{-1} $year^{-1}$ (Kairo et al. 2009; Costanza et al 1989; Sathiratai and Barbier 2001; Alongi 2002). Thus, the complete value of environmental services according to these authors reaches from approximately US \$4,000–20,000 ha^{-1} $year^{-1}$ and exceeds by far the direct annual return from timber and non-timber products. For example, charcoal and poles only contributed to US \$344 ha^{-1} $year^{-1}$ or 1.7% of the estimated total economic value in a Malaysian mangrove ecosystem (Chong 2006). Other authors estimate the annual income from forestry at about US \$150–379 (Kairo et al. 2009; Alongi 2002). In montane forests the annualised net revenues tend to be even lower due slow growth rates, and high tree diversity accompanied by low abundances of timber species (Knoke et al. 2009a, b).

A key problem for economists is how to convert these values into income for land users. As long as no prizes are paid for environmental values, even small prizes for timber or agricultural products may lead to degradation and deforestation. Gunawardena and Rowan (2005) show at an example from Sri Lanka that the internal benefits of shrimp farms are about 1.5 times greater than the internal costs, thus this project is profitable in terms of microeconomics. However, the external costs, or the costs to the society, are 6–11 times greater than the benefits. Thus, besides its profitability this project can be considered catastrophic for the society. Nevertheless, aquaculture expansion is the major cause for conversion of mangroves, followed by lumber and wood chip industries (Walters et al. 2008). Despite their obvious value for the societies, mangroves are among the most threatened global ecosystems.

Silviculture for maximisation of timber value does not match automatically with silviculture for maximisation of environmental services (for example, carbon sequestration). On a higher spatial scale, land-use management for maximisation of environmental services does not match automatically with demand of local populations for higher income. Risk compensation by land-use diversification can increase the proportion of forests in a land-use portfolio. Additionally, especially in cases of conflicting aims for landscape management in general and forest management in particular, payments for environmental services can partly provide compensation and thus, increase the attractivity for the conservation of natural ecosystems instead of conversion into more profitable landscapes. However, financial measures alone, without considering the potential of silviculture and added benefit approaches may be too expensive for being practicable. Forest management and silviculture in the future have to integrate the conflicting aims by integrative approaches, including NTFPs and timber production inside the forests, but also integrative land-use portfolios, risk management and financial measures such as PES or REDD plus or carbon markets outside the forests (Lamb et al. 2005). This is of particular importance for forests at ecological margins, i.e. when productivity declines and importance of environmental services increases.

It seems paradox that careful and appropriate site-specific selection systems which could equilibrate the demands of local population for wood products and the multiple ecosystem services for the global population are disappearing (mangroves) or have only poorly been developed (montane forests). This may be attributed to missing knowledge about ecology, and more profitable silvicultural systems on the one hand and to missing financial instruments providing economic benefits from sustainable management directly to the land user on the other hand.

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Chapter 21
Natural Regeneration and Management of *Podocarpus falcatus* (Thunb.) Mirb. in the Afromontane Forests of Ethiopia

Demel Teketay

Abstract *Podocarpus falcatus* is one of the two coniferous species naturally growing up to 45 m high and 250 cm in diameter in 11 out of the 14 floral regions recognized in Ethiopia. It is a multipurpose species with a wider range of socio-economic and environmental importance. The species provides high-class softwood considered superior to European deals. Suitably manufactured and conditioned, it produces wood suitable for many purposes. It is also useful for fuelwood, charcoal, poles, paper pulp, shade, and ornamental purposes. One of the most promising products from the trees is the oil extracted from its seeds, which is edible and used medicinally to treat gonorrhea. The trees also serve as parts of the habitat of various organisms. Despite its great importance, the species is on the verge of local extermination because of its unsustainable exploitation over the last several decades. Even after it has been banned from harvesting/cutting, its illegal exploitation continues unabated. Unfortunately, owing to a number of factors, there are neither large-scale plantations nor future planned plantation establishment programs of the species. This implies that urgent actions are required to address its unsustainable exploitation. The seed germination, seed and seedling banks, seedling survival and growth, regeneration along altitudinal, light, and moisture gradients as well as in some selected Afromontane forests and timber harvesting of the species are discussed. Human impact on the remaining populations of the species is described and recommendations to address the unsustainable exploitation are proposed.

Keywords Afromontane forests · Environmental gradients · Human impact · Natural regeneration · Seed and seedling banks · Volume and yield

Plant nomenclature follows that of Friis (1992). Readers are referred to the same publication for taxonomic details.

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21.1 Introduction

Podocarpus falcatus (Thunb.) Mirb. (referred to as *Podocarpus* hereafter) belongs to the family Podocarpaceae and is one of the two coniferous species in Ethiopia. The simple fact that *Podocarpus falcatus* (Thunb.) Mirb. (hereafter referred to as *Podocarpus*) has 15 different names[1] in Ethiopian languages, three in English (East African Yellow Wood, Podo, and Yellow Wood) and nine more in other languages of east Africa, tells us a lot about its wide distribution, and speaks volumes about its usefulness (Teketay 1994). The species provides high-class softwood considered superior to European deals. Suitably manufactured and conditioned, it produces a joinery wood suitable for many purposes, such as shop and counter fittings, display cabinets, drawer linings (because of its smoothness), and for handicraft work. The wood is also suitable for the manufacture of other furniture, bakery boards, and confectionery trays for which nontainting is required; also for cupboards, shelving or fittings where a bright, clean-colored wood is desirable. The trees also serve as parts of the habitat of various organisms. *Podocarpus* serves as the only available food source at times of food scarcity to frugivore animals in the forest since trees can be found with seeds all the year round.

One of the most promising products from the trees is the oil extracted from its seeds (Teketay 1994), which is edible and used medicinally to treat gonorrhea (von Breitenbach 1963). A traditional method of oil extraction from seeds in the Chercher Highlands of southeastern Ethiopia is reported elsewhere (Teketay 1994). The local people prefer oil obtained from *Podocarpus* trees to the ordinary oil they purchase from shops. The cake, which results from the extraction of oil from the seeds, has a potential to be used as a source of animal feed directly or mixed with other feed items.

The aim of this article is to present a review of available information on natural regeneration and management of *Podocarpus* in Afromontane forests of Ethiopia.

21.2 Habitat and Geographical Distribution

Podocarpus occurs occasionally in Afromontane rainforest, but is particularly characteristic of undifferentiated Afromontane forest (Friis 1992) or dry Afromontane forest (Bekele 1994; Teketay 1996), where it is frequently observed as one of the dominant species ("*Podocarpus* forest") or one of the co-dominant species (e.g. in "*Juniperus-Podocarpus* forest"), often persisting in relic forest patches (e.g. in gully and church forests). It is frequently found as a single tree left in derived

[1]Zegba/ZigbaZigba (Amarigna), Zegba/Zigba (Guragigna), Digiba (Hadiigna), Chido, Dijido (Kafanono), Zegba (Kambataigna), Elta (Afan Konso), Birbirsa/Birbirso/Birbissa, Danicho, Mekersa (Oromiffa/Afaan Oromoo), Dagucho (Gumuz), Decho, Degucho (Afan Sidama) and Ziga (Wolitigna); East African Yellow-Wood, Podo, Yellow Wood (Eng); Podocarpo (Ita) (Teketay et al. 2010).

grassland or farmland in areas with sufficient rainfall (Friis 1992). In areas covered by the Flora of Tropical East Africa, it is reported to occur in "upland rain forest" in the altitudinal range of 1,550–2,800 m with rainfall range from 1,000 to 1,500 (−2,000) mm year^{-1}. Its geographical distribution in Ethiopia[2] includes AR, BA, GJ, GD, HA, KF, SU, SD, TU, WG, and WU (Teketay et al. 2010). The species is also reported to grow naturally in Burundi, Democratic Republic of Congo, Kenya, Malawi, Mozambique, Rwanda, South Africa, Tanzania, and Uganda. It is considered as Afromontane Near-Endemic (Friis 1992).

21.3 Natural Regeneration

A good understanding of natural regeneration in any plant community requires information on phenology, seed production (quantity and quality of seed rain) and dispersal, the presence and absence of persistent soil seed banks or seedling banks, longevity of seeds in the soil, losses of seeds to predation and deterioration, triggers for germination of seeds in the soil, and sources of regrowth after disturbances. Tropical forest plants regenerate from one or more pathways, i.e. (1) seed rain: recently dispersed seeds; (2) soil seed bank: dormant seeds in the soil; (3) seedling bank: established, suppressed seedlings in the understory; and (4) advance regeneration or coppice: root or shoot sprouts of damaged individuals (Garwood 1989; Teketay 2005).

21.3.1 Seed Dispersal

Seeds of *Podocarpus* are dispersed by different wild animals. The epimatium that covers the seeds is eaten and the seeds dispersed by birds, baboons, colobus monkeys, apes, and probably other animals. Clean seeds and heavily chewed epimatium of the fruits were found under big trees of different species at Gara Ades Afromontane forest, southeastern Ethiopia, which was considered as a sign of bat dispersal (Teketay 1996).

21.3.2 Seed Germination

In the laboratory, the seeds *of Podocarpus* did not respond to other scarification treatments except mechanical scarification, i.e. removal of the woody seed coat.

[2]Floral Regions of Ethiopia: AR: Arsi region; BA: Bale region; GJ: Gojam region; GD: Gonder region; HA: Harerge region; KF: Kefa region; SU: Shewa region, above and to the west of the 1,000 m contour; SD: Sidamo region; TU: Tigray region, above and to the west of the 1,000 m contour; WG: Welega region; WU: Welo region, above and to the west of the 1,000 m contour (Hedberg and Edwards 1989).

Mechanical scarification increased percent germination of seeds (50%) significantly over the controls (nil) (Teketay and Granström 1997a). Similarly, over 90% germination of seeds was reported from mechanically scarified and surface-sterilized seeds incubated on sterile agar (Negash 1992, 1993). Evidently, the hard seed coat restricts germination in the fresh seeds, but there is no disturbance-triggered germination mechanism as in some other species, e.g. *Croton macrostachys* Del. (Teketay and Granström 1997b). As stated above, in the forest, seeds of *Podocarpus* are dispersed by a variety of birds, bats, colobus monkeys, and baboons, which clean the seeds and drop them on the forest floor. Emergence of seedlings is likely to occur over a long period, although the seed handling might possibly enhance germination, if the seed coat is affected.

21.3.3 Soil Seed and Seedling Banks

A few studies carried out so far (e.g. Teketay and Granström 1995; Teketay 1997a, 1998, 2005; Senbeta and Teketay 2002; Wassie and Teketay 2006; Lemenih and Teketay 2006) have shown that *Podocarpus* does not accumulate persistent soil seed banks in forests. This is associated with the absence of dormancy in the seeds, which is required to maintain the viability of seeds during their stay in the soil.

Following the germination of seeds, the next phase in the process of regeneration by seeds is the development of seedlings. Vulnerability of seedlings to hazards from environmental and biotic factors is higher at the early stages of seedling establishment (Fenner 1987; Whitmore 1996). One of the most effective adaptations for ensuring successful seedling establishment is the possession of a large seed, as in *Podocarpus*, which provides an ample reserve of nutrients during the period immediately after germination (Fenner 1985; Foster 1986). Climax species typically have bigger seeds, than pioneer species, which germinate immediately or within a few days after dispersal under forest canopy (Swaine and Whitmore 1988; Whitmore 1989, 1990). By immediate germination they escape seed-eaters and also degradation of their usually fatty storage tissues by micro-organisms and form a seedling bank on the forest floor in contrast to the soil seed bank of pioneers (Whitmore 1990).

Most of the seeds of *Podocarpus* are programmed to germinate readily soon after they are dispersed to form a carpet of seedlings on the forest floor, the "seedling bank" (Teketay 1997b). Hence, the formation of seedling banks under the forest canopy is the major natural regeneration route. Therefore, excessive exploitation of species, which is discussed in detail below, or clearing and conversion of the forest areas into permanent cultivation, which is the most common practice in Ethiopia (Teketay 1997a), will prevent regeneration of *Podocarpus* and other woody species. The destruction of seedlings coupled with poor long-distance dispersal and the lack of persistent soil seed banks may prevent the regeneration of *Podocarpus*. Artificially, *Podocarpus* can be propagated by seed seedlings, wildings, and vegetatively (Teketay 1996, 2005; Semagn and Negash 1996).

21.3.4 Seedling Survival and Growth

The survival and growth of naturally regenerated seedlings of *Podocarpus* were monitored for 3 years at Gara Ades forest (9°20′N and 38°35′E) under dense canopy (shade) and in open areas (gap) between 1992 and 1995 (Teketay 1997b). The population of seedlings declined progressively during 3 years of investigation both under shade and in gaps. The seedlings survived better under shade than in gaps. During the first inspection (1993), mortality of seedlings was low both under shade and in gaps. In the following 2 years, annual mortality was higher both under canopy and in gaps. The highest annual mortality was recorded in 1994 from seedlings growing in gaps. A few of the naturally regenerated seedlings disappeared and the cause of their disappearance could not be ascertained. In general, seedlings were damaged by grazing/trampling animals and probably people. On the other hand, several planted seedlings died as a result of drought during the long dry seasons. In Harenna forest, southeastern Ethiopia (6°40′ to 7°10′N and 39°30′ to 40°E), seedling mortality accounted for 20% of the total population investigated. The uprooted and missing seedlings contributed 40 and 60%, respectively, to the total seedling mortality recorded, suggesting that physical damage of seedlings by animals and human beings are the major contributors to seedling mortality (Tesfaye and Teketay 2005a). However, unlike other species, herbivory did not affect seedlings of *Podocarpus* (Tesfaye et al. 2002).

Mean annual height increment (MAHI) of the naturally regenerated seedlings was generally low (<15 cm). The seedlings growing in gaps had higher mean annual height than those growing under shade. The proportion of seedlings with positive height increments was much higher than those with no or reduced height increments. The reduction in height increment resulted mainly from damage caused by browsing animals and people.

21.3.5 Regeneration Along Environmental Gradients

21.3.5.1 Altitude

The distribution of seedlings of *Podocarpus* in Harenna forest exhibited marked restriction and distinct habitat preferences of seedlings (height < 150 cm) along the altitudinal gradient occurring between 1,500 and 2,100 m (Tesfaye and Teketay 2005a). Seedlings exhibited continuous occurrence along the altitudinal gradient except at 1,500 and 2,100 where a sharp decline in the densities of seedlings was observed. The saplings (height = 150–300 cm) occupied similar altitudes to that of the seedlings while trees (height > 300 cm) occupied altitudes from 1,500 to 2,000 m. The densities of seedlings recorded varied widely along the altitudinal gradient. The highest density, 3,050 seedlings ha^{-1}, was recorded at 1,800 m, and the lowest density, 750 seedlings ha^{-1}, was recorded at 2,100 m. Mean density of seedlings in the forest was 1,971 individuals ha^{-1}. The mean densities of saplings and trees in the forest were 32 and 67 stems ha^{-1} (range 8–75 and 42–117 stems ha^{-1}), respectively. Tree density increased with increasing altitude from 1,500 to 1,900 m and declined thereafter with no record of mature trees at 2,100 m.

Generally, at the lower elevations, i.e. 1,500–1,600 m, the seedling densities of *Podocarpus* were low, which could be attributed to human activity such as ground clearance, canopy thinning, and weeding associated with growing coffee plants or establishment of coffee farms. The relatively higher seedling populations, indicative of good recruitment, at the middle elevations, i.e. 1,650–1,850 m, could be attributed to reduced human activities and dominance of mature trees of the species. Compared with other dry Afromontane forests, *Podocarpus* had relatively higher seedling density in Harenna forest. This suggests suitability of the ecology of Harenna forest for *Podocarpus* to exhibit its potential for higher seedling recruitment in particular and healthy regeneration in general.

In the Munessa-Shashemene forest (7°13′N and 38°37′E), seedling densities varied considerably with altitude ranging between 43 and 503 individuals ha^{-1} at 2,300 and 2,600 m, respectively (Tesfaye et al. 2010). The mean density of saplings and trees was 102 individuals ha^{-1} of which saplings accounted for 64%. The densities of saplings and trees varied considerably across the altitudes and ranged between 43 and 198 individuals ha^{-1}. The highest density was found at 2,600 m and the lowest at both 2,300 and 2,700 m.

In Gara Ades (9° 20′ N and 38° 35′ E) and Menagesha (9°00′N and 38°35′E) forests, the seedlings dominated at the lower altitudes, i.e. 2,400 and 2,350 m, respectively (Teketay 1996, 1997b).

21.3.5.2 Light

Availability of understory light is both a cause and an effect of forest dynamics since light is a major environmental factor limiting survival and growth of many forest species (Whitmore 1996). The results of light levels measured in Harenna were (41 ± 15 and and 1,885 ± 20 μmole m^2 s^{-1} under canopy shade and in open areas/gaps, respectively (Tesfaye and Teketay 2005a). The forest zone at the upper elevation was more open than the forest zone at the lower elevation. Of the total 52 quadrats surveyed for seedling populations, 70% were found under canopy shade and the remaining 30% in the open. Similarly, about 79% of the investigated seedling populations were found under shade and 21% in the open. The fact that large quantities of seedlings were recorded under canopy shade suggests that *Podocarpus* is shade-tolerant. Similar results were reported from other works on the species (Teketay 1996, 1997a, b; Teketay and Granström 1997b; Fetene and Feleke 2001).

21.3.5.3 Soil Moisture

It has been long established that tree species differ in their requirements and range of tolerance to soil moisture in tropical forests (Swaine 1999), and correlations between soil moisture conditions and abundance of tree species had been demonstrated in tropical forests (Newbery and Gartlan 1996). The distribution of *Podocarpus* along the soil moisture gradient in Harenna forest showed strong negative correlation (Tesfaye and Teketay 2005a). Soil moisture ranged from 15.6 to 27.2% between 1,500 and 2,100 m and exhibited an increasing trend with increasing altitude. The highest soil moisture was recorded at 2,100 m, whereas the lowest was found at 1,500 m. The lower altitudinal range between 1,500 and 1,600 m had relatively drier soil while the upper elevation, i.e. 2,100 m, was wet. The negative correlation between density of seedlings and soil moisture is a good indication of the habitat requirements of the species, i.e. its preference for relatively dry forest habitats. This might also explain, at least partly, the absence of *Podocarpus* from the rainforests of southwestern Ethiopia (Chaffey 1979; Friis 1992), where the soils are extremely wet (Yeshitila 1998).

21.3.6 Regeneration in Afromontane Forests

The available information on the natural regeneration (density and frequency) of *Podocarpus* in some of the Afromontane forests, namely Chilimo, Gara Ades and Menagehsa, Gelawdiwos, Munessa-Shashemene, Harenna, Kimphee, Wof-Washa, and Zegie forests is summarized in Table 21.1.

Table 21.1 Natural regeneration of *Podocarpus* in some Afromontane forests in Ethiopia

Forest	Location	Density ha^{-1}	Frequency (%)	Source(s)
Chilimo	9°05′N and 8°10′E	348	54	Woldemariam et al. (2000)
Gara Ades	9°20′N and 38°35′E	874	65	Teketay (1997b)
Gelawdios	11°02′ to 12°33′N and 37°25′ to 38°41′E	9	7	Wassie et al. (2005)
Harenna	6°40′ to 7°10′N and 39°30′ to 40°00′E	1,065	53.8	Tesfay et al. (2002)
Kimphee	7°13′N and 38°37′E	17[a]	4[b]	Senbeta and Teketay (2003)
Munessa-Shashemene	7°13′N and 38°37′E	170[c]/102[d]	78[c]/89[d]	Tesfay et al. (2010)
Menagesha	9°00′N and 38°35′E	616	32	Teketay (1997b)
Wof-Washa	9°45′N and 39°45′E	246	–	Teketay and Bekele (1995)
Zegie	11°40′ to 11°43′N and 37°19′ to 37°21′E	6	5	Alelign et al. (2007)

[a]Relative density
[b]Relative frequency
[c]Seedlings
[d]Saplings and trees

21.4 Silviculture and Management

Growth and yield of indigenous trees in Ethiopia has not been a subject of much research, except those reported from studies carried out by the Southwest Ethiopia Forest Inventory Project (Chaffey 1978a, b, c) and a compilation of growth performance of some forest trees in the country (Örlander 1986). The available information from these sources is discussed here despite the fact that the information is relatively old.

Chaffey (1978a, b, c) reported the stocking density and the standing gross volume over bark of *Podocarpus* for two diameter at breast height (DBH) classes ($\geq$30 and 30–120 cm, the latter being the preferred sawlog range) with the reliable minimum estimate at 95% in three natural forests in Ethiopia, namely Megada, Munessa-Shashemene, and Tiro, where *Podocarpus* is the principal timber species (Table 21.2). At Megada, approximately 90% of the standing volume was composed of *Podocarpus*. The growing stock was composed of largely mature and overmature individuals, and there was a paucity of pole-size individuals. At Munessa-Shashemene, *Podocarpus* accounted for 21% of the stocking density and 62% of the standing gross volume, which was estimated at approximately 860,000 m^3. Again, the size class distributions of *Podocarpus* suggested a preponderance of mature and overmature stems and a paucity of regeneration. At Tiro, *Podocarpus* was less abundant accounting for only 3.2% of the total stocking density and 11.6% of the gross standing volume, which was estimated at 362,500 m^3. An earlier report (von Breitenbach 1963) indicated that mature trees from "virgin forests" yield 5–6 m^3 of sawlogs that allow a cut-out of 60–70% of lumber, on the average.

Species elimination trials were established in 1975 at different agro-ecological zones in the northern (Tigray), eastern (Welo), and southeastern (Harrerge) Ethiopia with the assistance of UNDP/FAO to identify suitable species for different sites. The trial involved mainly planting of exotic hardwoods and conifers, but few indigenous species (such as *Acacia albida* Del., *Cordia africana* Lam., *Hagenia abyssinica* (Bruce) J.F. Gmel., *Olea europaea* L. subsp *cuspidata*, (Wall. ex G. Don) Cif. *Juniperus procera* Hochst. ex Endl., and *Podocarpus*) were also tested. *Podo*carpus failed to survive in almost all sites except in Belete-Gera forest area,

Table 21.2 The densities and standing volumes of *Podocarpus* in Magada, Munessa-Shashemene, and Tiro forests

Sites	Density (stems ha^{-1})				Volume (m^3)			
	DBH $\geq$ 30 cm		DBH = 30–120 cm		DBH $\geq$ 30 cm		DBH = 30–120 cm	
	Total stocking	Reliable minimum stocking	Total stocking	Reliable minimum stocking	Gross standing	Reliable minimum	Gross standing	Reliable minimum
Magada	23.5	22.0	19.7	18.2	856,500	798,500	609,000	558,500
Munessa-Shashemene	12.4	11.4	9.9	9.1	553,000	463,000	259,000	231,000
Tiro	2.7	2.0	2.5	1.9	42,000	29,500	32,500	24,000

close to Jimma. At this site, survival of planted *Podocarpus* was 39% at the age of 9.8 years with a mean height of 8 m and volume over bark of 5 m^3 ha^{-1}.

Documented information on the silvicultural and management systems used in the natural forests is lacking in Ethiopia. The harvesting procedure employed, including for *Podocarpus*, is dominantly selective cutting of quality and large trees of economic importance (creaming off best mother trees). For instance, in Harenna forest, 22 tree species were under commercial logging. The five most logged species were *Podocarpus* (80%), *Pouteria adolfi-friederici* (Engl.) Baehni (5%), *Warburgia ugandensis* Sprague (3%), *Croton macrostachyus* Del. (3%), and *Olea capensis* L. (2%). In three stands investigated, the logging intensities ranged between 18 and 48 individuals ha^{-1}. The mean densities of natural regeneration in the logged stands varied between 267 and 273 individuals ha^{-1} for trees, 133 and 1,409 individuals ha^{-1} for seedlings and 35 individuals ha^{-1} for saplings (Tesfaye and Teketay 2005b). Logging hardly gives any consideration for minimizing damages to residual stock, and there is practically no effort for implementing post harvest operations, i.e. standard silvicultural procedures for forest renewal or regeneration. It can be said that properly defined silvicultural and management systems are serious gaps in Ethiopian forestry. Hence, knowledge on growth rates, rotation age, biology, and ecological requirements of most of the indigenous species is deficient. Understandably, this provides a research opportunity for interested researchers in and outside of Ethiopia. For most of the forests, there is no management plan to guide the actual operations on the ground.

21.5 Human Impact

Podocarpus has been among the few species exploited for their timber for quite a long time in the different parts of Ethiopia (von Breitenbach 1963; Chaffey 1979; Teketay 1992, 1994, 1996; Bekele 1994; Tesfaye 2008). For instance, the exploitation of *Podocarpus* started as early as 1920 in the Gara Ades dry Afromontane forest (Teketay 1992). Its trees were selectively cut, leading to the depletion of mature trees, i.e. loss of reproductive individuals in the population. The best individuals were exploited from forests without due consideration for their replacement or future regeneration. Such a highly selective logging operation exerted negative pressure on the harvested species, reducing their quality, i.e. eroding their genetic potential, and abundance, thereby, resulting in the local disappearance of merchantable trees. Moreover, the felling operation caused enormous damage to the young plants, standing trees, and residual stands. For instance, in Harenna forest, *Podocarpus* accounted for 80% of the total timber harvested in the Harenna forest between 1970 and 1995 (Tesfaye et al. 2002), and the saplings and pole-sized/mature trees were especially depleted in the heavily disturbed zone within the forest, mainly between 1,500 and 1,650 m, where coffee is naturally grown and managed. Canopy thinning, ground clearance, and weeding by the people are major forms of human disturbances in areas at lower altitudes.

The current low level of representation of saplings, pole-sized individuals/ mature trees in the forest may prevent the persistence of *Podocarpus* trees as dominant canopy component in the forest. In addition, when the most dominant trees in the canopy, such as *Podocarpus*, are harvested selectively, the other remaining species will be stressed by the dramatic change in understory light, temperature, and moisture (Saulei and Lamp 1991). Thus, the remaining species are likely to be less fit to the changing environment and the future of sustained timber yield and natural regeneration may be diminished.

Cognizant of the overexploitation of *Podocarpus* and the associated danger of serious decline in the density and diversity of its populations in the different parts of the country, and the potential subsequent local extermination of the species, its harvesting/cutting has been banned by proclamation since 1994 by the Government of Ethiopia. Despite the proclamation, cutting/logging of trees has continued unabated owing to no or inadequate legal enforcement of the proclamation.

21.6 Conclusions

Podocarpus is an extremely valuable timber species, which has been under intensive exploitation, both legally and illegally, from the natural forests of Ethiopia over the last several decades owing to its excellent softwood. Its wood is used to manufacture several products. The species is also the source of oil, which is obtained traditionally from the seeds, and feed for wild animals. The oil extracted from the seeds make the species more valuable than before and adds a strong component to its multipurpose nature, which helps to justify its sustainable management. Quite a number of organisms depend on it as part of their habitat. However, its unsustainable exploitation from natural forests around the country is leading to the verge of extermination of its populations from their habitats. Unfortunately, owing to a number of factors, there are neither large-scale plantations so far nor planned plantation establishment programs of the species. Hence, there is an urgent need to implement appropriate silvicultural and forest management measures that would help to fill the existing gaps, facilitate its continuous regeneration, and ensure its sustainable management/utilization. This requires (1) detailed surveys of the remaining populations across its habitats and geographical distribution to determine the actual and potential status of natural and artificial regeneration as well as knowledge gaps about the species; (2) development and implementation of realistic management plans appropriate to the different types of natural forests in which it grows; and (3) undertaking detailed multidisciplinary research on its reproductive and dispersal ecology, i.e. phenology, seed production, seedling recruitment, survival and growth, seed dispersal and dispersal agents, seed predation, etc.; (4) investigation on the actual and potential of seeds for both human food (oil) and animal feed (left-over cake after oil extraction from seeds) (Teketay 1994); (5) designing and implementing appropriate silvicultural and management systems, including enrichment plantations in the forests where the species occurs naturally; and (6) establishment of both small- and large-scale plantations, as needed, targeted to satisfy the demand of wood for the various wood products.

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Chapter 22
Mangroves: Sustainable Management in Bangladesh

Peter Saenger

Abstract Bangladesh faces two mangrove management challenges: the first is the management of the resources of the Sundarbans natural mangrove areas on a sustainable basis, while protecting the high levels of biodiversity. The second challenge is to manage the planted mangroves of the Bangladeshi shoreline under competing, and sometimes conflicting, management objectives and expectations. Management of the Sundarbans has been based on selective felling, with felling cycles and minimal DBHs adjusted for each of the main commercial species. The allowable annual cut is determined by ongoing forest inventories to ensure harvesting is equal or lower than the growth and reproduction rate. However, despite the adoption of sustainable yield, some degradation of the mangroves is occurring, the primary causes being human interference (e.g. illegal harvesting and pollution) and changed hydro-edaphic conditions (e.g. erosion and accretion, and increased soil salinity due to water abstraction). The protection from cyclone damage afforded by the Sundarbans mangrove forests, led the Forestry Department in 1966 to commence a programme of planting mangroves outside the protective coastal embankments in order to provide greater protection for the other coastal areas. Harvestable size is reached in 15–25 years, but prior to harvest, a new crop must be established, so that coastal lands are not unvegetated and liable to erosion. The mangrove greenbelt has brought many obvious benefits, but despite these benefits, there are problems which are not silvicultural, but result from population pressure; illicit felling of trees and unlawful grazing of coastal lands, threaten the mangroves and the encroachment in some areas by shrimp farms comprises a major concern. The protective benefits from the sustainable management of natural and planted mangroves in Bangladesh are beyond dispute – the minimal damage suffered by the coastal areas from the Indian Ocean tsunami on 26 December 2004 can be ascribed simply to the good condition of the natural and planted mangrove greenbelt of that country. When viewed together with the benefits that mangrove habitats bring to biodiversity conservation, it would seem to be obvious that mangrove greenbelts

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should be actively promoted. What has been shown in Bangladesh is that the silvicultural expertise needed has been developed.

22.1 Introduction

Bangladesh is confronted with two mangrove management challenges: the first is the management of the resources of the Sundarbans natural mangrove areas on a sustainable basis, while protecting the high levels of biodiversity within the largest contiguous mangrove area in the world, stretching for around 260 km along the Bay of Bengal from the Hooghly River Estuary (India) to the Meghna River Estuary in Bangladesh. The second challenge is to manage the planted mangroves of the Bangladeshi shoreline under competing, and sometimes conflicting, management objectives and expectations.

22.2 The Sundarbans Mangrove Forests

Productive forests are very limited in Bangladesh and the extensive mangrove forests of the Sundarbans have been an important historical resource (Curtis 1933; Das and Siddiqi 1985; Siddiqi 2001). Reaching between 80 and 100 km inland, the Sundarbans mangroves are traversed by numerous tidal creeks and rivers (Fig. 22.1), and mangrove forests cover a total area of around 10,000 km^2.

Fig. 22.1 The mangroves of the Sundarbans display a high species diversity, forming a rich harvestable resource

The name Sundarbans is derived from the term meaning "forest of *sundari* or *sundri*", a reference to the large mangrove tree, *Heritiera fomes* Buch.-Ham., which can reach the impressive height of 25 m. Felled mainly for its high-value timber used for shipbuilding, electricity poles, railway sleepers and house construction, it accounts for about 75% of total wood extraction and exports.

Other economic activities that take place in the Sundarbans include fishing, and thatching, bees' wax and honey collecting. The fertility of the region is maintained by the annual flood of the Ganges–Meghna–Brahmaputra Rivers that deposits huge amounts of sediment, carried down from the Himalayas (Hussain and Acharya 1994).

22.2.1 Managing the Forests

With the advent of British rule in India, control over the Sundarbans was assumed in 1828 and, since 1875, the Forest Department assumed responsibility for its management as a Reserved Forest. The first minimum exploitable diameter for *sundri* was established at this time. And in 1879, the head office of the Forest Department was established at Khulna to manage the entire Sundarbans areas – both the Indian and Bangladesh portions as a single integrated unit.

Over the last 130 years, management of the Sundarbans has been based on selective felling, with a fixed minimal DBH of 35 cm (Siddiqi 2001), and felling cycles and minimal DBHs were adjusted for each of the main commercial species, including *Heritiera fomes* Buch.-Ham., *Xylocarpus granatum* Koenig, *Sonneratia apetala* Buch.-Ham., *Excoecaria agallocha* L. and *Bruguiera* spp., whose ecological requirements and growth performances are summarised in Table 22.1. Various working plans have been developed, usually based on 40-year felling cycles, sometimes including improvement felling after 20 years.

Table 22.1 Ecological requirements, growth data and timber values of the major mangrove species of Bangladesh

Species	Habitat	Light	Salinity	MAI_{DBH}	Main uses
Avicennia spp.	Mudflats	High	Medium	0.64	Fuelwood, anchor logs
Bruguiera spp.	Backswamp	Low	Medium	0.35	Furniture, building
Ceriops spp.	Backswamp	Medium	Medium	0.05	Fuelwood, house posts, charcoal
E. agallocha	Backswamp	High	High	0.15	Matches, newsprint
H. fomes	Backswamp	Low	Low	0.13	Building, boats, electric poles
S. apetala	Mudflats	High	Medium	0.60	Packing boxes, building
Xylocarpus spp.	Backswamp	Low	Medium	0.27	Furniture, building

MAI_{DBH} mean annual DBH increment

Avicennia is represented by *A. marina*, *A. officinalis* and *A. alba*; *Bruguiera* is represented by *B. gymnorhiza* and *B. sexangula*; *Ceriops* is represented by *C. tagal* and *C. decandra*; *Xylocarpus* is represented by *X. moluccensis* and *X. granatum*

Data from Siddiqi (2001), Saenger (2002), and Iftekhar and Saenger (2008)

At present, felling cycles of 20 years are prescribed with minimal DBHs set for each mangrove species in each of three site quality classes (Siddiqi 2001). Site classes, which are an expression of growing conditions, particularly of soil salinity, are defined on vegetation height: site class 1, >15 m; site class 2, 10–15 m; and site class 3, <10 m. These measures were designed to manage the Sundarbans on a sustainable yield basis long before that concept was adopted elsewhere. The allowable annual cut is determined by ongoing forest inventories to ensure harvesting is equal or lower than the growth and reproduction rate. Thus, the system works relatively well, based on site classes, reasonable control and reliable forest inventories. However, despite the adoption of sustainable yield, some degradation of the mangroves is occurring.

22.2.2 Management Problems and Their Causes

The growing stock of the two major mangrove species, *H. fomes* and *E. agallocha*, has significantly decreased in recent years, accompanied by changes in canopy closure, basal area, and species composition and dominance (Siddiqi 2001; Saenger 2002; Iftekhar and Saenger 2008). For example, canopy closure of >70% characterised 78% of Sundarbans mangroves in 1959, was found in 65% of the mangroves in 1983, but has declined to less than 1% by 1996 (Siddiqi 2001). Similarly, the merchantable volume of *H. fomes*, the most sought-after and valuable timber, has declined from about 35 $m^3\ ha^{-1}$ in 1959 to around 20 $m^3\ ha^{-1}$ by 1983 (Siddiqi 2001).

Self-regeneration is adequate, although sometimes patchy, and browsing by deer is not a major problem. Top-dying of *H. fomes*, a disorder causing progressive dieback from the tops of the trees downwards, currently affects around 17% of the trees (Siddiqi 2001) and, until the exact cause is established, poses a potentially significant problem. Nevertheless, it seems that the primary causes of mangrove degradation are human interference (e.g. illegal harvesting and pollution) and changed hydro-edaphic conditions (e.g. erosion and accretion, and increased soil salinity due to water abstraction).

Since 1966 much of the Bangladesh Sundarbans has also been a wildlife sanctuary to protect the Royal Bengal tigers and their principal prey, the spotted deer. In 1997, 1,400 km^2 of the Sundarbans was declared as a World Heritage Site in recognition of its outstanding biological value; it contains the largest contiguous mangrove area in the world with around 30 of 85 mangrove species worldwide, providing habitat for 49 species of mammals, 261 species of birds, 50 species of reptiles, 8 species of amphibians and 87 species of fish (Hussain and Acharya 1994). The real challenge for the management of the mangroves of the Sundarbans will be to determine at what level of mangrove habitat degradation the biodiversity values will be adversely affected.

22.3 Mangrove Afforestation: The Coastal Greenbelt

The coastal areas of Bangladesh suffer severe cyclone damage almost annually. Available cyclone recordings for the area begin as early as 1584, while during the period from 1960 to 1970 alone, eight severe cyclones were recorded. The severe cyclone and associated storm surges of 1970, 1985 and 1991 were reported to have caused the deaths of approximately 300,000, 100,000 and 300,000 people, respectively (White 1974; Saenger and Siddiqi 1993). The cyclone of 1991 alone was estimated to have caused an economic loss of US$1.2 billion (BBS 1992). The protection from cyclone damage afforded by the Sundarbans mangrove forests, led the Forestry Department in 1966 to commence a programme of planting mangroves outside the protective coastal embankments in order to provide greater protection for the other coastal areas. Annual plantings of approximately 800 acres (324 ha) were undertaken on newly accreted land in the coastal areas of Patuakhali, Barisal, Noakhali and Chittagong districts to establish a coastal greenbelt.

22.3.1 Objectives of Coastal Afforestation

As the coastal afforestation programme proceeded, it became apparent that the plantations contributed to both the acceleration of land accretion and to the stabilisation of the char lands, with the ultimate potential of providing land sufficiently stabilised to be used for agricultural purposes.

The early success of the plantations resulted in the setting of additional objectives for coastal afforestation, including (1) conservation and stabilisation of newly accreted land and acceleration of further accretions, with the ultimate objectives of transforming a large part of this area to agricultural land; (2) production of timber for fuelwood and industrial use; and (3) creation of employment opportunities in remote rural areas.

These objectives were not ranked, as a result of which they often became mutually competitive. For example, when newly planted mangroves led to excessively accelerating accretion (Fig. 22.2), it frequently resulted in mangrove burial and death, thus obviating timber production. Similarly, the establishment of a second rotation for timber production was sometimes compromised by premature agricultural pursuits, such as buffalo grazing.

Despite a lack of clear hierarchical objectives, the afforestation programme was accelerated in 1974 and by 1980, 32,400 ha had been planted under mangroves. From July 1980 to Dec 1985, the World Bank-funded Mangrove Afforestation Project I was to plant approximately 20,000 acres of mangroves annually, and the Mangrove Afforestation Project II was to add a further 20,000 acres annually from 1986 to 1990. Although accurate figures are still being collated by the Space Research and Remote Sensing Organization (SPARRSO), current “best” estimates indicate a total plantation area of 80,000 ha (Saenger and Siddiqi 1993; Blasco et al. 2001).

Fig. 22.2 Plantations of *Avicennia marina* largely covered by sand deposition near Chittagong, with only the extreme tips of the crown above the surface

Table 22.2 Mean (±SE) survival, H, DBH, stemwood production (MAI_{vol}) and tree quality at different initial spacings for 5-year-old *Sonneratia apetala* plantations at Barisal, Bangladesh, from three replicated random blocks

Spacing (m)	Survival (%)	H (m)	DBH (cm)	MAI_{vol} $m^3\ ha^{-1}\ y^{-1}$	Unforked trees (%)
0.85 × 0.85	29 ± 14	9.1 ± 0.8	8.8 ± 1.3	16.1 ± 2.3	90 ± 7
1.2 × 1.2	52 ± 9	8.7 ± 0.3	7.9 ± 0.3	18.5 ± 2.7	90 ± 2
1.7 × 1.7	42 ± 15	8.9 ± 0.2	9.7 ± 0.5	11.8 ± 2.2	84 ± 2
2.4 × 2.4	37 ± 21	7.8 ± 0.5	13.6 ± 2.5	5.6 ± 1.2	73 ± 6
3.4 × 3.4	39 ± 21	7.9 ± 0.8	12.5 ± 3.1	2.0 ± 1.0	38 ± 16

Data from Siddiqi (1987, 1988) and Saenger and Siddiqi (1993)

22.3.2 Managing the Greenbelt

Much of the early coastal afforestation was based on *Sonneratia apetala* because of the ease with which this species could be propagated: fruits were collected during later summer, and allowed to decompose to yield on average 50 seeds per fruit. Broadcast over intertidal nursery beds (Fig. 22.3), they germinate with 4–7 days, and the seedlings are removed and planted out when around 15–25 cm high (Saenger and Siddiqi 1993). Survival rates after 6 months varied from 20 to 80%, depending on seasonal factors and on the age of the seedlings (Islam et al. 1992).

Optimal planting densities were determined from field trials (Table 22.2). For timber production without thinning, an initial plant spacing of 1.2 m appears optimal. Where thinning products have little or no economic value, it is preferable to adjust initial spacing to reduce the need for thinning and to optimise plant growth

Fig. 22.3 Intertidal nursery beds are used for raising seedlings of *Sonneratia apetala* for subsequent planting out into the newly accreted coastal areas

(Islam et al. 1993). However, for coastal protection where features such as multiple stems or reduced height are desirable, initial plant spacing in excess of 2 m appears to be most suitable. Although selective improvement felling is sometimes undertaken, this is generally not economically feasible, and planting density adjustments are used to minimise thinning.

Harvestable size is reached in 15–25 years, with heights between 7 and 20 m, mean DBHs of 10–20 cm and yields ranging from 5 to 20 m^3 ha^{-1} $year^{-1}$, depending on the site-specific growing conditions (Siddiqi and Baksha 2001). However, prior to harvest, a new crop must have been established, so that coastal lands are not unvegetated and liable to erosion at any stage during this process. As conditions under the mature mangroves will have changed considerably, the newly underplanted second rotation must be selected with due consideration to these changed conditions of semi-shade and drier, more consolidated sediments. *Excoecaria agallocha*, *Xylocarpus moluccensis* (Lam.) Roemer, *Heritiera fomes*, and *Bruguiera sexangula* (Lour.) Poir seem to suit these conditions well, and have been successfully used as a second rotation crop, all comprising high-value timber species (Siddiqi 2001). Third rotations are rarely mangroves, as soil salinities have been sufficiently reduced, and soil maturation advanced, to the stage that mesophytic plants (e.g. commonly *Acacia nilotica* (L.) Delile, *Casuarina equisetifolia* L., *Pithecellobium dulce* (Roxb.) Benth. and *Albizia procera* (Roxb.) Benth.) can be established, or the lands handed over for grazing purposes.

22.3.3 Greenbelt Benefits and Problems

The mangrove greenbelt has brought many obvious benefits (Siddiqi and Baksha 2001), including the stabilised coastal lands, which provide timber and fuelwood, on the one hand, and pasture land on the other. The protection from cyclones and storm surges is another important benefit, together with the increase in tourism and recreation in coastal areas. Perhaps less obvious, is the enhancement of coastal fisheries as a result of detrital input from the mangrove greenbelt, and the contribution these habitats make toward the conservation of floral and faunal biodiversity.

Despite these benefits, there are problems for the coastal greenbelts, which are not silvicultural, but rather, result from population pressure; illicit felling of trees and unlawful grazing of coastal lands, threaten the mangroves and the encroachment in some areas by shrimp farms comprises a major concern.

22.4 Future Trends

The degradation of natural mangrove forests, both in terms of area loss and forest structure, needs to be addressed urgently; illicit cutting and pollution require regulatory decisiveness, but the problem of water abstraction is more intractable.

Coastal greenbelts have been very successful in Bangladesh because scientifically sound plantation strategies were developed as a result of World Bank funding, and as a result of accumulated experience. The objectives for greenbelts has been prioritised as for coastal protection, as that underpinned all other benefits. And the protective benefits from the sustainable management of natural and planted mangroves in Bangladesh are beyond dispute – the minimal damage suffered by the coastal areas from the Indian Ocean tsunami on 26 December 2004 can be ascribed simply to the good condition of the natural and planted mangrove greenbelt of that country. The damage in other neigbouring countries, where greenbelts do not exist or where natural mangroves have been severely degraded, was immense, sadly, with great loss of property and lives. When viewed together with the benefits that mangrove habitats bring to biodiversity conservation, it would seem to be obvious that mangrove greenbelts should be actively promoted. What has been shown in Bangladesh is that the silvicultural expertise needed has been developed and is available.

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Part VII
Silviculture in Secondary Forests

Chapter 23
Review
Silviculture in Secondary Forests

Shadrach Olufemi Akindele and Jonathan C. Onyekwelu

Abstract Secondary forests are forests regenerating through natural processes after significant reduction in the original vegetation at a point in time or over an extended period, and displaying a major difference in forest structure and/or canopy species composition with respect to nearby primary forests on similar sites. Despite their large extent, existing and potential benefits, secondary forests are mostly overlooked. The increasing area of secondary forests necessitates their professional management. If properly managed, secondary forests can provide important social and environmental benefits, contribute to poverty alleviation and reduce the pressure on the few remaining areas of primary forest. However, only suitable silvicultural treatments can restore and increase the commercial value of secondary forests. This chapter discusses the degradation processes leading to secondary forest formation, their structures, growth and yield and regeneration processes. Three insightful and demonstrative case studies were also presented to illustrate key points.

Keywords Secondary forests · Silvicultural systems and techniques · Degradation process · Succession · Regeneration · Biodiversity conservation

23.1 Introduction

23.1.1 Definitions

Secondary forests refer to woody vegetation regrowing on land after natural and/or human disturbance of the original forest. Even though human disturbances in form of logging and agricultural practises are generally more frequent (Oliver and Larson

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1990), natural disturbances such as fire and hurricanes also destroy primary forests to give way to secondary forests. Secondary forests are products of secondary succession. The concept of secondary succession relates to the observed sequence of vegetational change from dominance of early to later successional species on disturbed or cleared sites. A more detailed definition of secondary forests has been given by Chokkalingam and de Jong (2001), who define them as forests regenerating largely through natural processes after significant reduction in the original forest vegetation through tree extraction at a single point in time or over an extended period, and displaying a major difference in forest structure and/or canopy species composition with respect to nearby primary forests on similar sites. From this definition, the following vegetation types are not regarded as secondary forests:

- Forests subject to low-intensity selective logging, that is when management interventions are significantly below critical natural disturbance frequencies and intensities.
- Forests subject to low-intensity, small-scale extractive activities (e.g. for non-timber forest products).
- Forests affected by small-scale natural disturbance.
- Intensively managed plantations.
- Forests regenerated largely through planting.

The definition provided for secondary forests include several key components such as significant disturbance, largely natural re-growth, and difference in structure and/or composition from adjoining vegetation. The thresholds for many of these components need to be determined at regional or ecosystem levels. Some variations exist in the type of secondary forests across the globe. For instance, on the one hand, in West Africa, most secondary forests are young, corresponding to forest fallows or logged-over residual forests. On the other hand, most secondary forests in Central Africa are old forests and generally result from the very slow colonisation of savannas, to which the present climate is favourable. In Costa Rica, high deforestation rates for cattle raising during the 1970s, followed by pasture abandonment due to a drop in export meat prices one decadeclater (Butterfield 1994), resulted in the development of secondary forests, particularly in the wet, Caribbean lowlands. These forests are characterised by low number of canopy tree species but with high relative dominance (Guariguata and Finegan 1997).

23.1.2 Degradation Processes

In the absence of human or natural disturbance, primary forests remain intact with the ability to regenerate and close gaps left whenever any ageing tree falls. However, when forest degradation processes set in, forest cover is reduced and such forests become secondary forests.

Several factors lead to forest degradation. These include land cultivation for agriculture, logging activities (which are often without due sustained yield principles), conversion of natural forests into plantations (usually monocultures) and other land-use types, implementation of developmental projects (i.e. construction of infrastructural facilities), forest fires, and intensive animal grazing. While some of the land-use types lead to permanent forest loss, others (such as intensive commercial logging) lead to forest degradation. The degradation process sets in because the forest is hardly allowed to recover before another felling cycle commences. This incessant nature of the exploitation activities has often led to a drastic reduction in the quantity and quality of timber obtained from such forests.

23.1.3 Structures, Growth and Yields in Secondary Forests

Like any other tropical forest ecosystem, secondary forests display multi-storey complex structures with relatively high tree species composition (Chokkalingam et al. 2000). By the nature of their formation, secondary forests differ from primary forests in structure, tree species composition and age class range, although there may be old secondary forests whose structure may resemble that of primary forests. Thus, the most determining factors that differentiate the two categories of forests are structure and species composition: a primary forest presents in general more tree species per hectare at a given age, while age class and diameter distribution is more varied; whereas in a secondary forest the number of tree species per hectare is usually limited to a few, and the age class and diameter distribution tends to be more homogeneous. For example, Onyekwelu et al. (2008) reported 51 tree species per hectare in a primary forest in Nigeria as against 31 for an adjacent secondary forest.

The structure of secondary forest varies greatly. However, they are simple when compared to matured forests. The characteristics that typify secondary forests include: high total stem diversity but low density of trees greater than 10 cm dbh, low basal area, short trees with small diameter, low timber volume and high leaf area indices (Brown and Lugo 1990). The structural characteristics change with age, with the rate of change affected by climate and soil type. For example, as age increases, the stand-weighted wood specific gravity of secondary forests increases (Saldarriage et al. 1986; Weaver 1986). When very young the abundance of saplings and climbers generally gives secondary forests a dense tangled appearance, which makes them difficult to penetrate (Blay 2002). However, after some years (especially on abandoned cultivated land), cohorts of trees of remarkably regular structure emerge, consisting mainly of a single fast-growing species such as *Musanga cecropioides* (in Africa). Those trees, which are dominant for a single generation, are mostly light-demanding and short-living species. They are later replaced by a mixture of less fast-growing, more shade-tolerant and longer-living trees, some of which are pioneer and some primary forest species (Blay 2002). In contrast, the secondary forest vegetation developing after logging, at least during its earlier stages, is very irregular in structure. Damaged survivors from the former forest are scattered among climber tangles and patches of "razor grass" (*Scleria* spp.), and dense stands of saplings may be present (Blay 2002). Secondary forests are generally characterised (depending on its level of degradation) by a less developed canopy structure and smaller trees when compared to primary forests. Due to the lack of a full canopy, more light will reach the floor, supporting vigorous ground vegetation.

The growth of trees in secondary forest is highly variable, too, with large differences between species, tree sizes, sites, and even between the same sizes of individual trees of the same species growing in the same site. In contrast, growth of individuals during successive periods is much less variable. Although trees that are growing fast continue to do so, the slow-growing individuals continue to grow slowly.

The productivity of secondary forests may vary in relation to factors such as site conditions (in particular topsoil and humus conditions), time since settlement and, more specifically, the number of crop–fallow cycles at a particular site, the type and intensity of land-use during the cropping stage, and the prevalence of disturbances such as accidental burning during the fallow stage (Blay 2002; ITTO 2002). Other factors that may influence the growth and yield of secondary forests include: the age of the stand, density, secondary forest type, etc., which is clearly evident from the data published by Brown and Lugo (1990) presented on Table 23.1. As succession progresses, total stem density tends to decrease and the stand increases in height, basal area and volume. The first 15 years of succession are characterised by rapid biomass accumulation up to 100 tonne ha^{-1} (Brown and Lugo 1990). The relative amount of woody biomass also increases rapidly during the first 15–20 years, followed by a steady but slower rate until maturity (Blay 2002). There are indications that volume growth is higher in advanced secondary forest than in young secondary forest, which may be attributed to the presence of bigger individual trees in advanced secondary forests. Weaver and Birdsey (1990) reported a significantly higher mean volume growth rate of 6.9 m^3 ha^{-1} year^{-1} in an advanced secondary forest in Puerto Rico than the 2.0 m^3 ha^{-1} year^{-1} in young secondary forest. However, within a specified time scale, growth is sometimes more active in young secondary forest, for example while growth increased by an average of 50% within a 5-year period in young secondary, advanced secondary forest increased by only 18% within the same period (Weaver and Birdsey 1990).

Gomide et al. (1997) reported a decreasing trend in growth parameters of a secondary forest in Amapa, Brazil. For example, during the first period of observation, the average diameter growth rate of 1.93 cm year^{-1} was recorded, which decreased to 0.34 cm year^{-1} after 2 years. Basal area increment changed from about 6 m^2 ha^{-1} year^{-1} to 1 m^3 ha^{-1} year^{-1} within 10-year period. This decreasing trend was attributed to canopy closure, competition and the progressive decline in pioneers in the forest (Gomide et al. 1997). An average volume growth of 3.5 m^3 ha^{-1} year^{-1} in 14 years was reported for the secondary forest in Amapa, Brazil.

Table 23.1 Structural and growth characteristics of secondary forests

Age (years)	Min. dbh measured (cm)	Basal area ($m^2\ ha^{-1}$)	Density (No. ha^{-1})	Canopy height (m)	Mean Dbh (cm)	LAI
Tropical wet forest						
4.5 (Poor soil)	All stems	4.3	4,060	6	–	–
4.5 (Rich soil)	All stems	16.3	3,867	12.2	–	–
9.5	All stems	12.7	2,200	11.5	–	–
15	–	26.9–37.1	400–610	–	20–25	–
Tropical moist forest						
10	10	12.8	342	10	–	5.8
20	10	16.9	461	–	–	6.9
35	10	18.6	495	18–19	–	6.6
60	10	24.5	441	–	–	5.6
80	10	24.0	604	–	–	6.4
Mature	10	38.4	570	25–35	–	7.5
2	–	–	–	7.2	6	7.5
2	–	–	–	8	12	6.9
4	–	–	–	10	13.8	11.6
6	–	–	–	12	17.6	16.5
Tropical dry forest						
8	3.2	11.8	2,038	25	–	–
3	1.0	–	512	2.5	1.8	–
7	1.0	–	2,270	4.7	4.6	–
10	1.0	–	2,670	5.8	7.2	–
Late secondary	1.0	–	2,260	10.4	20	–
Sub-tropical wet forest						
6	10	7.2	1,334	10	–	–
20	10	28.5	1,234	25	–	–
21	10	27.8	2,436	19	–	–
50	10	33.8	1,593	24	–	–

Source: Brown and Lugo 1990

23.1.4 Biodiversity

Most human-modified areas in the tropics, such as secondary forests, were largely considered hostile to biodiversity. However, a series of studies done in the last few years have painted a very different picture. Secondary forests play vital function and are becoming increasingly important in conserving biodiversity because they currently constitute a large proportion of tropical forest cover. One of the most notorious characteristics of secondary forests is the high biodiversity heterogeneity between stands only short distances apart, both in the canopy and in the understorey. This has been mainly attributed to phenological variations of colonising species at the moment of land abandonment (fallow period), the type of regeneration (re-sprouts versus seeds), past land-use type as well as the presence of different species of remnant trees, which can influence species composition (Brown and Lugo 1990; Blay 2002).

Most secondary forests differ in floristic composition from primary forests. It is generally believed that secondary forests always have fewer tree species per unit area than primary forest, but this is not necessarily so, especially in comparison with fairly old secondary forests. Although secondary forests contain often less canopy (tree) species diversity than primary forest, they could be the sources of important timber species and other species used for carving and the manufacture of household items (Blay 2002). Some economic tree species found in secondary forests include *Triplochiton scleroxylon, Ceiba pentandra, Daniella ogea, Lophira alata, Milicia excelsa, Nauclea diderrichii, Terminalia ivorensis, Terminalia superba, Alstonia boonei*, etc.

Since the occurrence and distribution of tropical forests are largely explained by geophysical characteristics, and the quantity and seasonality of rainfall, the composition and species population sizes of secondary forests are determined in part by species tolerance of prevailing environmental conditions, particularly rainfall and soils, and in part by local site history (Swaine and Hall 1983). Generally, the tree species that dominate secondary forests are apparently unable to regenerate under their own shade, as suggested by the absence of small size classes in stem-diameter distributions, changes in tree species composition across a forest chronosequence, and by monitoring tree recruitment over long periods (Guariguata 2000). During the early stage of secondary succession, the number of species per hectare increases rapidly but slows down after a few years. It is noteworthy that semi-deciduous and deciduous tree species characteristic of seasonal climates are often common in secondary rainforest. In West Africa, for example, *M. excelsa* and other mixed semi-deciduous forest trees often occur in clearings and young secondary rainforest. *Bombacaceae* are also often found in old and middle-aged secondary forests (Budowski 1970). Similarly, in a large sample of secondary forests in Costa Rica, 90% of tree species found in old-growth forest areas were represented by either seedlings, small trees, or large trees, with the number of species increasing with age of the forest, allowing biodiversity to recover faster than many expected (Hance 2009). Various factors influencing the recovery of species composition in secondary forests have been identified, among which are: stem coppicing and root sprouting mechanisms, seed availability, nature of disturbance, etc. (Uhl and Clark 1983; Murphy and Lugo 1986; Brown and Lugo 1990).

Unlike tree species, the understorey species assemblages in secondary forests are more diverse, richer, and have a higher density than the understorey assemblage in primary forests. Understorey plant species assemblages may have different patterns of diversity than tree species because of variable responses to different abiotic factors. Furthermore, the canopies of secondary forests may be more open than in primary forests, allowing higher levels of light to the forest floor, which may result in understorey species assemblages of higher diversity than adjacent primary (Laska 1997). In a study that compared the floristic composition, species diversity, richness, and density of three common families of understorey shrubs between two secondary growth forests (12- and 25 years old) and an adjacent primary forest in Costa Rica, Laska (1997) encountered 22 species (47%) occurring in secondary growth forests only, one species occurring in old-growth forest only and 24 species (51%) in both types of forest. The secondary forest understories were found to have higher species diversity, higher species richness and density than the primary forest.

23.2 Regeneration in Secondary Forests

The regeneration of secondary forests follows a wave of succession that usually occurs in forest openings following disturbances from fallen trees, forest clearing under shifting cultivation, use of heavy machinery and various natural disasters such as floods, hurricanes, wind-throws and die-back, etc. Swaine and Whitemore (1988) recognised two categories in secondary forest succession process: pioneer and secondary species. Pioneer species typically germinate, establish, grow and mature relatively quickly in the clearings and breaks created by the death of dominant trees. Many of the primary species regenerate by seedlings and young plants already on the forest floor, root suckers and rhizomes, seed in the soil and seeds with a very short seed dormancy that happen to be in fruit during disturbance of the area (Richards 1966). Early pioneer species are often fast-growing, short-lived "weedy trees". At the other extreme, many climax species are designed to tolerate resource scarcity. Their seeds are able to germinate in the dark forest understorey and seedlings can tolerate canopy shade for long periods, until disturbance creates an opportunity for growth. Shade-tolerant species reach their peak rates of photosynthesis at much lower light levels than their counterparts (Riddock et al. 1991). Eventually, the late pioneers are replaced by late successional vegetation that is diverse in architectural form and long-lived. The mature forest is the ecological unit that has reached its maximum diversity and number of species by containing all stages of the forest mosaic.

The rate of regeneration in secondary forest appears variable based on the stage (age) of forest growth. Jin et al. (2005) showed that the appearance rate of regeneration species during the restoration period of a secondary forest was very low during the initial 10 years, but was markedly higher after 15 years and had only little change afterwards. The stocking density of regeneration species increased gradually along with the restoration of secondary forest, it could reach 7,500 trees per hectare, with little change at later stage (Jin et al. 2005). Enrichment planting is perhaps the most popular and appropriate method of artificially regenerating secondary forests (Piotto 2007). It is employed to increase the stocking of valuable tree species. Other secondary forest regeneration methods include: refining and liberation thinning, post-abandonment, etc. Where enrichment planting is adopted, native or exotic species could be used. However, priority should be given to the appropriate native tree species, which are well known and adaptable to the area.

23.3 Silvicultural Systems and Techniques

Fundamentally, the silvicultural treatments used to stimulate the production of commercial timber species in tropical primary forests may also be applicable in the regeneration and tending of secondary forests. Experience has shown that young secondary forests are more receptive to silvicultural manipulations than primary forests because of their manageable tree size and rapid growth response

(Müller 2002). The silvicultural strategy adopted must consider tree species composition and structure of the forest. Specifically, information on the extent, location, condition, conversion processes, current and potential uses of secondary forest is necessary. Silviculture in secondary forests should be based in the first instance on existing natural regeneration (ITTO 2002) because new germination or planting of seedlings is more difficult to handle compared to tending the seedlings already present. Thus, one of the most important tasks is the assessment of existing natural regeneration. Since many important timber species are rarely found in secondary forests, the economic value of secondary forests can be increased through strategies that facilitate the growth of economic tree species (Piotto 2007). The basic questions for determining the silvicultural techniques of secondary forests are presented in Table 23.2.

The silvicultural techniques ultimately chosen will depend heavily on the priorities and objectives of the forest owner, the costs and benefits associated with the strategy as well as the economic, social, and environmental values of the land resources in their current and desired future states. The major systems used in secondary forests include enrichment planting within existing stands, refining and liberation thinning, post-abandoned secondary forest, assisted natural regeneration (ANR) and plantation establishment after clear-cutting.

Table 23.2 The four basic silvicultural questions for determining the management strategy for secondary forests

Silvicultural questions	Management strategy
What are the present stand and site conditions?	• Stand: species composition, structure, health, age, regeneration capacity, etc. • Site conditions: edaphic, hydrologic, etc. • Socio-economic context: Who uses the forest, what for, what kind of impact?
What are the stand and site histories?	• Determine the cause(s) of degradation: for example was the area under shifting cultivation? If yes, what intensity? Is the stand a logged-over forest? Did forest fire occur?
How would the site develop in the absence of planned management interventions?	• What will happen to the stand if there is no management? For example, ecologically (succession, etc.) and socially (conversion into other land-use, etc.)

What management strategies are needed to achieve a particular outcome (restoration, secondary forest management, rehabilitation)? Depending on who manages the forest, the question of who plans, who harvests and who monitors will influence how this will be done	• Participatory and adaptive management planning for the particular forest stand or the degraded site: silvicultural options, collaborative use management, multiple-use management
	  • Define objective
	  • Specify methods
	  • Specify monitoring of forest development, and adopt, if necessary, the strategy and the course of action

Source: ITTO 2002

23.3.1 Enrichment Planting

Enrichment planting is one of the major silvicultural methods used in the management of secondary forests and perhaps the most popular and appropriate (Piotto 2007). It is employed to increase the stocking of valuable tree species in degraded or secondary forest, where regeneration of the required species is scanty, partially successful or completely absent, without removing the trees already present (Montagnini and Jordan 2005; Piotto 2007). Enrichment planting is defined as a technique for promoting artificial regeneration of forests in which seedlings of preferred timber trees are planted in the understorey of existing logged-over forests and then given preferential treatment to encourage their growth (Lamprecht 1989). The use of enrichment planting requires canopy manipulation to optimise the growth and survival of the planted trees. The practise is influenced by the ability of the species to survive as young seedlings under existing natural forest stands.

Necessary conditions for successful enrichment planting include the provision of adequate light conditions, proper supervision, and follow-up maintenance (especially canopy opening treatments). In the tropics, enrichment planting has declined because of several reasons, which include (a) planting work is difficult to supervise; (b) seedlings have to be regularly released from regrowth; (c) a regular supply of seedlings is needed and (d) it is costly (labour demanding). Failures have been mainly attributed to the poor or improper selection of species and planting stock. In addition, lack of adherence to sound planting and tending practises, that is, insufficient overstorey opening prior to planting and insufficient follow-up tending operations also lead to some setbacks. In spite of the failures, enrichment planting still has the advantage of mimicking natural gap dynamics and protecting the soil by maintaining vegetation on site. According to Montagnini et al. (1997), the system can be successfully used to increase the value of secondary forests and prevent their conversion to other land uses, thus reducing deforestation. Enrichment planting is an important land-use strategy in the context of the current international attempt to curb deforestation and forest degradation in developing countries (Paquette et al. 2009). If managed successfully, enrichment planting presents an interesting opportunity for carbon sequestration in the tropics.

Due to the relative management complexity of enrichment planting, basic information about the species ecology is fundamental in selecting potential species and predicting their response to silvicultural treatments. Since trees are planted under a measure of shade, the species adopted are mainly shade tolerant species. Also pioneer and late successional species are used. Important silvicultural characteristics for species ideal for enrichment planting include: produce timber of high value, low crown diameter, regular flowering and fruiting, wide ecological range, fast growth rate, tolerance to moisture stress, good natural stem form, free of pests and diseases, drought resistance and ability to grow in low-nutrient soils (ITTO 2002).

The experience with enrichment plantings in secondary forests has generally been more favourable than in primary forests. Enrichment plantings have generally yielded promising results when applied in young secondary forests (Müller 2002), because of their manageable tree size and rapid growth response. However, enrichment plantings tend to be costly and labour-intensive. When high timber productivity is a major objective, a monocyclic system that relies on creating a future, even-aged stand by opening the middle and upper canopies shortly before tree harvesting is perhaps the most appropriate (Müller 2002). This strategy is required for pioneer/light-demanding species that need almost complete canopy removal to stimulate seed germination and sustain seedling growth and survival.

23.3.2 Refining and Liberation Thinning

Refining and liberation thinning are important in the initial stage to demonstrate measurable effects from secondary forest restoration, management and rehabilitation efforts (ITTO 2002). They also reduce the time in which a merchantable crop of timber and NWFP will become available. However, the operations of refining and liberation thinning are costly and only yield distant future returns. Refining refers to the elimination of undesirable trees, climbers, shrubs and other plants that will inhibit site occupation by desirable trees while liberation thinning refers to the cutting that relieves young seedlings, saplings and trees in the sub-canopy layer from overhead competition (ITTO 2002). Refining allocates growing space to the potential final-crop trees at the expense of others. One major setback of this method is that refining can jeopardise species diversity in secondary forests and may endanger the ecological integrity of the stand, thus care must be taken to prevent biodiversity loss (Grieser 1997). In the past, this method was used to eliminate tree species that were then thought to be “useless”, some of which were later discovered to be of high economic importance. Thus, the method is constrained by the limited knowledge of the usefulness of many secondary forest tree species and the continual discovery of use for species that were once considered “useless”. To minimise wastage, a reasonable compromise is to leave sub-canopy species and tree regeneration layers of the canopy as intact as possible, while removing only those trees and climbers that overtop the desired trees (ITTO 2002). Liberation thinning stimulates growth, since tree growth is directly related to the formation of a healthy and dense crown.

23.3.3 Assisted Natural Regeneration

ANR is an alternative and low cost approach to the restoration of native forest biodiversity and productivity. It is a simple, inexpensive and effective method for accelerating and enhancing the regeneration and establishment of secondary forest and shrub vegetation by protecting and nurturing the mother trees and their wildlings inherently present in the area (Ganz and Durst 2003; FAO 2010). The method entails assisting the natural processes of regeneration and planting new trees when necessary. The ANR aims to accelerate, rather than replace, natural successional processes by removing or reducing barriers to natural forest regeneration such as soil degradation, competition with weedy species, and recurring disturbances (e.g. fire, grazing, and wood harvesting). Seedlings are, in particular, protected from undergrowth and extremely flammable plants such as Imperata grass (FAO 2010). It also aims to strike a balance between high-cost restoration planting to restore biodiversity to small areas and the establishment of commercial plantations over large areas to restore productivity (Shono et al. 2007). ANR offers significant cost advantages because the costs associated with propagating, raising, and planting seedlings are eliminated or reduced. Some of the processes involved in ANR methods include: cutting or pressing the weeds around existing naturally established seedlings, protecting the area from fire and interplanting with desired species if necessary. ANR differs from natural regeneration, which allows some human intervention but precludes tree planting.

According to Shono et al. (2007), ANR is most suitable for restoring secondary forests where some level of natural succession is in progress. As a first condition, sufficient tree regeneration must be present so that their growth can be accelerated. Seedlings of pioneer tree species are often found among and below the weedy vegetation even on a seemingly weed-dominated land. The minimum required number of pre-existing seedlings to implement ANR depends on the acceptable length of time for the forest to be restored and site-specific conditions that influence the rate of forest recovery. As a general reference, a density range of 200–800 seedlings (>15 cm in height; counting clumps in 1 m^2 as one seedling) per hectare has been suggested for ANR reforestation, and it has been estimated that at least 700 seedlings per hectare are needed during the early treatment period to achieve canopy closure within 3 years (Jensen and Pfeifer 1989; Shono et al. 2007). To ensure further successional development, remnant forest should be in proximity so that there would be sufficient input of seeds. Most importantly, it must be possible to prevent further disturbances such as fire, grazing and illegal logging because the success of ANR ultimately depends on the continued protection of the site.

Various steps are involved in the implementation of ANR, among which are (see Jensen and Pfeifer 1989; Cohen et al. 1995; Friday et al. 1999; Shono et al. 2006 for more details):

1. Marking woody regeneration;
2. Liberation and tending of woody regeneration;
3. Suppressing weedy vegetation;
4. Protection from disturbance;
5. Maintenance and enrichment planting.

23.3.4 Plantation Establishment

Another silvicultural strategy for managing secondary forest to overcome degradation, while ensuring financial returns is the establishment of commercial tree plantations (Lamb 1998). Few secondary forests will recover unaided, thus there has been increasing interest in using industrial plantation to increase biodiversity in the landscape (Lamb 1998). Numerous studies have demonstrated the catalytic effect of plantations in fostering the regeneration of native forest species in the understorey. The need to integrate biodiversity conservation in commercial plantations is becoming increasingly important, and plantation trials involving high-value native trees and species mixtures are underway in many countries. The use of plantation as a silvicultural strategy for regeneration in secondary forest is often referred to as restoration or ecological rehabilitation plantation. This type of plantation differs from commercial industrial plantation in that trees are invariably planted with uniform distribution of trees in industrial plantations, while in ecological rehabilitation plantation, the trees are planted in clumps or some other configuration that will be more effective in producing the desired conditions for ecosystem development (Evans and Turnbull 2004). For restoration planting, the primary objective will be ecological in nature, whereas for rehabilitation planting, the objective is increasing the structural complexity and biodiversity of the plantation, in addition to the potential for making economic return.

Because the practises used for restoration and rehabilitation plantation depend very much on the nature of the degradation at a specific site, the results can be variable. Thus, there is a need of criteria and indicators to judge whether the plantation is achieving its objectives (Evans and Turnbull 2004). Due to differences in forest type, environmental conditions, time since planting began, it is not possible to have a universal set of criteria and indicators. Potential indicators for assessing the success of restoration plantation are suggested by Lamb (1993), like the presence of a litter layer, and a flora and fauna close to the original state of the undisturbed or untouched forest ecosystem.

For the management of severely degraded secondary forests, the common practise is to establish plantations after clear-cutting. Areas that are poorly stocked and could take very long to naturally regenerate are usually developed into plantations. This method usually leads to land-use change from natural forest to forest monoculture or mixed plantation. The advantage of this method is that it is used to salvage the secondary forest, by preventing them from being converted to agricultural land or other land-use forms and thus ensuring that the land remains under forest cover. In many tropical countries, forest plantations are usually established within forest reserves by completely clearing degraded or secondary natural forests (Chen et al. 2004; Evans and Turnbull 2004; Onyekwelu et al. 2006). This practise usually results in land-use change from degraded natural forests to plantation forests. Chen et al. (2004) reported that about 50,000 ha of the hoop pine plantations in southeast Queensland, Australia were established on previous natural forest land. In Nigeria, virtually all the existing forest plantations, especially those within the tropical rainforest zone of southwestern Nigeria, were established on lands that once carried degraded natural forests. For a more extensive appraisal of silvicultural strategies and techniques in rehabilitation, we refer to Weber et al. (Chap. 30).

23.3.5 Post-abandonment Secondary Forest

Post-abandonment secondary forest is an unconventional secondary forest management method that is most widely used in secondary forest regeneration and management. In post-abandonment secondary forests, the forests are left to restore themselves without any deliberate human intervention or rehabilitation measures. This is attributed to the ability of secondary forests to recover and eventually return to their original "species rich" situation, even after significant degradation. Post-abandonment secondary forest is defined as a process whereby the forests are deliberately allowed to regenerate largely through natural processes after total abandonment of alternative land-use on formerly forested lands (Chokkalingam et al. 2000). This unconventional secondary forest management method has been practised in developing countries, where systematic management of natural forest is not common. The level of success for post-abandonment as a management strategy for secondary forests will require that all forms of degradation activities (e.g. timber exploitation, fuelwood and NWFP collection, fire, encroachment by farmers) must cease or be reduced to the barest minimum. If this is not done, the forest may not be able to recover. The management challenge in these forests is to maintain a certain species composition and structure in the long term and to guarantee the regeneration of the desired species.

23.4 Users and Uses

Secondary forests are becoming the predominant forest types in many tropical countries, thus they are gradually having to provide the productive (in terms of both wood and non-wood forest products) and environmental functions of primary, old-growth forests. Moreover, the relative close proximity and accessibility of secondary forests to human settlements have made them readily available as a source of wood products, such as timber, construction wood, fuelwood and charcoal, carving wood, etc. Secondary forests are used by a wide array of people including the forest dwellers, forest adjacent households, commercial producers and users of forest products, nature lovers and ecotourists. Thus if properly managed, secondary forests can provide a wide variety of goods and services to society, especially to local communities that depend on them. The main users are the sedentary group (forest adjacent households) who rely on various components of the forest all year round for their subsistence. The secondary forests provide not only wood products but also a wide range of goods and services to the local users including medicinal plants, honey, thatching grass, fodder, saplings, seeds, cultural/ceremonial sites and food (vegetables, fruits, game meat). In addition to these benefits, secondary forests also play significant roles in soil and water conservation as well as carbon sequestration. In some tropical countries, secondary forests play a very important economic role as the source of wood raw materials for some export products.

Forest land can be categorised into various uses depending on the nature of goods and services they provide. The species growing in secondary forests often have multi-purpose functions, which is an important feature to be taken into account for its management. Secondary forest use boosts local economy and provides tangible benefits to those who live in and around the forests. FAO estimates that 80% of the people in the developing world relies on NTFPs (mostly from secondary forests) for some purpose in their everyday life, thus the local communities are the principal users of secondary forests. For example, plants for medicinal uses are important components of secondary forests, thus their abundance and distribution, especially in the understorey, plays important role in rural health care delivery. The categorisation may be based on such factors as provision of food and economic empowerment for rural dwellers, maintenance of environment, provision of opportunity for recreation activities, habitat for wildlife, watershed protection, general conservation including minimisation of soil erosion and the production of wood for various uses. For each category of use, it might be subdivided, or combined to be used in the same area of forest land.

Building on the important uses of secondary forest, ITTO (2002) and Müller (2002) identified the following benefits of secondary forests:

- Fallow within shifting cultivation systems, which are often an integral component of small farmers' agricultural systems for the regeneration of soil fertility and the containment of pests and diseases.
- Fuelwood and charcoal, which are the primary energy sources for many rural people in tropical regions, are important secondary forest products.
- Non-timber forest products (NTFPs) (e.g. bamboo, rattan, edible fruits, nuts, leaves, medicinal plants, game, etc.), which are harvested for their economic, social and nutritional importance.
- Wood for local needs (house-building, posts) and for sale (sawn wood, veneer wood, industrial wood).
- Environmental services such as protection of soil erosion; regulation of water regime and reduce water loss.
- Through run-off on hillsides; fixation and storage of significant amounts of carbon, thus contributing to mitigation of global warming; refuges for biodiversity and biological corridors in fragmented/agricultural landscapes; contributing to reducing fire risk; and conservation of genetic resources.
- The use of secondary forests may reduce pressure on primary forests, thus reducing deforestation rates. However, this only applies if the products from the secondary forests are suitable for the same uses as those derived from primary forests, if the financial rewards are comparable, and if economic conditions do not encourage the simultaneous use of both types of forest.
- Recreational activities.

23.5 Conclusions

One of the daunting challenges is the sustainability of the utilisation of the vast resources from secondary forests. Consequently, appropriate management options that take into account the peculiarities of secondary forests and the needs of people who depend on their resources must to be identified and implemented. The selected management strategies should be based on a sound analysis of the general social, economic, institutional, and ecological context. Hence, the particular ecological and socioeconomic criteria and indicators adopted should be linked to site-specific objectives and goals (ITTO 2002).

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Chapter 24
Secondary Forests and Fuel Wood Utilization in Africa

Joseph Adeola Fuwape

Abstract The utilization of fuel wood as a major source of primary energy in Africa was reviewed with a focus on its management in secondary forests. Fuel wood sources, the state of regional energy crises, and the silvicultural management of fuel wood in secondary forests are all factors that were considered in this study. About 95% of fuel wood consumed in the continent is from secondary forests and is used for both domestic and industrial purposes. Its consumption varies with vegetation distribution, population density, demographic characteristics, and the state of local energy crises.

Preventing the loss of forest resources through deforestation is the greatest challenge of fuel wood utilization. Silvicultural practices for management of fuel wood include natural regeneration, coppice, coppice with standard, and agroforestry techniques. The great demand for wood production through these methods had led to an increase in intensive forest management and the adoption of community participatory approaches in the formulation and implementation of forest policy.

Keywords Coppice · Fuel wood · Natural regeneration · Secondary forest · Sprout

24.1 Introduction

Fuel wood is the main source of heat energy in rural communities in Africa (FAO 2000). The volume of fuel wood consumption in the continent increased from 399.5 million m^3 in 1980 to 635.1 million m^3 in 2000 (FAO 2003). With the projection that the number of people who rely on fuel wood as major energy source will increase from 583 million in 2000 to 820 million people in 2020 (Arnold et al. 2003), it is expected that fuel wood consumption will increase to 850.2 million m^3 in 2020 (FAO 2003).

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Table 24.1 Classification of some African countries according to their dependence on fuel wood

Fuel wood situation	Percentage contribution of fuel wood to total energy consumption		
	Heavy reliance (over 70%)	Medium reliance (30–70%)	Low reliance (less than 30%)
Acute scarcity	Botswana, Burkina Faso, Burundi, Cape Verde, Chad, Comoros Djibouti, Kenya, Mauritania, Sudan, Somalia	Ethiopia, Mauritius, Swaziland	
Deficit	Angola, Benin, Gambia, Guinea, Madagascar, Malawi, Mozambique, Togo, Uganda, Congo, Zambia, Tanzania	Nigeria, Cameroon, Senegal	Algeria, Morocco, South Africa, Egypt
Satisfactory	Ivory Coast, Ghana, Sierra Leone, Zimbabwe		

Source: FAO (1986, 2000, 2003)

Besides being the dominant fuel in rural households, fuel wood is also used in rural and urban cottage industries to create energy for the production of tobacco, brick tiles, sugar, bread, alcohol, oil palm, and other processed food items (Fuwape 2008). The fuel wood trade has also contributed to the socioeconomic development of the local people. Although energy consumption patterns in Africa appear to be region specific, it varies with vegetation distribution, population density, demographic characteristics, level of income, and the availability of biomass resources and alternative fuels. Thus, the percentage of fuel wood in total national primary energy varies from one country to the other. The contribution of fuel wood to total primary energy is very high in countries in East and Central Africa where it constitutes 75–86% of total energy. In Kenya, fuel wood accounted for 78% of total energy consumption and 82% in Tanzania (Karekezi and Kithyoma 2002). For countries in the rainforest and savanna regions of West Africa, fuel wood contributes 60–70% of total primary energy, while in North Africa and nontropical parts of Southern Africa, a low range of 6–12% was reported (FAO 2000). The relative importance of fuel wood in relation to energy consumption in African countries is presented in Table 24.1. The profile of fuel wood consumption in Africa revealed that the household sector was responsible for 86% of the consumption followed by 10% for the industrial sector and 4% for transportation and commercial sectors (FAO 2000, 2003; Fuwape 2007).

24.2 Fuel Wood Crisis in Africa

The availability of fuel wood varies with forest type. Central Africa accounts for the highest proportion of African forest cover with 2.69 million km^2, followed by 680,000 km^2 forest cover in West Africa, 250,000 km^2 in East Africa, 58,665 km^2 in Southern Africa, and 15,694 km^2 in North Africa (FAO 2000). Thus, the largest supply of fuel wood has come from rainforest and savanna

Table 24.2 Consumption of fuel wood in selected ten Africa countries in 1994, 1996 and projection of fuel wood consumption till 2030

	1994	1996	2000	2010	2020	2030
Nigeria	118.8	121.9	143.4	157.7	169.6	174.9
Ethiopia	56.6	61.2	68.4	75.5	81.8	84.7
South Africa	42.5	19.1	51.3	56.7	61.4	63.6
Tanzania	42.2	43.6	51.0	56.3	61.0	63.2
Congo DR	38.0	46.1	45.9	50.7	54.9	56.9
Uganda	29.8	30.5	36.0	39.7	43.0	44.5
Kenya	28.7	19.4	34.7	38.3	41.5	43.0
Sudan	26.3	28.1	31.8	35.1	38.0	39.4
Mozambique	21.7	22.9	26.2	28.9	31.3	32.4
Ivory coast	19.0	20.5	23.0	25.4	27.5	28.5

Source: FAO (2000), Arnold and Person (2003)

vegetations while the least comes from desert regions in North and Southern Africa. This is confirmation that there is variation in biomass production between the moist and dry natural forest zones.

The combined effects of population increase in certain areas and the variability of biomass production have culminated in situations of acute scarcity and deficit supplies of fuel wood. These factors have created an energy crisis in different parts of Africa.

The unprecedented increasing demand for fuel wood (Table 24.2) in response to a high rate of population growth (UN 2007) has resulted in scarcity of fuel wood in arid and semiarid regions of the continent. Acute scarcities of fuel wood exist in areas where availability is lower than the quantity required to meet minimum fuel needs. For example, in arid, subarid, and mountainous areas, where the fuel need is between 1.4 and 1.9 m^3/inhabitat/year, the available resources varied from 0.1 to 0.7 m^3/inhabitat/year (FAO 1986). About 50 million people living in Botswana, Burkina Faso, Burundi, Cape Verde, Chad, Djibouti, Ethiopia, Lesotho, Somalia, Mali, and Mauritania (Table 24.1) are affected by acute fuel wood shortages.

24.3 Sources of Fuel Wood

Secondary forests serve as the major source of fuel wood. More than 95% of fuel wood utilized in Africa is collected from secondary forests that developed through natural regeneration after the disturbance of the original vegetation (Chokkalingam and De Jang 2001; FAO 2001). The destruction of virgin forests by indiscriminate logging, fuel wood collection, shifting cultivation, animal grazing, thunderstorms, flooding, or wind fall, often result in plant succession processes that are responsible for the development of secondary forests (Bearer et al. 2008). At the early stage of the succession process, the number of plant species per hectare is often very high. After the forest canopy has been opened, the pioneer trees and sprouts of coppiced trees quickly emerge. This is followed by growth of shade-tolerant species.

Biomass accumulation and diversity of fuel wood plant species vary with the physiognomy of the original vegetation, rate of plant succession, and the type of secondary forest.

The major types of secondary forests in Africa include post-extraction secondary forest, swidden fallow secondary forest, rehabilitated secondary forest, post-abandoned secondary forest, and post-fire secondary forest (Blay 2002).

The swidden fallow secondary forest develops as a result of intensive shifting cultivation in an original fertile primary forest. This forest covers extensive areas in Africa and is characterized by scattered trees interspersed with grasses. The structure of the swidden forest varies with the initial vegetation. In the rainforest or moist forest belt in West and Central Africa, the swidden forests are more heterogeneous in canopy and understory. They contain few large remnant trees and more saplings and climbers than the primary forest. The regeneration of trees in the swidden fallow forest is usually slow because of depleted soil nutrients (Amiebenomo 2002). The preferred firewood species in fallow forests varies with vegetation zone. Stocking of firewood species depends on the stage of successional growth of the fallow forest. In Malawi, *Julbernardia paniculata* and *Bracystegia* species account for 52% of the stocking (Abbot and Lowore 1999).

In the swidden forest, firewood is often collected from wood lots in free areas, fallow land, community wood lots, shelter belts, trees and shrubs scattered along roads, trees used for life fences, hedges, and as wind breaks. Although several varieties of woody plants are collected as fuel wood, there is preference for tree species that have good combustion characteristics such as a high heating value, fast rate of ignition, long flame duration, and low ash content (Fuwape et al. 2001). Generally, high wood density tree species are preferred to low density wood materials because they burn longer and release a greater quantity of heat per unit weight.

The quantity of firewood extracted from the different sources varies from one country to another. Post extraction secondary forests are the successions created after the destructive timber extraction process of the original forest. The structure and stocking of firewood species in post extraction forests depend on the silvicultural technique adopted during the logging operation, whether it was a polycyclic or monocyclic system. In polycyclic systems, selective felling is adopted for extraction of selected timber species, while all merchantable logs are removed at the same time in monocyclic systems. The damage to the forest is more intense in monocyclic than polycyclic systems. The firewood species in the post extraction forest depend on the initial composition of the natural forest. Examples of the firewood found in West Africa include *Acacia albida, Albizia zygia, Cassia siamea, Prosopis africana,* and *Terrminalia laxifolia*. In Central and Southern Africa, the common firewood species are *Acacia karroo, Trema orientalis, Virgilia divarcata, Ptaeroxylon obliquum, Milletia grandis,* and *Rapanea melanophloeos.*

The post fire secondary forest is mainly found in the arid and semiarid vegetation belts. This type of forest emerges after the initial primary forest has been destroyed by fire. The degraded forest is usually colonized by pioneer species, which are aggressive and have the ability to survive in hostile terrains. Some of the preferred fuel wood species in the post fire forest in West, North, and Central Africa include *Acacia, Auriculiformis, Pterocarpus, Sesbania, Cassia,* and *Prosopis* species.

In the rehabilitated secondary forest, regeneration is enhanced by protection of the degraded forest to facilitate natural growth. The protection is enforced through legislation that restricts and forbids human interference and activities within the forest. Forest rangers are usually employed to guard the rehabilitated forest and forest reserves. In the rainforest belt, this type of secondary vegetation is characterized by a dominant layer of scattered tall deciduous trees with a dense undergrowth, whereas in the savanna, they contain few trees, a lot of woody shrubs, and dense undergrowth and grasses. The common fuel woods in this zone include *Acacia, Cassia, Albizia,* and *Vitex*.

24.4 Silvicultural Management of Secondary Forests

The silvicultural techniques adopted for fuel wood production in secondary forests depend on the nature of vegetation, the type of secondary forest, amount of annual rainfall, site characteristics, human resources, socioeconomic issues, and environmental considerations.

The major silvicultural tools in practice across the continent include natural regeneration, coppice, coppice with standard, and enrichment planting.

In a natural regeneration system, the forest is allowed to recover through the development of sprouts from tree stumps and the natural emergence of wildings and seedlings of shade-tolerant trees. Management of the forest involves only boundary maintenance and fire tracing. Full recovery of forest structure often takes a long time because of slow growth rate of most indigenous tropical species. At the maturity stage, self-thinning processes often occur where there are closed forest canopies. Self-thinning is responsible for 20–30% of the total above-ground dead branches that are collected for firewood.

Coppicing is also used as a traditional system of forest management for the production of short rotation fuel wood biomass. The technique involves cutting trees at low stump height. The numbers of sprouts per stump is often reduced to two during the first thinning operation. Thereafter the sprouts are allowed to grow to rotation age depending on the type of species and site conditions. A coppiced forest is harvested in coupes on a rotational basis. The cycle length is influenced by the performance of different plant species at different sites. In the savannah region, coppicing of *Cassia* and *Albizia* is done in cycles varying from 5 to 10 years. Biomass accumulation of coppiced stands varies with species; an average annual increment of 45 m^3/ha was recorded for Eucalyptus in northern Nigeria (FAO 2001). Slow growth rates were, however, reported for *J. paniculata*, when compared with *Combretum apiculatum*, *Combretum molle*, and *Brachystegia thonningii* in Malawi (Abbot and Lowore 1999).

Coppice systems with a short rotation period and high biomass accumulation were reported for *Gliricidia*, *Leucaena*, *Cassia*, and *Albizia* species in the lowland rain forests of Nigeria, Ghana, Kenya, Uganda, and Ivory Coast (Onyekwelu and Akindele 2006).

Coppice with standard, which is a variation of the coppicing system, has also been used in managing secondary forests. In this technique, the forests are divided into periodic coupes according to their maturity. Selective felling of trees is carried out in cycles with respect to when the blocks reach a predefined density requirement. Biomass yield varies with different plant species and site quality. Nygard et al. (2004) reported that woody growth of selected fuel wood species at five different sites in Burkina Faso ranged from 0.745 to 3.0 m^3/ha per year. The dominant fuel wood species in Burkina Faso and Ouagadougou are *Anogeissus leocarpus, Acacia seyal, Combretum nigircans,* and *Sclerocarya birrea*. In Malawi, *Amythethophylla* and *B. thonningii* attained an average height of 3 m and basal diameter of 4.5 cm in 4 years (Abbot and Lowore 1999).

Fuel wood crops have been effectively grown with food crops through agroforestry practices in Kenya, Tanzania, Uganda, Burundi, Uganda, Zambia, Nigeria, Togo, Ivory Coast, Mali, Nigeria, Ghana, and Gambia. The species that are often chosen for fuel wood production in agroforestry farms are the desirable species that have fast growth rates, good combustion properties, good nitrogen fixing characteristics, and the ability to adapt well to site conditions. In this type of practice, effective management of components takes cognizance of characteristics of the tree species, planting density, and spatial arrangement. Tree crowns and roots are subjected to systematic thinning and pruning operations to minimize interference and optimize yield of food crops. Selective pruning of the tree crowns facilitates the growth of associated intercrop while below the ground, competition of roots is minimized by pruning-induced root die-back (Cannel 1983).

It has been reported that the trees, which prove most useful for fuel wood in agroforestry practices are those that naturally colonize deforested areas. The nature of such pioneer species endows it with adaptability, aggressiveness, and the ability to survive hostile terrain. Some of the species that have been successfully grown in swidden and rehabilitated secondary forests include *Acacia, Aurianliformis, Casuarinas*, *Gliricidia, Gmelina, Sesbania, Cassia, Eucalyptus, Teraninalia,* and *Prosopis* species (Fuwape and Akindele 1997). The favored fast-growing fuel wood species in humid tropical rain forests include *Terminalia catappa, Gliricidia sepium, Gmelina arborea, Casuarina equisetifolia*, and *Sesbania grundifolia*, while in savanna and semiarid regions, *Acacia nilotical Tamarindus indica*, *C. siamea*, *Eucalyptus* species, and *Prosopis* species are preferred (Nair 1989).

The biomass yield varies with the type of fuel wood species, site quality, and silvicultural practice. In agroforestry systems such as alley cropping that involves coppicing, the rate of biomass recovery is rapid because the stump easily generates sprouts. In monocyclic systems where regeneration depends on young wildlings (undergrowth) and seeds, woody biomass recovery rates are slow due to the slow growth rate of indigenous tree species.

The rate of natural regeneration and growth of sprouts varies with the vegetation zones in West Africa, Central Africa, East Africa, Southern Africa, and North Africa. In the moist forests of West and Central Africa, the broadleaf trees sprout and grow faster than species in the savanna region. The densities of sprouts and natural regeneration are higher in the moist forests of West and Central Africa than in dry forests or semiarid and arid vegetation types in North and Southern Africa.

24.5 Conclusions

The secondary forest is the main source of fuel wood in Africa. The swidden secondary forest, post fire, post extraction, and rehabilitated forests are the major types of secondary forests in the continent. The structure of the secondary forests is different from that of primary forests in terms of heterogeneity of canopy and plant diversity. The quantity of fuel wood available in the secondary forest varies with vegetation zones. The growth rate and sprout densities were higher in the rainforest zones of West and Central Africa, but low in savanna and dry vegetation regions in North, East, and Southern Africa. Since fuel wood will continue to be the most reliable source of energy in rural communities in Africa for a long time to come, there is need for radical review of forest policy by governments in African countries. The forest policy review should stimulate programs of action required to improve forest biomass production. There should be a concerted political will and government commitment toward integrated participatory management approaches that engage the members of the community in planning, designing, establishing, and maintaining the secondary forests. African leaders should develop national policy and programs which attach priority and commitment to production and utilization of fuel wood. People living in the rural community should be encouraged to manage the secondary forests through community approach participatory systems. A forest-based energy sector that is well planned will reduce the poverty level by providing employment opportunities and reducing energy costs and greenhouse gas emissions.

The advantages of silvicultural treatments in secondary forests for fuel wood utilization include sustainability in fuel wood supply, reforestation of disturbed and destroyed forest cover, and a reduction of inadequate supplies of fuel wood. The disadvantages include human interference with the natural regeneration of secondary forests and introduction of exotic plant species in some secondary forests in Africa. Governments of different African countries should provide more funding and trained manpower to enhance developments in secondary forests. Policies and regulations concerning the rehabilitation and protection of secondary forests should be implemented in order to optimize the benefits of all products and services from the secondary forests.

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Chapter 25
Rehabilitation of Degraded Natural Forests by Enrichment Planting of Four Native Species in Ethiopian Highlands

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Abstract Many natural forests in the highlands of Ethiopia are heavily degraded due to unsustainable forest management. Therefore, a study was conducted in the Munessa-Shashemene forest to design sustainable forest management strategies. The present study investigates the survival, growth, and photosynthetic performance of enrichment planting of four species (*Cordia africana*, *Juniperus procera*, *Prunus africana*, and *Podocarpus falcatus*). Planting was undertaken in gaps in the degraded natural forest. Results indicated that survival was different among species. Two years after planting, only 23% of the *C. africana* and *P. africana* seedlings had survived, while *J. procera* and *P. falcatus* showed higher survival rates of 76 and 47%, respectively. The development of the height over the first 2-year observation period was reasonable for *J. procera* and *P. falcatus*. Inadequate height development was registered for *P. africana*, which was strongly affected by browsing and for *C. Africana*, which suffered from drought. *P. falcatus* exhibited the lowest photosynthesis and transpiration rates, which were associated with the highest water use efficiency of all the four species. Enrichment planting especially with *J. procera* and *P. falcatus* can be recommended to restore the degraded natural forests in Ethiopian highlands.

Keywords Highland of Ethiopia · Enrichment planting · Survival · Height growth · Water use efficiency · *Juniperus procera* · *Cordia africana* · *Prunus africana* · *Podocarpus falcatus*

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25.1 Introduction

The Ethiopian highlands (1,500 m asl, above sea level) constitute around 45% of the country's total area and is inhabited by about four-fifths of the human and two-thirds of the livestock population. They constitute the largest mountain complex in Africa. Over 14 million ha (hectares) (i.e., 27% of the area) in the highlands is seriously degraded (Yirdaw 2002). At present, due to deforestation caused by high population pressures, many areas of the highlands are covered with wooded grasslands in which secondary tree species dominate (Friis 1992; Yirdaw 2002).

The same holds true for the Munessa dry Afromontane forest, which is heavily degraded and characterized by many gaps. It has been disturbed, either by illegal logging and/or subsistence farming by the local people. What remains is a fragmented, logged over, and despoiled forest situated among a growing rural population with ever sharper sights on it as their next livelihood option. However, until now there has been a lack of scientific studies in the highlands of Ethiopia, which can directly contribute to the sustainable management of the remaining natural forest resources by silvicultural means.

Enrichment planting is commonly used for enhancing the density of desired tree species in secondary forests and is considered as an option to restore the degraded tropical and subtropical forests. To test this option for degraded forests in the Ethiopian highlands, the project "Silvicultural contributions towards sustainable management and conservation of forest resources in the highlands of Ethiopia" was initiated in 2006. It constitutes an integral part of the project funded by the DFG-German Research Council, "Functional Ecology and Sustainable Management of Munessa Forest, Ethiopia," conducted in the Munessa Shashemene forest.

The overall goal of the silviculture project was to provide scientific knowledge on how natural forests in the highlands of Ethiopia can be restored. The present study was conducted to investigate whether enrichment planting with economically and ecologically important indigenous tree species is an appropriate silvicultural measure for the rehabilitation of degraded natural forests (Girma et al. 2010).

25.2 Materials and Methods

25.2.1 Study Area

The study is conducted in the Degaga block (38°53′ East and 7°27′ North) in the Munessa Shashemene Forest. The Munessa Forest has an estimated area of 23,000 ha, of which approximately 11,000 ha is covered by the natural forest at an altitude ranging from 2,100 to 2,700 m asl (Silvanova 1996). The natural forest is a mixed Podocarpus forest (Friis 1986; Cheffey 1978). The study area belongs to warm subhumid and cold humid agro-ecological zones of Ethiopia with a mean annual rainfall of 1,343 mm and temperature of 15°C. The dry season extends from

October to May with an average rainfall as low as 10 mm. A thorough investigation on the soils of the study area was made and according to FAO soil classifications, they were characterized as Humic Umbrisols (Fritzsche et al. 2006; Lundgren 1971).

25.2.2 Layout of the Experiment and Planting Material

Selection of the experimental units was based on a map of the natural forest in the study area, from which five blocks, each equalling 5 ha in size, were selected. In each block, four gaps were randomly selected using a line transect technique (De Vries 1979; Otta and Juday 2002). A gap is an opening in the natural forest, which has a size of at least 450 m^2, but not larger than 1,800 m^2 and is not stocked by ecologically or economically valuable woody vegetation. Each gap represents an experimental plot (Fig. 25.1), which was protected from livestock by fencing. The layout of the experiment is a randomized block design with four enrichment planting levels. The four plots in each block were randomly assigned to the four enrichment planting levels. These include four valuable native species: *Juniperus procera Endl.*, *Cordia africana Lam.*, *Prunus africana (Hook. f.) Kalkm.*, and *Podocarpus falcatus (Thunb.) Mirb.*

The four species vary in their shade tolerance. *C. africana* represent shade intolerant early secondary forest trees whereas *P. africana* and *J. procera* represent intermediate tolerant trees. *P. falcatus* represent shade tolerant late primary forest trees. Seedlings were raised in a local nursery. The planting material of the four species at the time of planting was 1 year old. The quality of the four species at planting was fairly good. The seedlings of *P. africana* and *J. procera* had the highest average height of 56.13 ± 1.70 and 53.03 ± 0.79 cm, respectively. The initial average height of *C. africana* was 43.33 ± 0.76 cm, whereas *P. falcatus* with 26.23 ± 0.51 cm were the shortest seedlings in height.

In each enrichment planting plot, 196 seedlings (Fig. 25.1) were planted in mid-June 2007. Seedling spacing was variable, as the size of the gaps was not all the

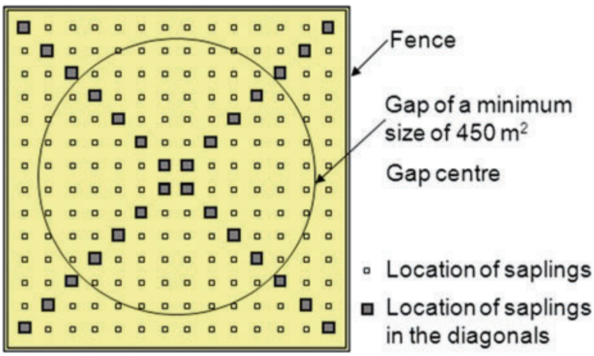

Fig. 25.1 Layout of a plot

same. However, the minimum distance between two adjacent plants was 1.95 m. In each plot, the arrangement of the seedlings was undertaken in a way to allow the observation of 28 individuals placed in the diagonals at the four cardinal directions (Fig. 25.1). So, at each direction, there were seven different distance levels to the gap center. As the seedling spacing was not the same for all plots, the relative distance (distance of an individual placed in the diagonal to gap center × 100/gap radius) is constant for all plots having the following levels: 21.8, 43.5, 65.3, 87.0, 108.8, 130.5, and 152.3%.

25.2.3 Data Collection

Survey of the enrichment plantings including survival, total height, root collar diameter, and damage (browsing, drought) was undertaken at the 3rd, 12th, and 24th month after planting. Light measurements at each seedling position in the diagonal were made using a plant canopy analyser (LAI-2000; LI-COR, Lincoln, Nebraska). Data related to net photosynthesis and transpiration rate were collected 8 months after planting in February 2008 from four selected plots, each representing one tree species. Sampling was based on a young leaf from the upper one third of the seven individuals located in the diagonal at the north exposition. Six measurements [at 7 a.m., 9 a.m., 1 a.m. (13), 3 a.m. (15), 5 p.m. (17), and 7 p.m. (19)] per each species were undertaken using LC Pro – PS (ADC Bioscientific Ltd. UK 2002). Water use efficiency was calculated for each seedling at the different measurement times [(net CO_2 fixed/H_2O loss or assimilation rate in μmol m^{-2} s^{-1})/transpiration rate in μmol m^{-2} s^{-1}].

25.3 Results and Discussion

Survival rate was significantly different among species (Fig. 25.2). Two years after planting, *J. procera* had the highest survival of 76%, followed by *P. falcatus*, which had a survival rate of 47%. Over the same time frame, less than a quarter of the *P. africana* and *C. africana* seedlings had survived. Mortality of *C. africana* seedlings started rapidly after planting. More than one-third of the seedlings died over the first 3 months after planting and this high mortality continued to reach 71% 1 year after planting. The same mortality rate was registered for *P. africana* after 1 year. Nearly all dead *C. africana* seedlings had shriveled leaves, indicating drought to be the main cause of mortality. The same holds true for *J. procera* and *P. falcatus*. However, browsing was the main cause for the high mortality of the *P. africana* individuals as the fence was not dense enough for some animals (e.g. the bushbuck).

Tropical forest seedling survival under different light conditions has been investigated in the last two decades (Chazdon et al. 2009; Swaine and Whitriore

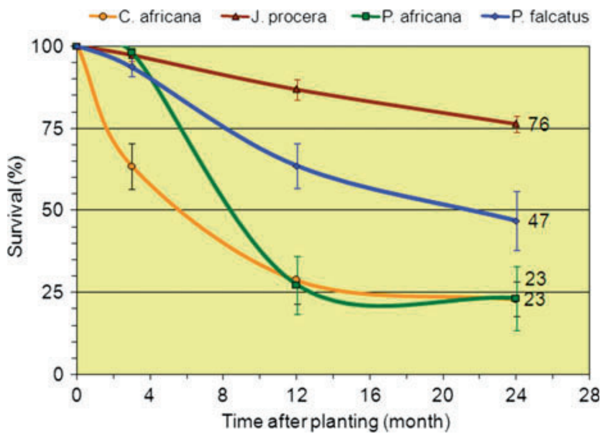

Fig. 25.2 Survival rate of the enrichment planting over the observation period of 2 years

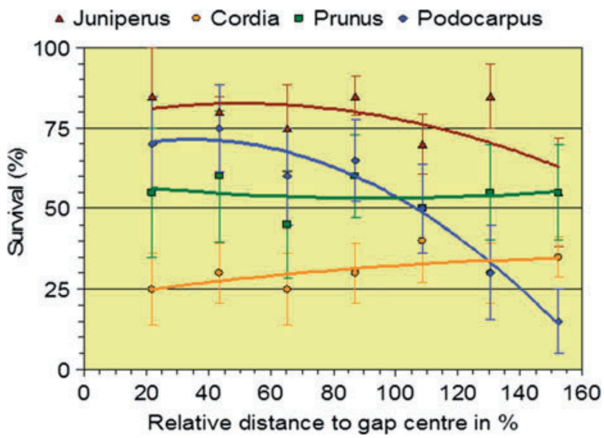

Fig. 25.3 Relationship between survival rate of the enrichment planting and the relative distance to gap center (<80 = within the gap, ≥80 – <120 = at the gap edge, and ≥120 = under the canopy of the mature stand)

1988). There are different reports regarding seedling survival percentages in tropical forests and other forest types. In the presented study, wider differences in survival range were found. The survival percentages of the four planted species (23–76%, Fig. 25.2) were comparable to the results reported from gap planting experiments in the Munessa Shashemene forest by Tesfaye (2007). He reported 29% to 96% survival among seven studied species' seedlings. Ådjers et al. (1995) also registered survival rates between 40 and 85% for *Shorea johorensis*, *Shorea leprosula,* and *Shorea parvifolia* 2 years after planting.

P. falcatus was the only species which showed a clear reaction to the different light conditions of the gap (Fig. 25.3). Survival rates of *P. falcatus* decreased dramatically under the canopy of the mature stand.

The height development of the enrichment planting over the observation period of 2 years is presented in Fig. 25.4. It can be seen that the development of the height of *J. procera* and *P. falcatus* was reasonable. However, the increase in height was

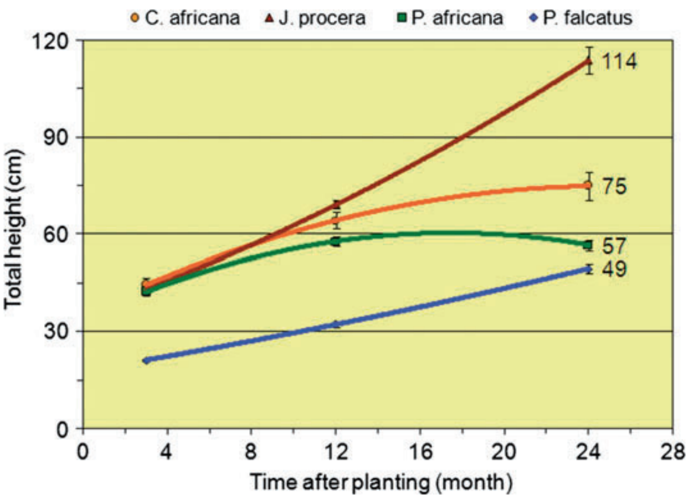

Fig. 25.4 Development of the mean total height of the enrichment planting over the observation period

less in *C. africana* compared to *J. procera*. The main shoots of the *C. africana* plants were shrivelled because of drought. The height increment of *P. falcatus* and *P. africana* was slow. The height development of *P. africana* was noticeably disturbed due to herbivore (mainly bushbuck).

Over the observation period of 2 years, the highest annual height increment was achieved by *J. procera* (40.6 ± 2.4 cm a^{-1}), followed by *C. africana* (17.5 ± 2.5 cm a^{-1}), and then by *P. falcatus* (16.0 ± 0.8 cm a^{-1}), while *P. africana* had the lowest increment (7.9 ± 0.9 cm a^{-1}). Similar growth rates of seedlings were found in tropical forests by other authors (e.g. Denslow 1987; Denslow and Hartshorn 1994; Nicotra et al. 1999). Generally, variation in growth of tropical tree seedlings was reported due to their inherent species specific ability, but can also be related to increased light availability, reduced below ground competition, increased soil water, and/or nutrient availability. This is confirmed by many other studies (e.g. Teketay 1997; Wang et al. 2005; Tesfaye 2007; Tesfaye et al. 2010).

Results of gas exchange observation on selected saplings of the four species showed that the diurnal patterns of assimilation rate, transpiration rate, and water use efficiency (Fig. 25.5) were different among the four species. *C. africana* had peculiar pattern and reached its highest assimilation and transpiration rates at the noon hours. *P. falcatus* achieved the highest water use efficiency in the late afternoon. When comparing the cumulative pattern (Fig. 25.5), it can be seen that *C. africana* has the highest assimilation and transpiration rates, followed by *P. africana*, *J. procera*, and *P. falcatus*. This indicates that *C. africana*, in comparison to the other three species, is more susceptible to drought since it transpires much of the available water during the dry season. On the other hand, *P. falcatus* had exhibited lower photosynthesis and lower transpiration rates, which resulted in

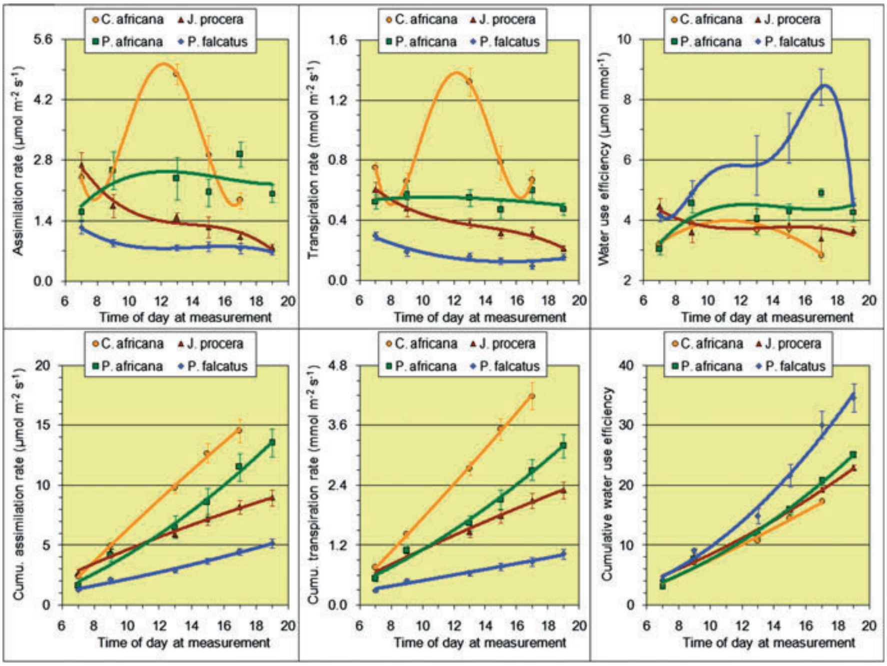

Fig. 25.5 Arithmetic mean (*top*: diurnal pattern, *bottom*: cumulative pattern) of photosynthesis, transpiration, and water use efficiency of seven individuals located in the diagonal at the north exposition, 8 months after planting in February 2008

higher water use efficiency than the other species. These results suggest that *P. falcatus* is the most drought tolerant species used in the enrichment planting. *J. procera* and *P. africana* can be considered as medium tolerant to drought. Many authors (Denslow et al. 1990; Kitajima 1996) have agreed to the poor prediction ability of plant photosynthetic measurements for their growth rates. However, in this study such measurements enable comparisons of the species' physiological plasticity.

25.4 Conclusions

The assessment of the success/failure of the enrichment plantings require long-term observation periods to gain sufficient information of their long-term survival and growth. However, the results of the short-term observation period were promising: (1) enrichment plantings have potential to be practiced in the highlands of Ethiopia to restore the degraded natural forest, in particular *P. falcatus* and *J. procera* are appropriate species for enrichment planting; (2) *P. falcatus* should be planted in open gaps, as its survival rate is low under dense canopy shelter; (3) the success of species, which proved to be susceptible to drought as *C. africana*, cannot be achieved in the event of scarcity of rainfall particularly at an early stage after planting; (4) in addition, the results clearly demonstrated that the success of species susceptible to browsing, such as *P. africana*, can only by achieved by intensive protection measures.

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Chapter 26
Sustainable Timber Harvesting in Fragmented Secondary Forests in Paraguay? An Inquiry Through Modelling

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Abstract Secondary forests are an increasingly important component in the neotropical landscape matrix. As deforestation rates in many Latin American countries continue to be high, secondary forest resources may contribute to timber supply. Using a process-based forest growth model, we studied the prospect of secondary forests in eastern Paraguay for sustainable timber harvesting under different logging and fragmentation scenarios. The timber yield is low under all scenarios (0.75–1.73 m^3 ha^{-1} $year^{-1}$). Only silvicultural treatments may increase the commercial value of secondary forests. For large landholdings, labour-and capital-extensive land-use systems such as secondary forest management may be an option to diversify farm income. This particularly holds for Paraguay facing an increasing scarcity of timber resources.

26.1 Introduction

To date old-growth forests are the major focus of research on forest fragmentation in the neotropics. Most of these studies deal with ecological issues (e.g. Laurance et al. 2004; Laurance 2008; Broadbent et al. 2008). Secondary forests on abandoned agricultural land play an increasing role in the landscape matrix (Helmer 2000;

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Kammesheidt 2002), but have received much less attention, although constituting highly fragmented components with remnants of primary vegetation. In general, secondary forests are found on privately owned farmland so that pure conservation strategies are unlikely to be successful, particularly because of the lack of incentives such as payments for environmental services. Thus the management of secondary forests for timber and non-timber forest products may be an option.

To enhance the general understanding of stand dynamics, we assess first the long-term impact of different fragmentation scenarios on the bole volume and species composition of primary and secondary forests without logging operations. Hereafter we study the timber use option by asking (1) what are the most promising logging strategies (polycyclic vs. monocyclic system), considering the sustainability of timber yields in fragmented secondary forests? and (2) to what extent does the combined effect of logging and fragmentation alter species composition? Due to the lack of long-term empirical data, we use a process-based forest growth model to answer these questions.

26.2 Materials and Methods

26.2.1 Study Site and Data Collection

The study was conducted in the District of Choré (24°06′–17′S, 56°22′–34′W), eastern Paraguay. The gently undulating landscape ranges from 80 to 230 m asl. Annual rainfall is about 1,300 mm with a dry period between May and August; mean annual temperature is 22.4°C. The primary vegetation cover is “sub-tropical moist forest” (Hueck 1966) with an average canopy height of 25–30 m. Mean bole volume is 160 m^3 ha^{-1}. On 0.4 ha a total number of 42 tree species $\geq$5 cm dbh (diameter at breast height) were recorded; the most prominent families were Myrtaceae, Sapotaceae, and Rutaceae (Kammesheidt 1997). Soils are nutrient-poor Oxisols. In 1988, at the time of data collection, primary forests were already confined to ridges. Besides, the man-made landscape matrix consisted of differently aged secondary forests (up to 15 years old), regrowing on land previously cleared for tobacco or cotton cultivation, cropland, and pasture.

For modelling, the data of a primary forest and two differently aged secondary forests (3- and 10-year-old), respectively, were chosen. The sampling area was 0.2 ha for each of the three stands. Four plots of 500 m^2 were randomly established. Trees $\geq$5 cm dbh were measured and identified; saplings (1–4.9 cm dbh) were counted and identified in 100-m^2 sub-plots nested in the major sampling unit. Remnant trees of the original forest were absent in the secondary forest stands. Further details about the land use history and sampling methods are found in Kammesheidt et al. (2002).

26.2.2 Plant Functional Types

Tree species were categorised into eight plant functional types according to successional status and maximum attainable height using the concept of Köhler et al. (2000). The grouping is based on (1) the abundance pattern of species in the inventory, (2) autecological information given in Lopez et al. (1987), and (3) additional field observations (see species list: http://www.uni-kassel.de/usf/archiv/dokumente/specieslists/paraguay/paraguay_pft.pdf). All large and medium-sized late and mid-successional species were assumed to be commercial, though few of them still only have a potential value. Species of the remaining ecological groups are non-commercial, but some are used as firewood and fence posts.

26.2.3 Fragmentation and Logging Scenarios

Two fragmentation scenarios were chosen (a) the simulated stand is surrounded by primary forest and, on one side, by agricultural land (i.e. "light fragmentation"); (b) the simulated stand is entirely embedded in a matrix of non-forest land (i.e. "severe fragmentation"). Two polycyclic systems with 10- and 30-year cutting cycles and a monocyclic system with logging intervals of 60 years were simulated under both fragmentation scenarios. The minimum felling diameter (MFD) is 35 cm dbh. Logging damage to the residual stand is assumed to be light because capital-poor farmers will only use on-farm facilities such as draught animals or light tractors to extract timber.

26.2.4 Simulation Model

The process-based forest growth model FORMIX3-Q simulates the spatio-temporal dynamics of an uneven-aged mixed forest (Ditzer et al. 2000; Huth and Ditzer 2000, 2001; Tietjen and Huth 2006). The model belongs to the FORMIX3 and FORMIND forest model family which has been applied to many tropical forests (e.g. Köhler et al. 2003; Köhler and Huth 2004; Rüger et al. 2008). The model describes tree competition in patches, following the gap approach and hence being inside a patch spatially non-explicit (Shugart 1984). The parameterisation and validation of the model for this simulation study is found in Kammesheidt et al. (2002).

26.3 Results

26.3.1 Stand Dynamics Without Logging Under Light Fragmentation: Scenario A

In the primary forest, bole volume was rather stable at around 160 m^3 ha^{-1} over the simulation of 300 years (Fig. 26.1a). Medium-sized late successional species (group 3) were the dominant plant functional type followed by large late successional species (group 1). In the secondary forest, bole volume reached primary forest values after some 50 years, though stabilising at these levels only in the second half of the simulation period (Fig. 26.1b). Large and medium-sized successional species accounted for the bulk of bole volume over the first 150 years simulated. In later stages, the individual share of plant functional types in bole volume reached values similar to the primary forest.

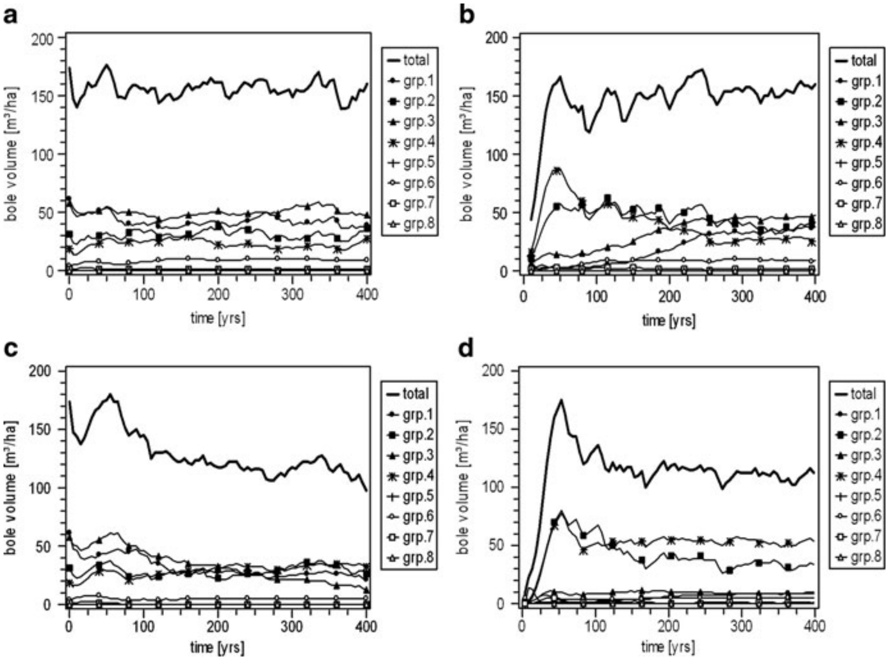


Fig. 26.1 Development of bole volume (m^3 ha^{-1}) by plant functional types (PFTs) applying different cutting cycles under different fragmentation scenarios in a primary and secondary forest, respectively, over a simulation period of 300 years. Light fragmentation scenario (**a**) Primary forest, (**b**) 10-year-old secondary forest at simulation start. Severe fragmentation scenario: (**c**) Primary forest, (**d**) 3-year-old secondary forest at simulation start. The minimum felling diameter is 35 cm. PFTs: (1) Large late successional, (2) large mid-successional, (3) medium-sized late successional, (4) medium-sized mid-successional, (5) medium-sized early successional, (6) small late successional, (7) small mid-successional, (8) small early successional

26.3.2 Stand Dynamics Without Logging Under Severe Fragmentation: Scenario B

The primary forest lost about one-third of the initial bole volume by the end of the 300 years simulation period (Fig. 26.1c). Medium-sized late successional species lost dominance with time, while large and medium-sized mid-successional species became dominant in bole volume. As under the light fragmentation scenario, the secondary forest showed a peak in bole volume after some 50 years, but dropped to lower values thereafter (Fig. 26.1d). Medium-sized and large mid-successional species gained a predominant proportion in bole volume, whereas all other plant functional types played only a minor role over the simulation period.

26.3.3 Logging Under Light Fragmentation in Secondary Forests: Scenario A

Short cutting cycles of 10 years led to a lower level of bole volume throughout the simulation (Fig. 26.2a, b). Only the monocyclic system with 60-year cutting cycles allowed bole volume to recover to initial values. Under both logging scenarios, mid-successional large and medium-sized species kept a dominant share in bole volume, though the proportion of medium-sized late successional species increased steadily.

With light harvesting intensity, first logging would be possible after 30 years (Table 26.1). The next cutting cycle had to be suspended due to understocking. The accumulated bole volume would not allow for more intensive harvesting in short cutting cycles prior to year 50. The polycyclic system with longer cutting cycles accumulated sufficient timber volume only after 60 years. Timber yield under a monocyclic system (60-year cycle) was higher than under a polycyclic system with 30-year cutting cycles. Highest overall timber yields, however, were obtained with short cutting cycles (10 years) and high extraction rates.

26.3.4 Logging Under Severe Fragmentation in Secondary Forests: Scenario B

Logging contributed to an accelerated decline of bole volume under both management scenarios, but most clearly under short cutting cycles (Fig. 26.2c, d). In contrast to fragmentation scenario A, where large and medium-sized mid-successional species were equally abundant (cf. Fig. 26.1a, b), the latter plant functional type was predominant under all cutting cycles in fragmentation scenario B. All other plant functional types played only a minor role in bole volume.

First logging was only possible after 50 years (Table 26.1). Timber yield over the whole simulation period was highest under 10-year cutting cycles and lowest under the 60-year cutting cycle, indicating that a monocyclic system is inappropriate

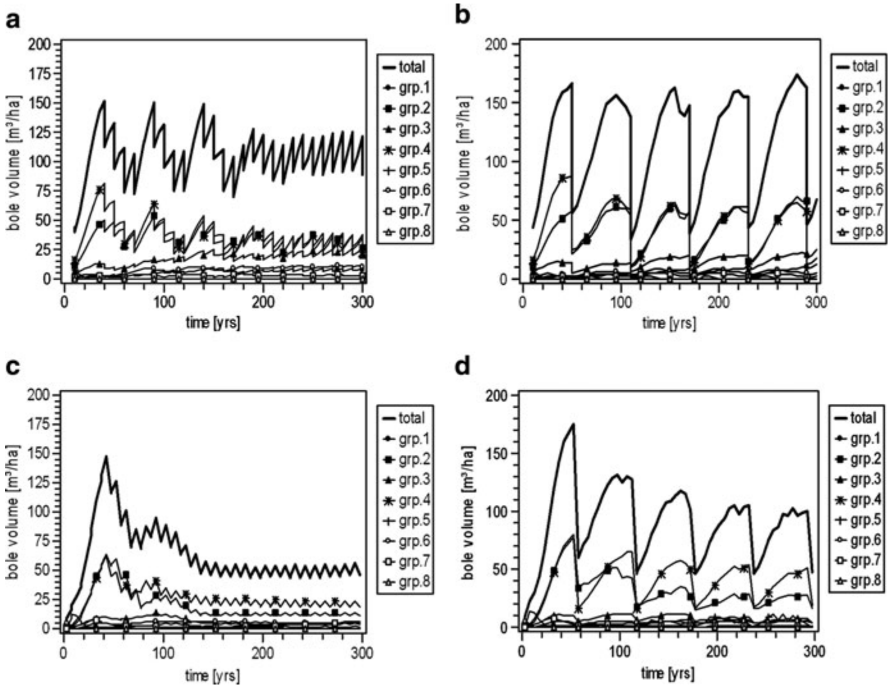


Fig. 26.2 Development of bole volume (m^3 ha^{-1}) by plant functional types (PFTs) applying different cutting cycles under different fragmentation scenarios in secondary forest stands over a simulation period of 300 years. Light fragmentation scenario: (**a**) 10-year cutting cycle (8–20 stems removed), (**b**) 60-year cutting cycle (both stands 10-year-old at simulation start). Severe fragmentation scenario: (**c**) 10-year cutting cycle (8–20 stems removed), (**d**) 60-year cutting cycle (both stands 3-year-old at simulation start). The minimum felling diameter is 35 cm. PFTs, see Fig. 26.1

Table 26.1 Timber yield (m^3 ha^{-1}; year of first timber harvest in parenthesis) of different logging scenarios (the length of cutting cycle, range of stems allowed to be cut, and the minimum felling diameter = 35 cm dbh constitute the variables) under different fragmentation scenarios (A = stand is surrounded by primary forest and, on one side, by agricultural land; B = stand is entirely embedded in a matrix of non-forest land) over a simulation period of 300 years

Cutting cycle	Harvested stems per cycle	Gross volume extracted (m^3 ha^{-1})				Times logging omitted[a]
		Yield at first logging	Yield per cycle		Total yield	
			Range	Mean±SD		
Scenario A						
10 years	3–8	3.7 (30)	8.1–11.4	8.7 ± 2.7	261.1	1
10 years	8–20	21.4 (50)	8.7–25.0	19.3 ± 6.6	520.2	2
30 years	20–30	37.7 (60)	37.3–40.8	34.4 ± 1.1	344.1	–
60 years	–[b]	75.6 (60)	75.6–83.8	79.7 ± 3.7	398.5	–
Scenario B						
10 years	3–8	8.0 (50)	8.0–10.3	9.1 ± 0.5	273.2	–
10 years	8–20	19.9 (50)	11.2–24.5	15.1 ± 4.2	393.8	–
30 years	20–30	35.2 (60)	35.2–40.4	38.2 ± 1.5	343.7	–
60 years	–[c]	64.5 (60)	28.3–64.5	42.3 ± 15.0	211.5	–

[a]Considering all regular cutting cycles beyond the first timber harvest
[b]Variable, ranging from 54 to 60 harvested stems per cycle
[c]Variable, ranging from 20 to 53 harvested stems per cycle

under severe fragmentation. In most cases, timber yield was lower than under light fragmentation (Scenario A), except for the short cutting cycle with low harvesting intensities. Differences between fragmentation scenarios were least pronounced under the 30-year cutting cycle.

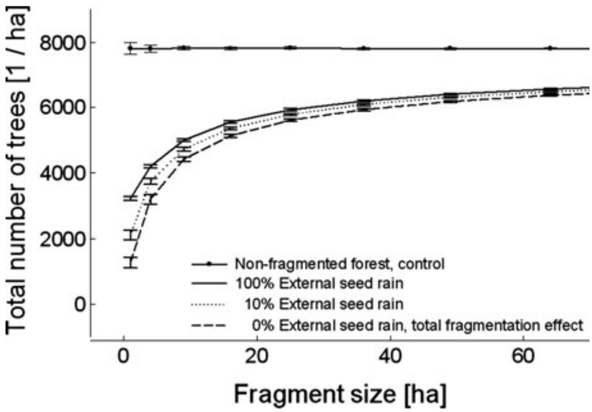
26.4 Discussion

Overall the forest growth model simulates a progressive direction of succession, although severely fragmented stands under an intensive logging regime stabilise bole volume at about half the value of the primary forest (i.e. 80 m^3 ha^{-1}). The progressive succession in the simulated stands is supported by results in older secondary forest embedded in a similar landscape matrix (e.g. Saldarriaga et al. 1988; Finegan and Delgado 2000). However, the mortality rates in the model refer to the literature of primary forest fragments (Laurance et al. 1998). Early and mid-successional species dominating secondary forests are presumably more adapted than late successional species of primary forests to better light conditions but show higher turnover rates due to their shorter life spans (Lieberman et al. 1985). Short logging cycles may partly conceal these dynamics.

The recruitment rate is the single most important source of unpredictability in the model. This applies both to the quantity of seeds and the composition of the seed pool. Both parameters are determined by the capability to adapt to altered conditions in fragmented forests related to pollination, seed dispersal, and pest regime. For example, shifts in the pollinator population and inbreeding of tree species may have a negative impact on the quantity and survival of seeds (cf. Nason et al. 1997). There is also a dramatic shift in the faunal composition (e.g. Redford 1992; Köhler et al. 2002; Banks-Leite 2009). As animals are a major vector in seed dispersal, wind-dispersed species may be favoured. Medium-sized and large trees are more common among this group of species. Within these growth forms, early and mid-successional species may benefit because of their fast growth and early fecundity (Laurance et al. 2006), and perhaps being less specialised with regard to pollinators. The model only considered the first two factors.

The seed input from outside plays a vital role in successional processes, particularly in small forest fragments. In our study, we assumed that forests surrounding the simulated stand support a constant, though reduced seed input (70% of ingrowing seedlings ha^{-1} $year^{-1}$ compared to a non-fragmented primary forest in scenario A and 10% in scenario B). Pütz et al. (unpublished data) simulated with a similar process-based forest growth model for Atlantic rainforests in Brazil. External seed rain may compensate for structural changes, i.e. increased mortality due to edge effects, in fragmented forest stands, particularly focusing on shade tolerant late successional species (Fig. 26.3). Depending on the size of forest fragments, external seed rain is able to offset between 13 and 30% of total observed changes for these species groups. However, in no case external seed rain could fully compensate the adverse effects of fragmentation. Apart from the size of forest remnants, their spatial

Fig. 26.3 Total number of late successional species per hectare (>1 cm dbh) depending on fragment size and percentage of external seed rain in Atlantic rainforests of Brazil; mean ± SD of 10 replicates and 50 averaged time steps



distribution pattern is another important threshold condition having an impact on both species composition and standing bole volume in secondary forest fragments (Köhler et al. 2003). Timber yields are low, i.e. 0.75–1.73 m^3 ha^{-1} $year^{-1}$, even in only lightly fragmented secondary forests. For comparison, Finegan (1992) calculated total timber yields of 4 m^3 ha^{-1} $year^{-1}$ in 25-year cutting cycles in a secondary forest of lowland Costa Rica. The more favourable climate in Costa Rica and the fact that the above figure refers to secondary forests naturally dominated by a few particularly fast-growing species may partly explain the pronounced difference. The high timber yields under short cutting cycles and light fragmentation compared to a monocyclic system point to the beneficial effect of canopy openings. Thinning operations may also stimulate growth and concentrate yield increase on a limited number of commercial species. Under severe fragmentation, enrichment plantings are the only possibility to enhance the commercial value of secondary forests and to keep a certain level of biodiversity. There are promising results with some of the commercial species in the study area in enrichment planting trials in Brazil (Montagnini et al. 1997).

The generally low timber yield in fragmented secondary forests raises the issue whether there are any opportunities to integrate secondary forest management as land use component into farm activities in the study site. Only for large landholdings, labour and capital-extensive land use systems such as secondary forest management may be an attractive option to diversify farm income (cf. Coomes et al. 2000). This particularly holds for Paraguay as a country with an increasing scarcity of timber resources (FAO 2005).

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Part VIII
Planted Forests: Silviculture in Plantations

Chapter 27
Review
Plantation Forestry

Jonathan C. Onyekwelu, Bernd Stimm, and Julian Evans

Abstract Planting trees in tropical countries is becoming an increasingly important forestry activity as many tropical countries that depended on wood supply from natural forests are recognizing the need to establish plantations to augment supplies from dwindling and unsustainable natural forests. The total area of tropical forest plantations increased from about 6.7 million ha in 1965 to 109 million in 2005. Though most species used for tropical plantations are fast growing, their growth rate can be improved substantially through appropriate silviculture such as site-species matching, site nutrient management, use of hybrid species (clonal plantation), etc. This chapter reviews recent advances in tropical forest plantation establishment and management. Subjects that were specifically covered include: extent of tropical forest plantations, principles of productive forest plantation establishment and management, growth and yield of important tree species, silvicultural techniques for improvement of growth, impact of new aspects for silviculture, etc. Two insightful and demonstrative case studies were also presented to illustrate key points.

Keywords Choice of species · Forest plantations · Global extent · Growth · Nutrient management · Productive plantations · Silvicultural techniques

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27.1 Introduction

Planting trees in tropical countries is an increasingly important forestry activity. The changing emphasis from exploitative management of natural forests to managed natural forests and plantation forests, seen in temperate regions over the last 100 years, has been taking place in tropical countries largely over the last 20 years (Evans 1999a, b, c, d).

This chapter draws significantly on Evans and Turnbull's (2004) *Plantation Forestry in the Tropics,* but readers are encouraged to consult the many excellent regional accounts available, for example, Jacovelli et al.'s (2009) *Tree Planting Guidelines for Uganda,* or Kumar's (1995) *Nursery and Plantation Practices in Forestry* which focuses on India, or Wadsworth's (1997) *Forest Production for Tropical America*. In addition there are numerous accounts of the silviculture of individual species or tree genera suitable for the tropics.

27.2 Definition and Meaning of "Forest Plantation"

The difficulty associated with the definition of forest plantation has been pointed out (Kanowski 1997; Jaakko Pöyry 1999; Evans et al. 2009). This is compounded by the difficulty of distinguishing plantations from other forest land uses. It is not always easy to distinguish forest plantations from enrichment planting or from rehabilitation of degraded forest ecosystems and sometimes from natural forests; especially where the natural forests are mostly dominated by single or very few tree species. Most definitions suggested for forest plantations have adopted the degree of management to differentiate plantation forests from other forms of forest land uses (Jaakko Pöyry 1999; Carneiro and Brown 1999). The earliest definition, which was adopted during the World symposium on "man-made forests and their industrial importance" (FAO 1967) states that plantation forests are: *forest stands established artificially by afforestation on lands which did not previously carry forests or on land which carried forest within the previous 50 years or within living memory and involving the replacement of the previous crop by a new and essentially different crop*. This definition has undergone metamorphosis, culminating with the definition of FRA (2000): *forest stands established by planting or/and seeding in the process of afforestation or reforestation. They are either of introduced or indigenous species which meet a minimum area requirement of 0.5 ha; tree crown cover of at least 10% of the land cover; and total height of adult trees above 5 m.*

In 2005, FAO embarked on a Global Thematic Study of Planted Forests (FAO 2006) which led to two significant developments. First, it proved helpful to distinguish two types of planted forests: productive and protective. Secondly to recognize that many forests today, particularly in Europe, while not looking like a conventional "plantation" have a history of regeneration by planting, hence adoption of the preferred term "planted forests." The implications of this are far reaching in our understanding of the contribution planted forests make to global would supply: an accessible account is in Evans (2009a, b).

This chapter deals with plantation forestry mainly in productive plantations, which are defined as "plantations of introduced and in some cases native tree species established through artificial regeneration, i.e., direct seeding and planting, mainly for the production of wood and non-wood goods." Productive plantations can be subdivided into industrial plantations, and plantations for rural development.

Plantations are established for a variety of reasons and vary in composition and structure, as well as in intensity of management. Generally, forest plantations are relatively simple production systems, usually even-aged monocultures, mostly managed to optimize the yield of wood from a site, protect or reclaim an environment and provide benefits and/or amenities that are important to the community (Kanowski 1997; Evans 1998; Carle et al. 2002). Historically, the main aim of forest plantation establishment is to supplement the supply of industrial wood from natural forests (Pandey and Ball 1998; Carle et al. 2002), although there has been a continual diversification of purpose. Industrial plantations are established totally or partly to provide wood for industrial uses (sawn logs, veneer logs, pulpwood) while non-industrial plantations are established mainly for non-industrial uses (provision of fuelwood, or non-wood products, soil protection, environmental protection, etc.). While industrial plantations account for 80% of total plantation area, non-industrial plantations account for only 20%. Productive forest plantations are primarily established for wood and fibre production while protective forest plantations are primarily established for conservation of soil and water. In the last few years, industrial plantations may encompass also afforestation and reforestation activities for managing carbon stock under CDM and other mechanisms.

27.3 Extent of Tropical Forest Plantations

The area of tropical forest plantations has witnessed a phenomenal growth since the middle of the twentieth century, especially within the past three decades. Tropical forest plantations covered an area of 6.7 million ha in 1965 and 21 million ha in 1980. By 1990, the area tripled to about 62 million and almost doubled to about 100.2 million ha by the year 2000 (Table 27.1). The total area of forest plantations in tropical countries is estimated at about 109 million ha by the year 2005. Annual change in plantation area was higher between 1990 and 2000 (about 3.8 million ha) than between 2000 and 2005 (about 890,000 ha). This indicates that more plantations were established in tropical countries in the past than in the present.

Tropical Asia has by far the largest area of plantations and is planting greater area than any other tropical region (STCP 2009). For example, over 85% of new plantings between 1990 and 2005 took place in Asia, with China and India accounting for a much greater part of the plantings (Evans and Turnbull 2004; Carle et al. 2009). In tropical South America, Brazil, Argentina and Chile are the

Table 27.1 Extent and changes in tropical forest plantations (1990–2005)

Region	Area of forest plantations			Annual change rate	
	1990 (000 ha)	2000 (000 ha)	2005 (000 ha)	1990–2000 (ha/y)	2000–2005 (ha/y)
Africa	10,029	10,586	10,864	55,730	27,800
Asia + tropical China and India	42,944	77,263	85,062	3,431,870	779,900
Caribbean	407	426	482	1,930	5,600
Oceania + tropical Australia	312	400	431	8,750	3,100
North and Central America	166	1,489	1,565	132,300	7,600
South America	8,179	10,066	10,722	188,720	65,600
Total tropical world	62,037	100,230	109,126	3,819,300	889,600

Adapted from Evans and Turnbull (2004) and FAO (2005)

leading plantation countries while South Africa, Sudan and Nigeria have the largest area of forest plantations in tropical Africa (see also Chamshama and Nwonwu 2004). In almost all the regions, annual rate of planting was higher during the period 1990–2000 compared to 2000–2005, the only exception being the Caribbean (Table 27.1). The redefinition of tropical forest plantations to include rubber (*Hevea brasiliensis*) is the principal reason for the significantly higher plantation figure in 2000 than 1990.

More than a 100 tree species are used in forest plantation establishments in the tropics and subtropics but only few species dominate. Before 2000, four genera (*Acacia, Eucalyptus, Pinus* and *Tectona*) were recognized as dominant tropical plantation species (Evans and Turnbull 2004) mainly due to the large extent of their plantations. Only three genera (*Acacia, Pinus* and *Eucalyptus*) account for more than 50% of all tropical tree plantations (FAO 2003). The inclusion of *Hevea brasiliensis* as plantation tree species has increased the number of major genera to five because its plantations account for a large proportion (5%) of tropical forest plantations.

In tropical forest plantations for industrial uses, *Eucalyptus* is the most widely planted genus, comprising 24% (8.6 million ha) of the productive forest plantation area. Pine, with 6.4 million ha, is also important, as is rubber (also 6.4 million ha, although some of this may not be available for timber harvesting). Another widely planted tree species is teak (Tomaselli 2007). Other tropical plantation species of increasing importance are *Gmelina arborea, Araucaria* spp., *Leucaena leucocephala*, *Casuarina* species, *Dalbergia sissoo, Terminalia* spp. and *Swietenia macrophylla* (FAO 2001a, b, c, d; Varmola and Carle 2002).

Both indigenous and exotic tree species are used for forest plantation establishment in the tropics and sub-tropics, with exotic species dominating. The domination by exotic species is attributed to their superiority in growth performance over indigenous species, coupled with their ability to suppress noxious weed species. Also, preliminary plantation trials in the tropics involved mostly exotic species while the indigenous species were mostly excluded from the trials for various reasons. In some occasions, when international or bilateral agencies made huge loans available for forest plantation establishment in developing tropical countries, they usually decide that exotic species with fast growth and yield for pulp and paper be planted.

Direct government investment, other policy actions, and private investment, are the main drivers of tropical forest plantation development. Government policy has been a significant factor in plantation development since the advent of large-scale plantation establishment in the 1960s. Most of the plantation estates planted prior to 1995 were established directly by governments or with government funding assistance. However, direct government investment in forest plantation establishment is currently decreasing. Governments now aim to facilitate expansion of state plantations in other ways, such as joint ventures with private landowners, taxation incentives, and loans at subsidised rates or grants. Governments continue to play a significant role in setting an institutional framework that allows for the development of an efficient plantation sector and competitive markets for plantation-based products (Jaakko Pöyry 1999).

FAO was asked to coordinate a process to strengthen country capacity to balance the social, cultural, environmental and economic dimensions of planted forest management and to increase their contributions towards sustainable livelihoods and land use. The process involved experts in planted forests from governments, the private sector (corporate and smallholder), NGOs and academics and identified critical niches for a set of voluntary guidelines (FAO 2006).

27.4 Purpose of Plantations and Species Selection

The purpose of plantation has usually been one of the following four categories: (a) industrial wood production, (b) domestic wood production, (c) environmental protection (see Weber et al. this volume), and (d) rural development (Franzel et al. 1996). In recent years, an additional purpose of carbon sequestration via plantations is emerging.

As already mentioned, we like to put emphasis on productive forest plantations in this chapter, so we focus on plantations for industrial and domestic wood production, whether on large or small scale. Of course, most of the economic, environmental, societal impact comes from large-scale forest plantations, which sometimes lead to critique, that must be taken seriously. On the other hand, especially the socio-economic impact of small-scale plantation projects can no longer be neglected. If this type of afforestation is both initiated and carried out by farmers, it is argued to have more positive and fewer negative impacts, e.g., less social conflict, than large-scale afforestation by non-farmers (Schirmer 2007).

Species choice is the most important decision, once a plantation project is initiated. The choice of species depends mainly on three questions: (1) What is the purpose of the plantation? (2) Which species are potentially available for planting? (3) What will grow on the sites available and how well will they grow? (see Evans and Turnbull (2004) for an overview on factors influencing the choice of species).

From the experiences with large-scale plantation projects in recent decades, we have some sound information of suitable species and related end-uses (fuelwood, pulpwood, sawn timber, etc.). The knowledge base on plantation species, including exotics, is already substantial. This is true for plantations with species of the genera *Eucalyptus, Acacia, Pinus* and *Tectona*, which cover a large area of plantations outside their native range, e.g., in Africa, Asia, and Latin America. In addition to the choice of species, the selection of the most productive provenance within a species is essential for a successful plantation project (Mead 2005). Results from the international seedlot trial for *Eucalyptus camaldulensis* on 32 sites in 18 countries showed that growth gains of several 100% can be achieved by selecting the best provenance for the prevailing condition (FAO 2002b, cited in Mead 2005).

On the other hand, large-scale plantations established with native species are still limited. There are exceptions to this, notably, *Araucaria cunninghamii* in Queensland, Australia, *Cunninghamia lanceolata* in sub-tropical China and *Tectona grandis* in India. New aspects, like the stipulation on the conservation of biodiversity or the restoration of degraded forest ecosystems, have contributed at least regionally to an increased proportion of native species in new plantation projects. Redondo-Brenes (2007) provided results from a study in Costa Rica, where the government has provided incentives for reforestation programs since 1986 and initiated a *Payment for Environmental Services* program in 1996, which yielded reforestation programs with native species throughout the country. His study aimed to provide information about growth, carbon sequestration, and management of seven native tree species (*Vochysia guatemalensis, Vochysia ferruginea, Hyeronima alchorneoides, Calophyllum brasiliense, Terminalia amazonia, Virola koschnyi*, and *Dipteryx panamensis*) growing in small and medium-sized plantations in the Caribbean and Northern lowlands of Costa Rica. The results of the research enhanced the criteria elaborated in previous research findings to improve species choices for reforestation and silvicultural management in Costa Rica and in other regions with similar ecological features. Furthermore, they support the concept that tropical plantations can serve diverse economic, social, and ecological functions that may ultimately help reduce atmospheric CO_2 accumulation.

One – and this may be the most important – reason that plantations established with native species are still at the initial stage is that availability of reproductive material, whether seed or seedlings, of native species cannot be guaranteed.

For many parts of the tropics, especially if large-scale plantation projects are envisaged, it becomes clear that native species are in fact no real alternative – apart from the important exceptions noted above – because in a limited time horizon the critical seed supply with native species is not existing or cannot be realized. This is very often the case in countries, where an organized national forestry programme and legal framework of forest regulations is still missing or at the beginning of development. In such cases, provided that the financial basis to establish a new plantation project is already there, the choice of species is solely dominated by the availability of high numbers of seeds and seedlings on the market. The consequence is that native species will still be neglected, because it is much simpler to use introduced ones which are readily available in the market in a high number with a reliable quality. If the discussions on the use of native species in large-scale plantation projects shall not remain on the academic level only, but is intended to be put seriously into practice, high emphasis must be put in developing strategic concepts on seed management and their practical implementation.

Reforestation with native species is considered a preferable option for sustainable development, overcoming some of the ecological drawbacks of the foregone deforestation and concurrently contributing to the conservation of the region's biodiversity. However, lack of knowledge of the biology of the trees providing seed resources, e.g., about population densities, mating systems and reproductive phenology, as well as of their seed germination eco-physiology and the establishment of saplings, poses a severe challenge for any reforestation project (Stimm et al. 2008).

Planning aspects for plantation programmes must focus on the conservation and sustainable use of forest genetic resources. Forest plantations with native species are not only an option to provide sustainable supply of timber and NWFPs and to minimize the pressure on natural forests but can also be an important complementary contribution to a future-oriented *dynamic conservation*.

For instance in Ecuador, only 167,000 ha of plantations were successfully established by the year 2000 (FAO 2003). Most of the plantations, however, consist of introduced species, mainly *Eucalyptus* spp. and *Pinus* spp. Nowadays more emphasis is put on plantations with native species, e.g., *Alnus acuminata*, *Cordia alliodora* or *Ochroma pyramidale* (see also Brandbyge and Holm-Nielsen 1986; Borja and Lasso 1990; Aguirre et al. 2002a, b; Predesur 2004). But very little attention is still attributed to the provenance of the material from gen-ecological zones and the importance of using autochthonous planting material. Hansen and Kjaer (1999) stressed that appropriate genetic material may not only enhance production and quality but also the health and stability of plantations.

For the South of Ecuador, Stimm et al. (2008) performed a first selection of priority species, where potentially suitable native species were selected from the more than 200-tree species using the following criteria: high local acceptance, economical value (timber and non-timber products), endangered species or species with a high significance for the ecosystem "tropical mountain rain forest" and species typical of certain successional stages such as pioneers or representatives of the climax vegetation. Finally, some 15 native species with a promising potential for reforestation were selected. Consequently, data are needed to delineate "seed transfer zones," or regions within which reproductive material, i.e., seeds or seedlings, can be moved with little or no impact on its population fitness. This problem was already approached for the Province of Loja, South Ecuador (Günter et al. 2004).

27.5 Species: Site Matching

Compared to temperate region, forest plantation silviculture is still a recent phenomenon in the tropics. Many developing countries that previously depended on wood extraction from natural forests are now recognizing the need to establish commercial wood sources, mainly due to high annual deforestation rate, coupled with the dwindling and unsustainable nature of wood supply from natural forest. Foresters agree that large-scale plantation development, especially on marginal lands, is essential in many tropical and subtropical countries (Brown et al. 1997), particularly those with high population densities, and with high forest product needs. The greatest immediate gains in yields from forest plantations will depend highly on appropriate matching of species with available site. Where there is appropriate species-site matching and when management prescriptions are effective, plantations will usually remain healthy and productive (Brown et al. 1997). Much of plantation silviculture is concerned with achieving the best match between species and planting site, a task that is not always easy given the highly variable sites and numerous species in the tropics. Individual trees and provenances may respond differently relative to each other in different environments, a situation known as genotype × environment interaction. Genotype × environment interaction analyses have been used extensively in crop science in developing high yielding cultivars. They are also used in forestry to quantify the performance of provenances or clones across a wide array of sites (Butterfield 1996). Interactions are principally with climatic parameters at variety and provenance level, which underline the need to accurately define site and the exact genetic origin of planting materials. The first step in matching species with site is to determine the limitation imposed by the environmental factors that collectively constitute the site. The second step is the screening of groups of species that will grow well in the site and that are most suitable in meeting the objective of plantation management.

In many locations where plantations have only recently been established, little is known about the potential capabilities for increasing productivity as well as potential problems that may limit yields (Brown et al. 1997). The availability as well as the quality of sites for forest plantation establishment must be known to ascertain their appropriateness to meet the objectives of the planned afforestation programme. The key is to select the site that, when planted, will lead to the establishment of successful forest plantation, which demands the description of the tree's environmental requirements and the characteristics of the site.

The environmental potentials of a tree species' natural habitat can initially provide the best guide to the sort of conditions the planting site should have (Evans and Turnbull 2004). In most cases, the results from a site where a tree species is growing (either natural or exotic) strictly apply only to that site; their application to another site usually involves the assumption of site comparability, an assumption that may or may not be justified. Thus, there will often be the necessity for field trials for precise matching of species and provenances to particular sites (Eldridge et al. 1993). Seven factors have been used to characterize the climate of a site (Webb et al. 1984; Booth 1996):

- Mean annual rainfall (mm);
- Rainfall regime (uniform/bimodal, winter, summer);
- Dry season length (consecutive months less than 40 mm rainfall);
- Maximum temperature of the hottest month (°C);

- Mean minimum temperature of the coldest month (°C);
- Mean annual temperature (°C);
- Absolute minimum temperature (°C).

These factors, which have been discussed in detail by Evans and Turnbull (2004), have generally proved to be useful discriminators of regions that are climatically suitable for growing particular trees. The selection of tree species through the use of analogous climates is important only as a first step; it must be amplified by an evaluation of more important localized factors such as soil, slope and biotic factors. Some tools to assess the growth capability of a tree species in a given site have been developed using climatic and soil variables, e.g., soil aeration, pH, salinity, slope and texture (Booth 1998; Hachett and Vancley 1998). The application of these tools requires some knowledge of how well the species grows in relation to these characteristics. The tools have been used in mapping locations suitable for the establishment of some plantations (Booth et al. 1989; Thwaites 2002). However, such tools can only give guidelines as to which species might be appropriate for the site. Thus, final match of species to the available planting site may not preclude the need for species trials, since climatological or ecological matching may not reveal the adaptability of a species. Without such trials, the choice of tree species may, in most cases, be a risky business, it may result to large-scale failures.

Therefore, when the best possible information has been collected on the characteristics of the potential site, the next step is the selection of the suitable tree species to be planted. The aim is to choose species that are suited to the site, which will remain healthy throughout rotation, which will produce acceptable growth and yield, and which will meet management objectives. Many tropical foresters are still in the initial stages of determining which species are best adapted to plantation management (Butterfield 1996). Choosing the appropriate tree species is the most important decision once a forest plantation project is initiated. This is because species choice to a large extent determines the success or failure of the planting programme. The species selected influences the silvicultural and management practices, as well as the utilization of the final crop. Consequently, the selection process must be carefully done. In general, tree species selected for plantation establishment on a specific site should be able to exhibit their maximum growth potential, with high productivity per unit area, and should not have negative impacts on the soils. In addition, the selected species should preferably be able to grow in monoculture and, to some extent, have the ability to grow in mixture with other species. Generally, suitable tree species are gap invaders, pioneers or early secondary species. Late secondary and climax species are often not so suitable for plantations.

The most logical action is to choose from indigenous species whose natural ranges include the target site. If the choice species are indigenous, the problem of matching them with site is of little importance. This is because indigenous trees have the virtue of a long history of inherited adaptation to environmental and site conditions of their area. However, this kind of adaptation favors the perpetuation of their own kind, which does not necessarily include high production rates, stem straightness or other desirable attributes. Where exotic species are to be used, it becomes inevitable to match species with site because not all species can grow well on sites outside their natural habitat. Before using exotic species for large-scale planting, it is necessary to have some assurance that when planted, the selected species will thrive and produce the desire products at the end of rotation. This can be achieved through detailed species field trial.

The dominance of exotic species in forest plantations in the tropics has led to enormous number of field trials in the tropics over the years. Conventional field trials, which will always be a critical part of species selection programme, are relatively straightforward and can be carried out without specialized equipments, though they are usually long-term experiments that may be subject to numerous hazards (Evans and Turnbull 2004). The trial plots can vary in size depending on the scale of trial and number of replications and should cover the whole range of ecological zones within the intended planting site. For such trials, detailed growth performance records should be maintained throughout the experimental period. They should produce results that will enable different species to be compared scientifically and the site variations to be estimated with some precision. Guidelines for field species trials in the tropics have been given by Burley and Wood (1976) and Briscoe (1990). Evans and Turnbull (2004) identified four stages for selecting species for industrial plantations:

- Species elimination: evaluate many species, eliminate failures and identify promising ones
- Species refinement: examine genetic variation within the promising species, in particular compare provenances
- Industrial scale trials: large-scale trials to provide stand growth data, to test methods of cultivation, and to evaluate the likely species on the range of sites encountered in the project
- Tree improvement: identification of land races, breeding, clonal propagation, etc. to create better forest stands for later planting and subsequent rotations

In designing species trials, certain general points have to be borne in mind. These include:

1. Carefully examining all available knowledge about the performance and requirements of species already tested and failed. The objective of the new species trials should also be defined as precisely as possible.
2. The trials must cover the whole range of sites to be planted.
3. Though the ultimate aim may be to select only one or two species for planting, a fairly large number of species may be tried before the most suitable ones can be selected.
4. Since nursery practices may influence the success or failure in plantations, the role of nursery in species trial is very important. Thus, a high degree of control over nursery conditions to ensure that they are reasonably constant and uniform is essential.

27.6 Seed Collection, Supply, and Storage

Continuous and long-term availability and supply of high quality seed and plant material for any kind of planting activity is one of the fundamental challenges for sustainable plantation management and requires the establishment of production standards. The installation of a reliable programme for managing tree seed resources on a national or regional level is a very first but nevertheless important step for the realization of successful planting activities. To conceptualize and achieve standards, the approval and monitoring of seed sources of priority species is one of the basic steps, which will be accomplished by seed certification and control of seed procurement.

In planning plantation programmes with indigenous species, the major seed sources will be a network of natural stands, where vigorous trees with desired characteristics have been observed phenologically and selected for seed collection. In areas where sustainable forest management has already been established, "seed production stands" can be assigned for seed harvesting. The establishment of such stands is a precondition for a professional management of seed production, and may include the establishment and management of conservation stands.

The provision of genetically suitable seed and other reproductive plant material of good physiological quality from selected indigenous seed sources is the main goal, which must be achieved. "Suitable" includes the location, use and maintenance of clearly defined and well-documented seed sources.

In many parts of the tropics, most of the reforestation projects still deal with exotic species of the genera *Eucalyptus* or *Pinus*, while native species have not been used in the region so far due to the reasons already mentioned. Establishing plantations with native species is an option to contribute to the conservation of the regional biodiversity, but knowledge about the reproductive biology of such species is very limited. Nevertheless, it is indispensable for the production of adequate numbers of high quality seedlings in tree nurseries.

To receive sufficient genetic amplitude for large-scale plantations, Graudal et al. (1997) recommended harvesting at least 50 individuals of one provenance or species. Unfortunately, reproductive biology and mating systems of many native species are still unknown.

For practical purposes therefore, it is necessary to monitor a higher number of individuals of a certain seed zone for seed harvesting. This, however, is very difficult in the tropics where many species appear with very low abundances. The resulting problems can be demonstrated with an example from the Rio San Francisco valley (Zamora Chinchipe Province, Ecuador): *Cedrela sp.* with DBH > 30 cm has an abundance of about 0.8 ha^{-1} and *Prumnopitys montana* approx. 0.6 ha^{-1} in the lower parts of the RBSF (Günter and Mosandl 2003). Every year about 40 and 30% of the individuals in each population of these two species showed fructification. We assume that the reproductive phase of these two species starts only with a DBH of 30 cm. In this case, the establishment of a seed bank with a sufficiently broad genetic basis would require collecting seeds from an area of at least 160 ha for *Cedrela sp*. and 270 ha for *Prumnopitys montana* with respect to the Rio San Francisco provenance. Monitoring of those areas necessitates an enormous input of time and manpower. For comparison, forest inventories for African mahoganies in unlogged forest revealed a relative abundance of large individuals (e.g., ≥80 cm DBH, which is thought to be a minimum diameter for seed production of *Entandrophragma* spp.) of 0.5–2 individuals per ha. Forests containing *E. cylindricum* have been shown to fruit anywhere from zero to two times during a given year (see Hall 2010, this volume). In that case, the establishment of an *Entandrophragma* spp. seed bank with a sufficiently broad genetic basis would require collecting seeds from an area between 25 and 100 ha. The density (N/ha) of mature big-leaf mahogany *(Swietenia macrophylla)*, i.e., trees exceeding 30–40 cm diameter, in Bolivian natural forests along a gradient of increasing dry season length has been found to be between 0.2 and 0.5 individuals per ha (see Grogan et al., this volume). Applying the above-mentioned rule of thumb, this would mean that the establishment of a big-leaf mahogany seed bank for a specified provenance with a sufficiently broad genetic basis (min. 50 individuals) would require collecting seeds from an area between 100 and 500 ha.

For mitigating the scarcity in native forest reproductive material, the establishment of competent tree seed centres and tree seed programmes is of utmost importance.

27.7 Domestication and Tree Improvement

Unlike in agriculture and animal husbandry, domestication of tropical forest tree species has only been practiced since the dawn of the twentieth century. Tropical trees being domesticated are found in primary forests, secondary forests, communal fallow lands, plantations and farms. The use of provenance selection, an early stage in the domestication process, gained international importance in the 1960s, although techniques such as clonal selection have been used for centuries in a limited number of species – *Salix, Populus, Cryptomeria japonica*, etc. (Leakey and Newton 1993). In forestry, the term "domestication" has mostly been applied to genetic improvement of trees for industrial plantations but recently, it has been taken to encompass the identification and characterisation of germplasm resources; the capture, selection and management of genetic resources; and the regeneration and sustainable cultivation of the species in managed ecosystems (Fig. 27.1). The Working Group on "Product Domestication and Adoption by Farmers" during the conference on 'Domestication and Commercialization of Non-timber Forest Products in Agroforestry' (Leakey et al. 1996) defined domestication as 'a progression from collection and utilization of products, through protection, management and cultivation, which culminates with genetic manipulation'.

Tree domestication is a dynamic process, which develops from deciding the tree species to domesticate and proceeds through background socioeconomic studies, the collection of germplasm, genetic selection and improvement to the integration

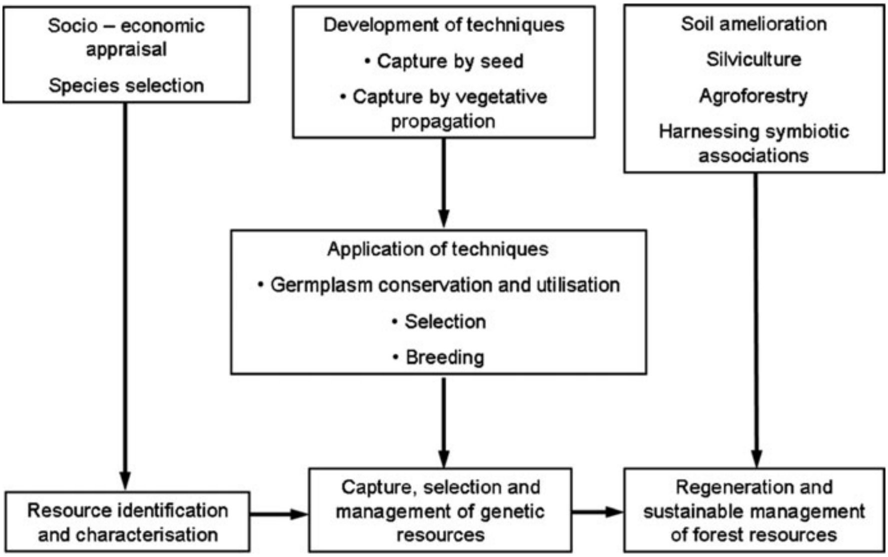

Fig. 27.1 Aspects/stages involved in the domestication of tropical trees
Source: Leakey and Newton 1993

of domesticated species in land use. It should be an ongoing process in which genetic and cultivation improvements are continuously refined. However, domestication is not only about selection. It integrates the four key processes of the identification, production, management and adoption of tree resources. It is highly species-specific and thus, an extremely variable process.

Techniques for selection and breeding can be very simple or quite complex. Finkeldey and Hattemer (2007) gave a comprehensive account on these techniques and on breeding strategies as well. For majority of species, it may consist simply of identifying and collecting seeds from individuals with desirable traits and developing appropriate propagation and cultural practices. For widely planted and important timber species, the full domestication process may involve the systematic sampling and characterisation of genetic variation, development of optimal propagation and silvicultural techniques, and intensive breeding, including the use of molecular genetics technologies and sometimes hybridization (Finkeldey and Hattemer 2007). Domestication seeks to bring out the maximum human benefit within a species as it is genetically refined from a wild tree to a cultivated plant.

Breeding research is focusing on heterosis breeding, where heterosis is defined as superior trait expression of heterozygous genotypes in comparison with corresponding homozygote (Finkeldey and Hattemer 2007). One method of heterosis breeding is hybridization between forest species, which can result in an improvement of desired trait expressions, such as better resistance against pests and pathogens or better growth performance. Finkeldey and Hattemer (2007) summarized some illustrious examples in their textbook. The establishment of seed orchards, either clonal or seedling seed orchards, is one of the cornerstones in breeding programmes, e.g., for teak in Thailand (Finkeldey and Hattemer 2007).

In this context, the development and application of vegetative bulk propagation techniques has contributed essentially to the success of breeding programmes, e.g., on eucalypt hybrids in Brazil (Finkeldey and Hattemer 2007). In Aracruz, Brazil, within the last 30 years, Eucalypt plantations established from seedlings have successively been replaced by clonal plantations, which grow enormously fast with a MAI of 60 m^3/ha/year and more. In the case of development of clonal plantations, it is reasonable to use a large clone assortment. As for seed collections, where seed should be collected from at least 20–50 trees of a population (see Dawson and Were 1997), care needs to be taken to maintain a broad genetic diversity (Finkeldey and Hattemer 2007). Hence clonal mixtures are recommended (where in some countries the law regulates the number of clones that must be contained in any one mixture, e.g., in Canada >50) (Zsuffa et al. 1993).

The tropical forest ecosystem is rich in natural resources, particularly trees that provide food, fuel, fibre, medicines and various other products, including construction and building materials that have provided indigenous people with many of their daily needs for millennia. These resources are particularly important for rural economies, enhancing economic empowerment, rural employment, etc. It has been predicted that food tree species will become of greater importance within the next few years (Ayuk et al. 1999), especially for sustainable development of rural livelihoods that depend on them. This is based on the increasing demand and the emerging domestic and international markets for their products. However, despite their importance, these tropical forest resources have been greatly neglected. Most of them have continued to grow in the wild, with continual decreasing yield due to old age and the fact that they have been harvested for decades. Consequently, a lot of them have died or are in the process of doing so, while many others are currently endangered, with a high possibility of going into extinction in the near future. In Nigeria, quite a number of forest food tree species (e.g., *Vitellaria paradoxa, Chrysophyllum albidum, Irvingia gabonensis, Treculia africana,* etc.) have been classified as endangered (FORMECU 1999). Allowing them to go into extinction will endanger the livelihood of millions of rural dwellers in the tropics and reduce the rich biological diversity of the tropical ecosystem. Artificial regeneration and subsequent improvement of the species (domestication) appears to be a very viable option of saving these species and ensuring that their products are supplied on sustained basis. The objective of domestication is to enhance the performance of trees in terms of improved products (e.g., timber, fruits, and medicines) and/or improved environmental services (e.g., amelioration of soil fertility) (Simons and Leakey 2004). The genetic benefits obtained from the domestication of mango (*Mangifera indica* L.), *Citrus* spp., breadfruit (*Artocarpus altilis* (Z) Fosb.) and avocado (*Persea americana* Miller) in the tropics has shown that large genetic gains are possible with the domestication of wild tropical forest food tree species.

There is great urgency to achieve these benefits if the severity of the current man-made episode of species extinction (the so-called "Sixth Extinction" (Leakey and Lewin 1996)), is to be defused. In this respect, Sanchez and Leakey (1997, cited in Leakey and Simons 1998) sees domestication of forest food trees as one of the three determinants for balancing food security with natural resource utilization.

Strategies for individual species vary according to their functional use, biology, management alternatives and target environments (Simons and Leakey 2004; Finkeldey and Hattemer 2007). Domestication can occur at any point along the continuum from the wild to genetically transformed state. The intensity of domestication activities for a single species will be dictated by a combination of biological, scientific, policy, economic and social factors. The need to rapidly domesticate tropical forest food tree species has been stressed and is now one of the three pillars of World Agroforestry Centre (formerly known as International Centre for Agroforestry Research, ICRAF) programme (Leakey and Simons 1998).

The selection of an elite tree for domestication could vary depending on the desired product(s). For indigenous forest food tree species, rapid progress will be made if indigenous knowledge is used. Usually, indigenous people know the best individual trees in terms of yield, fruit size, taste, flavour, etc. The people can be asked to report the existence of superior trees, thus reducing the task of screening large numbers of trees. Unlike fruit trees, chemical screening process may be required for medicinal trees. However, the magnitude of this task can probably be reduced by starting on a population basis, since it is likely that trees from certain environments will be richer in the required components. For timber trees, stem form, bole straightness, log size are the first selection criteria. Various forms of "plus-tree" (i.e., elite selection) provenance and progeny selection are well known. For fruit trees, crown size, fruit yield, fruit size, fruit taste, etc. are the main selection criteria.

Research work on the domestication of forest food tree species in the tropics is still at preliminary stage and covers seed germination and methods of seed pretreatments (Onyekwelu 2004), prospects of vegetative propagation (Shiembo et al. 1996), phenotypic variation of fruits and kernels (Anegbeh et al. 2003), selection of multiple traits for potential cultivars (Atangana et al. 2002; Ngo Mpeck et al. 2003), germplasm collection focused strongly on the species identified through farmers' input, priority setting exercise (Leakey and Simons 1998); integration into agroforestry (Leakey et al. 2003; Kumar and Nair 2004; Simons and Leakey 2004); uses, management, economic and nutritional importance (Okigbo 1978; Ayuk et al. 1999; Leakey 1999; Okafor 1991). Atangana (2000) quantified the morphological variability of fruit and kernel traits of *I. gabonensis* in Cameroon and Nigeria respectively. However, phenotypic variation in the fruit has only been descriptive and limited to only *I. gabonensis* (Ladipo et al. 1996; Anegbeh et al. 2003) while no known documented study on the silviculture of the tree species in the nursery and plantation exists, which should be the next step in the effort to domesticate these species.

Since 1998, the World Agroforestry Centre, in collaboration with a range of partners, has been developing a participatory approach to domestication of indigenous trees. In Solomon Islands, several varieties of nut have been selected and domesticated (Evans 1999b), including *Canarium* (ngali nut, three species), *Barringtonia* (cut nut, three species), *Terminalia catappa* (beach almond), *Gnetum gnemon* and *Pandanus* (screw pine, several species and numerous varieties). Much of the current work of the Tree Improvement and Genetic Resources Programme at CSIRO Forestry and Forest Products (CSIRO FFP) is conducted within the context of species domestication. Presently, the Australian Tree Seed Centre (ATSC), in collaboration with its many research and development partners, has started to domesticate 70 species in 22 genera. An essential precursor to this work has been the assembly of biogeographic information on particular species and genera, which are frequently published in monographs and annotated bibliographies and as electronic Forestry Compendium (Doran and Turnbull 1997; CABI 2000; Kalinganire and Pinyopusarerk 2000).

New aspects have evolved quite recently: The introduction of biotechnology, including genetic modification (GM) of trees, to plantation forestry has the potential to increase the productivity of planted forests and create novel products and desired qualities (Sedjo 1999, 2004). Genetic transformation has been reported for poplars and some other, mainly temperate, tree species (Owusu 1999; Fladung and Ewald 2006; Ishii 2006). Prospective benefits, like a reduction in lignin content, ability of phytoremediation or resistance against pests and diseases, need to be balanced against the risks associated with genetic transformation, e.g., horizontal gene transfer from GM trees to non-target organisms. Burdon and Walter (2004) reviewed risks of transgenic exotic pine and eucalypt plantations and strategies of risk management.

27.8 Plant Propagation

Another problem is the propagation of plants, especially from seed of lesser known species. As with many tropical tree species, the knowledge of optimum propagation is scarce (but see Kumar 1995; Vozzo 2002; Schmidt 2007). Our experiments with native Ecuadorian species (Stimm et al. 2008) corroborate the need to develop species-specific appropriate propagation techniques and protocols, otherwise planting material for reforestation purposes might not meet the high qualitative standards and required numbers.

The development of vegetative propagation techniques is often complementary and aims at the identical reproduction of plants with desirable features such as high productivity, superior quality, or high tolerance to biotic and/or abiotic stresses (Jaenicke and Beniest 2002; Ishii 2006). The reasons for vegetative propagation very often include an irregular seed production in nature, low survival of seeds in storage and seed quality, realisation of gain in domestication and tree improvement programmes, production of uniform material and genetic modification. Cutting propagation has been reported for a number of tropical and subtropical species (Rimbawanto 2002; Ahmad 2006; Baker and Walker 2006). Rooting success is species-specific as well as clone-specific. Most important factors for the rooting of cuttings, their survival and subsequent growth is the age of the mother tree (e.g., the younger the tree the better the rooting), time of harvesting the cuttings, substrate, humidity, hormones and sugars. Rejuvenation of physiologically old plant material can be initiated through (repeated) hedging of mother stock from clonal gardens.

Especially, tissue culture techniques give the possibility for bulk propagation on a small area, independence of season, and the possibility to produce virus-free plant material (Ishii 2006; Jain and Ishii 2003; Jain and Häggman 2007). Nevertheless, these techniques can have some shortcomings, e.g., a relatively high financial investment for the basic facilities and equipments, high running costs, and a higher incidence of mutations through somaclonal variation. The most common techniques in use are shoot tip micropropagation, multiplication via axillary shoot formation, nodal segments or adventitious shoots, embryo culture from seeds and somatic embryogenesis.

The overall objective in nursery production is high quality planting stock, which is a prerequisite for high survival and good early growth in the field (Mead 2005). Fertilization in the nursery aims at the supply of essential mineral nutrients for accelerated seedling growth and is therefore one of the important cultural activities in forest nurseries. Nursery stock make a considerable demand on soil nutrients and there is little or no nutrient cycling in the nursery because their leaves are hardly shed off, coupled with the short life span of seedlings in the nursery. The loss of organic matter during cultivation, leaching and activities of microorganisms, which results to degradation of nursery soils regardless of the original fertility of the site (Nwoboshi 2000), makes the use of fertilizers in forest nursery almost inevitable. There is ample evidence that the application of fertilizers in the nursery improves seedling growth and maintains or improves nursery soil fertility. Fertilizer application has been shown to markedly improve the physiological quality of planting stock. For some tropical plantation tree species (e.g., *Tectona grandis*, *Gmelina arborea, Pinus* spp., etc.), fertilizers are applied at the nursery stage to enhance their survival and initial growth after transplanting to the field.

Suspected nutrient deficiencies, indicated by symptoms such as chlorosis, should always be confirmed by soil or foliage tests because symptoms can be caused by many factors. Unless those tests show other nutrient deficiencies, nitrogen and potassium are the only fertilizers that are typically applied to nursery stocks during the growing season. Application rates are determined by experience or from soil or foliage tests, and the fertilizers applied by drop or rotary spreaders. Some nurseries inject soluble fertilizer solutions into the irrigation system or apply them through a spray boom behind a tractor. Liquid fertilizer solutions are injected into the irrigation lines in the headhouse and applied to the crop through nozzles.

A promising approach to the improvement of seedling quality in nurseries for reforestation purposes could be the inoculation of seedlings with mycorrhiza (see also Suzuki et al. 2006). Urgiles et al. (2009) successfully applied mycorrhizal roots which enhanced the growth of tropical tree seedlings in the nursery. Most tree species in tropical mountain rain forests are naturally associated with arbuscular mycorrhizal fungi. Previous studies in southern Ecuador of 115 tree species revealed that only three species were not associated with arbuscular mycorrhizal fungi. Urgiles et al. (2009) suggested that seedlings of tropical tree species raised in the nursery may need to be associated with arbuscular mycorrhizal fungi to survive transplantation shock in higher numbers. Methods for establishing plantations with native tree species are not yet established for Ecuador. Assessment of plant growth and mycorrhizal status of 6-months-old *Cedrela montana* and *Heliocarpus americanus* revealed an improvement in growth and diverse associated fungi through mycorrhizal root inoculation in comparison with moderate fertilization. Moderate fertilization did not suppress mycorrhization.

27.9 Plantation Establishment

Satisfactory establishment of tropical plantations depends on adequate site preparation. This is important for all species but absolutely critical for Eucalypts. The latter require weed-free conditions and well-cultivated soil to achieve the remarkable growth rates of which they are capable.

Physical properties of soil, e.g., soil compaction, are often responsible for the poor establishment of trees. Soil cultivation of various types and intensities, ranging from simple pit planting to mechanical ripping, may help to overcome this problem. Physical treatments – cultivation – aim to significantly improve root development and rooting depth by reducing soil bulk density, improving internal drainage and temporarily effecting weed control. Site preparation for the next rotation has to be done sensitively and should focus especially on conservation of organic matter (Evans 1999a, b, c, d).

27.9.1 Spacing

Usually two patterns of planting are used in tropical plantation establishment, square (the most common) and rectangular planting patterns. Triangular or other alternative patterns are seldom practiced (Evans and Turnbull 2004). The stocking (planting density; number of trees planted per hectare) is one of the main silvicultural decisions in the establishment of plantations and is realised in the distance between trees (spacing). On the one hand, it is a factor affecting cost, because close spacing requires a higher number of seedlings, but on the other hand, close spacing can induce the self-pruning of trees from branches, especially in hardwood species, and hence improve the quality of timber. In close spacing, a higher number of stems also improve the opportunity for a selection for desired growth form and other important silvicultural characters. Spacing in combination with thinning is used to manipulate not only the quality but also the size of the crop trees, e.g., at close spacing (high densities) the mean size of the average tree will be low, whereas at wide spacing the mean size will be high. In forest plantations in the tropics, initial spacing smaller than 5×5 m (growing space per tree is < 25 m^2, stocking is > 400 trees/ha) are generally implemented, and spacing of 3×2 m (rectangular pattern; growing space per tree is 6 m^2, stocking is 1,667 trees/ha) or 3×3 m (square pattern, growing space per tree is 9 m^2, stocking is 1,111 trees/ha) are common in plantations established for timber production. For biomass and fuelwood plantations much smaller spacing is used, e.g., 1×1 m (growing space per tree is 1 m^2, stocking is 10,000 trees/ha). The description of impacts of various spacings on environmental features, on the management of stands and on costs and revenues go beyond the scope of our current review, therefore we recommend the study of this topic elsewhere (e.g., Evans and Turnbull 2004).

27.9.2 Weed Control

Weed control until canopy closure greatly aids the establishment of trees. A good compromise is to maintain a 1-m diameter weed-free zone around every tree from the time of planting until canopy closure. This can be made manually, mechanically or via the application of specific chemicals (herbicides). The intensity of weed control varies according to species, site and climate, i.e., in areas with year-round rainfall, weed control may be needed six or more times in the first year. For details on weed control, weeding practices and methods we refer to the comprehensive review of Evans and Turnbull (2004).

27.9.3 Protection

Fire poses a major threat to tropical plantations, especially in the drier tropics and sub-tropics, and savanna habitats. Fire damage is often heavy, in particular when plantations are poorly maintained and a large amount of fuel from woody debris and litter is available (Saharo 1999). All tropical plantations will require protection from browsing animals, domestic livestock or wild, until trees are at least 4 m tall and sturdy enough to resist damage. Both fencing and shepherding are commonly used to prevent such damage. Many tropical plantations suffer from termite damage, and local control, e.g., by termiticides, will be necessary.

27.9.4 Mixtures

Critical views, especially on large-scale plantations are coming up regarding the goods and services provided to regions and rural communities (FSC certification etc.). Although biodiversity levels in plantations are often assumed to be lower than those in natural forests, plantations can host a high number of native animal and plant species endemic to primary and secondary forest ecosystems. Currently biologists and foresters are seeking a better understanding of the value of planted forests for biodiversity conservation (see Carnus et al. 2006). Barlow et al. (2007) did a comparison of bird communities of these habitats and neighbouring primary forest in north-east Brazilian Amazonia. They demonstrated that species richness was highest in primary forest and lowest in *Eucalyptus* plantations, and community turnover between habitats was very high. Monthly line-transect censuses conducted over an annual cycle showed an increase in the detection of canopy frugivores and seed predators during the peak of flower and fruit availability in primary forest, but failed to suggest that second growth or *Eucalyptus* stands provide suitable foraging habitat at any time of the year. The conservation value of both secondary forest and plantations was low compared to conclusions from previous studies. Their results indicate that while large-scale reforestation of degraded land can increase regional levels of diversity, it is unlikely to conserve most primary forest species, such as understory insectivores and canopy frugivores.

Brockerhoff et al. (2008) provided a comprehensive review of the function of plantation forests as habitat compared with other land cover, examined the effects on biodiversity at the landscape scale and synthesised context-specific effects of plantation forestry on biodiversity. Natural forests are usually more suitable as habitat for a wider range of native forest species than plantation forests but there is abundant evidence that plantation forests can provide valuable habitat, even for some threatened and endangered species, and may contribute to the conservation of biodiversity by various mechanisms. Afforestation of degraded or abandoned agricultural land can provide complementary forest habitat, buffer edge effects, and increase connectivity. The authors provided context-specific examples and case studies to assist impact assessments of plantation forestry, and offered a range of management recommendations.

Cummings and Reid (2008) presented an overview of stand-level approach practices they have adopted in managing flooded gum (*Eucalyptus grandis*) plantations infested with lantana (*Lantana camara*) to enhance their biodiversity value. Experiments designed to overcome barriers limiting regeneration of native forest trees yielded insights into the management of former timber plantations for biodiversity. Thinning and burning stimulated regeneration of native species. Retained canopy cover was proportional to the richness or abundance of native woody shrubs, understory trees and native perennial herbs, indicating that management intensity can be varied to promote a range of conservation values.

To compensate for the large-scale reduction in biodiversity, a landscape approach including a mix of production areas with habitat components and corridors for biodiversity seems to be a powerful instrument (van Bodegom et al. 2008). Other measures, e.g., increasing the level of genetic diversity, are a complementary and hence useful instrument. In this sense, one measure to facilitate biodiversity rehabilitation in plantations is to establish mixtures of tree species, preferably mixtures of native species, and/or with exotics. Marjokorpi and Salo (2007) analyzed operational standards and guidelines for biodiversity management in tropical and subtropical forest plantations.

Additional reasons for an establishment of mixed species plantations may include (1) providing a diverse range of products, including NWFPs, (2) diversification of risk of production, (3) obtaining a greater yield of products, (4) providing a nurse crop (see e.g., Kelty 2006; West 2006).

From a silvicultural point of view, it is necessary to reflect on the patterns of mixtures, i.e., whether to establish small areas of monocultures (stands) of each species, i.e., "coarse grained" mixture or (stand) "mosaic," or whether the plantation (stand) should be established as a "fine grained" or "intimate" mixture of different and intermingled tree species. Kelty (1992) and Binckley et al. (1997) have reviewed the experience with tree mixtures, but only recently more information on mixtures was added from the tropics and subtropics (Kelty 2006; West 2006).

Because of the different biological characteristics of the various species, it is often assumed that species in mixture are able to occupy different niches in the ecosystem, which can overlap, and that mixtures are able to exploit the resource more economically than monocultures. Besides this complementary resource use between species, a facilitative improvement in nutrition of a valuable timber species growing in mixture with a nitrogen-fixing species may arise (Carpenter et al. 2004; Forrester et al. 2005, 2006; Kelty 2006; Bouillet et al. 2008; Piotto 2008; Oelmann et al. 2010). This means that the combination of species in mixtures and their growth and production can be greater than the production in monocultures and can improve economic returns (see also Montagnini and Piotto, this volume).

Kelty (2006) and West (2006) cited some examples of the better performance of mixtures, e.g., experimental plantations with mixtures of *Eucalyptus saligna* and *Falcataria moluccana* in Hawaii (DeBell et al. 1997), *Cedrela-Cordia-Hyeronima* in Costa Rica (Menalled et al. 1998), and *Grevillea robusta* and *Toona ciliata* in North Queensland, Australia (Keenan et al. 1995). Erskine et al. (2006) could demonstrate for the humid tropics of Australia that diverse plantations can achieve greater productivity than monocultures. In Costa Rica, *Jacaranda copaia* and *Vochysia guatemalensis* grew significantly faster in mixtures than in monocultures. A mixture of *J. copaia, V. guatemalensis*, and *Calophyllum brasiliense* produced 21% more merchantable volume than a monoculture of *J. copaia*, which grew the fastest among the three species (Petit and Montagnini 2006).

In a meta-analysis, Piotto (2008) found that mixed plantations did not have higher height growth rates, but that the diameter growth rate was higher in mixed plantations, with a moderate but statistically significant size effect. Nitrogen-fixing tree species had a positive effect on the diameter growth rate of non-fixing species, with a large and statistically significant size effect. This study suggests that mixing tree species generally increases plantation growth rate. Piotto stated furthermore that mixed tree plantations can play an important role in satisfying economic needs by shortening rotations yet adding other ecological benefits.

27.10 Nutrition of Tree Crops

Most tropical soils available for forest plantations (e.g., Oxisols, Alfisols, Ultisols, etc.) are naturally low in inherent fertility and deficient in nutrients such as Calcium (Ca), Magnesium (Mg), Potassium (K) and Phosphorus (P), but contain excessive quantity of Aluminium (Al), Iron (Fe) and Manganese (Mn) (Lal 1997). Also, Boron (B) deficiency is widespread in tropical plantations (pines and eucalypts) leading to shoot dieback, forking and multiple leaders. In addition, there are some measures of nutrient losses from plantation site during rotation, through leaching or chemical reactions in the soil by which nutrients become bound permanently to soil particles. Because the nutrients available for tree growth are vulnerable resources, it is important for losses to be minimised or replaced both in the short term during any one rotation and in the long term over many rotations or else tree growth and health will be compromised. This can be achieved through efficient site nutrient management (Nambiar and Brown 1997; Nambiar 2008). Unlike in natural forests, site nutrient management is particularly important in forest plantations given their intensive management system. However, since tropical forest soils differ greatly both in their ability to supply essential nutrients and in the extent to which they may lose nutrients through leaching, it is very difficult to generalize about how nutrients should be managed on any plantation site. Site nutrient management begins with assessment of the nutrient status of the site. At present, there is no simple and straightforward method of going to any site and easily assessing its nutrient status, often the assessment requires complex and long-term experimentation (West 2006). Using simply measured characteristics of a particular site to determine the nutrient deficiency of the site is usually difficult.

The ultimate test for determining the nutritional requirements for forest plantation site is through long-term field fertilizer experiments (Evans and Turnbull 2004). These experiments can be preceded by series of integrated short-term experiments (e.g., soil analysis, foliar analysis, plant tissue analysis, etc.), which can provide immediate information on the need for fertilization. This integrated approach has been used to identify deficiencies of macro- and micronutrients for tropical plantation tree species. However, it is possible to maintain or improve the fertility of forest plantation sites without fertilization, e.g., through efficient nutrient cycling and residue management.

27.10.1 Fertilizer Application

Although fertilization has not been a common practice in tropical forest plantation establishment and management, it is increasingly becoming an important means of improving tree nutrition, thereby accelerating the rates of nutrient cycling and tree growth on nutrient-deficient sites (Nwoboshi 2000; Evans and Turnbull 2004). With the intensification of silviculture through conversion to high density, high yielding, short rotation forest plantation systems, the use of fertilizers to improve the growth of forest stand will more and more become an accepted cultural practice. The huge success of tropical pine plantations in Australia has been attributed to the use of fertilizer (Simpson 1998). Singh and Singh (2001) reported greater growth rate in NPK fertilized plots than in unfertilized plots for nine tropical plantation species in India. In many parts of Asia, P-fertilizer application at planting is the standard practice for *Gmelina arborea* and *Acacia mangium* plantations (Evans and Turnbull 2004).

In plantation forestry, fertilizers are used to: (1) correct a specific deficiency; (2) establish a crop on nutritionally poor site or (3) stimulate tree growth (Evans and Turnbull 2004). In addition, fertilization will be needed to sustain rapid growth on all but the most fertile sites (Fox et al. 2006).

Nevertheless, fertilization in forest plantation is very expensive; it should only be used if the resulting gains in production can be justified economically, which can be judged by the size of growth response (i.e., higher yield or shorter rotation); in addition to the cost of application, interest rates, etc. It is very important that for each species the right kind of fertilizer is used and in the right amount and time to match the requirements of the tree. If excess amount of fertilizer is applied, the nutrients will be leached while insufficient quantity will not produce the desired growth response from the trees (Bruijnzeel 1997). This is because forest soils differ greatly in their ability to supply the nutrients and the extent to which they may lose nutrients through leaching (West 2006).

27.10.2 Time of Fertilizer Application

Fertilizers can be applied in a forest plantation at any one of four stages Evans and Turnbull (2004): (1) at establishment, usually within 3 months of planting; (2) during the post-establishment phase up to canopy closure when deficiencies begin to show; (3) at pole stage to boost thinning response and generally stimulate growth and (4) as a pre-felling application, 3–10 years before felling, to add increment before the end of rotation. In stages (1) and (2), care has to be taken that fertilization is not fostering competing weeds and grasses, too. No matter the stage, fertilizer application should coincide with the nutrient demand and expected growth response.

27.10.3 Plantation Establishment and Pre-canopy Closure Application

While application at planting enables seedlings to establish a vigorous root system essential for its good functioning, post-establishment application facilitates rapid growth rate and augments the high nutrient demand of the young seedlings. Fertilizer application at establishment is done to accomplish the following: correct known nutrient deficiencies; minimize planting shock to the seedlings by making nutrients liberally available; hasten crown closure; minimize the period the seedlings are subject to mortality. Some fertilizers are most conveniently applied at the time planting e.g., a one-off application of 15 g of borax which will correct boron

Table 27.2 Effect of fertiliser application on the growth of tropical forest plantation species 36 months after planting (adopted from Otsamo et al. 1995)

Species	Treatment	Height (m)	DBH (cm)	Basal area (m^2/ha)	Crown diameter (m)
Gmelina arborea	Plowing + NPK	5.3	7.2	11.3	3.4
	Plowing	3.7	4.1	4.0	2.9
Acacia mangium	Plowing + NPK	10.4	9.9	21.8	3.4
	Plowing	8.9	7.7	13.5	3.0
Paraserianthes falcataria	Plowing + NPK	9.3	10.3	22.8	4.3
	Plowing	5.0	5.5	6.3	3.8

deficiency for the rest of the rotation. Waring (1973) demonstrated that delaying the application of fertilizer on P-deficient sites could result in a loss of productivity that may never be recovered. This necessity for early fertilization also holds true for other elements, especially when they are so deficient as to restrict survival, establishment and early stand development.

Seedlings should not be fertilized the year of planting unless the fertilizer is a low release formulation, the fertilizer is thoroughly mixed into the surrounding soil prior to planting, or it is uniformly broadcast on top of the soil just prior to or after planting (McKenna and Woeste 2006). This is because the roots of young trees are damaged by contact with concentrated nitrogen fertilizers, and bare-root seedlings planted with fertilizer near the stem are prone to drought. Since fertilizers increase the growth of weeds that retard the growth of first-year seedlings, first-year seedlings should not be fertilized unless good weed control programme is planned (Pope et al. 1982; McKenna and Woeste 2006). Additional fertility will not help if factors such as water or light are limiting growth.

The higher the rate at which nutrients are supplied to seedlings, the faster their growth, thus the quantity of fertilizer needed by the seedlings will have to be increased steadily as the seedlings grow larger. Thus, throughout the early years of the plantation until canopy closure, fertilizer could be applied as often as practicable and the amount should increase progressively with time (exponential loading) to keep pace with the ever-increasing amount needed by the seedlings (West 2006). The application of NPK fertiliser significantly increased the growth of young plantations in Indonesia (Table 27.2). Seedling response to fertiliser application is dependent on tree species as shown by Bennett et al. (1996).

27.10.4 Post-canopy Closure Application

After canopy closure, leaf mass remains constant and litter breakdown and translocation of nutrients within the tree provides the nutrients necessary for new growth (Miller 1995). With the reestablishment of nutrient cycling system associated with forest canopy closure, the demand on soil nutrient capital is reduced and growth response to fertilizer application at this stage is usually variable (Nwoboshi 2000).

Provided nothing happens to disrupt this nutrient cycling, the nutrients in the plantation site should remain more or less in a steady equilibrium, thus eliminating any need for fertilization. However, if the effects of the fertilizers applied during pre-canopy closure stage wears off and stand growth stagnates, fertilizers may be applied after canopy closure and positive responses (in terms of additional growth) obtained. Thus, where the nutrients' demand of the plantation has not been fully satisfied during the early years, fertilizer application may be beneficial for post-canopy closure stand growth. Post-canopy closure fertilizer application should only be embarked upon where the value of the tree's economic component increases with stand age (Nwoboshi 2000). It is widely held that post-canopy closure fertilization will lead to a marked increase in growth rate if the stand is thinned at the same time with fertilizer application (Miller 1981; Carlyle 1995). Thus, to maximize returns on investment, post-canopy closure fertilization should be conducted in conjunction with a first or second thinning operation (Fox et al. 2006). To fully reap the benefits of post-canopy fertilization, it should be concentrated on the sites and tree species that will be most responsive to the treatment.

27.10.5 Soil Fertility Management Without Fertilizer

Although the years preceding canopy closure in forest plantations are characterised by major shift of nutrients from soil to tree biomass, the years proceeding canopy closure are characterised by efficient internal nutrients re-use, which implies that there can be a rapid recharge of soil exchangeable nutrients. The efficient management of this internal nutrient re-charge and nutrient cycling can lead to long-term forest plantation site fertility and site nutrient sustainability, thus precluding fertilizer.

Nutrient accumulation and export from fast-growing plantation sites has become an important consideration for long-term site quality and sustainability of production in short rotation, high-yielding forest plantation ecosystems. Some nutrients are lost through timber removal, while others are lost through the bark, branches, leaves, twigs, etc., especially where whole tree harvesting is practiced. Some researchers hold that the fast growth rate of tropical plantation species depletes site nutrient base and thus portends danger for long-term sustainability of production, while others opine that the decrease in productivity in successive rotations, where it exists, is due to inappropriate management practices such as soil compaction during site clearing and preparation, topsoil and litter repositioning, removal or burning of logging debris, harvesting methods, management of harvest residues, etc. (Khanna 1998; Kumar et al. 1998; Chen et al. 2004; Onyekwelu et al. 2006). Evans (1998) concluded that plantation forests are likely to be sustainable in terms of wood yield provided that good practices are maintained.

It is possible not only to sustain but also to increase productivity in successive rotations. However, this requires clear definition of end-use objective(s) and a holistic management view (Carle et al. 2002). To be sustainable, successive rotations will require the integration of the following strategies into the management plan (Carle et al. 2002): (1) tree improvement programs, (2) nursery practices, (3) site and species/provenance matching, (4) appropriate silviculture (site preparation, establishment, weeding, fertilizing, pruning, thinning), (5) forest protection, and (6) sound harvesting practices. Burning and excessive cultivation during site preparation, mechanical land clearing, soil compaction, inappropriate harvesting methods, and poor forest protection must be avoided. If well managed, increased productivity might result as was reported for second rotation stands of some species (Long 1997; Evans 1999a). If current plantations are not harvested by whole tree method and if successive stands are managed on long rotations, site nutrient capitals in successive rotations are likely to be maintained at the original level (Kimmins 2004; Onyekwelu et al. 2006). In addition, management of soil organic matter is of particular importance as it contains the bulk of the nutrients (Evans 1999a; Mathers and Xu 2003). Since the foliage, branches and bark, contains a reasonable amount of the nutrients, site fertility and productivity can further be improved by leaving these components on the site following harvest. Evans (1999a) concluded that "under certain conditions, nutrient export may threaten sustainability, but usually more important for maintaining site (plantation) quality are care with harvesting operations, conservation of organic matter, and management of weed environment. Plantation forestry appears to be entirely sustainable under conditions of good husbandry, but not where wasteful and damaging practices are permitted."

Using the example of *Pinus patula* plantations in Usuta forest in Swaziland, Evans (2002) demonstrated how a forest plantation can be productive over three rotations without the use of fertilizers (Tables 27.3 and 27.4). The report further demonstrated that productivity of plantations can be improved in successive rotations if they are properly managed. Also, the case study by Onyekwelu (2011, in this volume) on the sustainability of site productivity in *Gmelina arborea*

Table 27.3 Comparison of second and third rotations of Pinus patula on granite and gneiss derived soils at 13/14 years of age (Evans 2002)

Rotation	Stocking (n/ha)	Mean height (m)	Mean DBH (cm)	Mean tree vol. (m^3)	Vol/ha (m^3/ha)
Second	1386	17.5	20.1	0.205	294
Third	1248	18.7	21.2	0.233	326
% change		+7.1	+5.6		+11.0

Table 27.4 Comparison of second and third rotations of Pinus patula on gabbro dominated soils at 13/14 years of age (Evans 2002)

Rotation	Stocking (n/ha)	Mean height (m)	Mean DBH (cm)	Mean tree vol. (m^3)	Vol/ha (m^3/ha)
Second	1213	16.7	20.0	0.206	244
Third	1097	16.8	21.7	0.227	255
% change		+0.05	+8.3		+4.6

plantations demonstrates how the nutrient status of a tropical forest plantation site can be built up to its original level without the use of fertilizer.

27.11 The Dynamics of Stand Growth

Forest tree growth in the humid tropics is continuous due to the long duration of rainy season and because precipitation exceeds or equals potential evapotranspiration, thus the soil is continuously moist and there is hardly any season in which the soil dries out (Nwoboshi 1982; Evans and Turnbull 2004). This favourable condition results in a fast growth rate of trees. Many tropical forest plantation tree species, especially the exotics, are noted for their fast growth rate and the ability to attain rotation within a relatively short time, a situation that makes them attractive for short rotation forestry. This fast growth rate could be attributed to the ability of the trees to effectively utilize site and environmental resources.

Forest stand growth is dynamic, and continually changing. Forest stand dynamics is defined as the changes in forest structure, function and composition through time (Oliver and Larson 1996). Thus the dynamism of forest stand leads to changes in stand density, stand composition and stand structure. The knowledge of forest stand dynamics is applied in many areas of forest plantation establishment and management (Cannell and Last 1976; Kerr 1999). Several characteristics of the growth dynamics in forest plantations set them apart from uneven-aged stands and other types of forest structure (Bettinger et al. 2009). These differences are summarised in Table 27.5. Forest plantation stand does not usually end with the same number of trees it had at establishment. Although seedlings are almost uniform at stand establishment, the dynamic nature of forest stand results to the differentiation

Table 27.5 Comparison of several characteristics of the growth dynamics of forest plantations and uneven-aged forests

Growth characteristics	Forest plantations	Uneven-aged forest stands
Tree per unit area	Decreases with age	Varies through time
Mortality rate of stems	Decreases with age	Stays relatively constant over time
Mortality rate of volume	Decreases with age	Stays relatively constant over time
Height of canopy	Increases with age, then plateaus	Stays relatively constant over time
Canopy cover	Ranges from non to full	Ranges from full to one containing gaps
Average tree diameter	Increases with age	Fluctuates with harvest entries and mortality
Diameter distribution	Bell-shaped curve	Reverse J-shaped curve
Basal Area	Increases with age, then plateaus	Fluctuates with harvest entries and mortality
Timber growth rate	Rises, peaks, then declines	Stays relatively constant over time
Timber yield	Increases with age, then plateaus	Fluctuates with harvest entries and mortality

Source: Bettinger et al. 2009

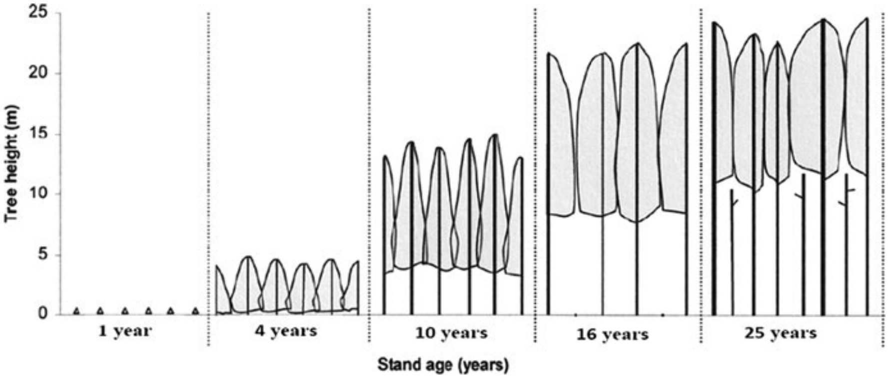

Fig. 27.2 Schematic presentation of forest stand development from establishment to maturity. The diagrams represent spacing, tree height and crown dimensions. The diagram for 16 years stand demonstrates the effect of thinning, which has resulted in a smaller number of trees but with large, uniform crowns (Wilson and Leslie 2008 – modified)

of trees into different tree sizes (Fig. 27.2). The change from one stage of development to another in forest ecosystem is influenced by species, site, climate and exogenous disturbance events (Wilson and Leslie 2008).

Four major stages can be distinguished in forest plantation development: (1) seedling stage, (2) sapling stage, (3) pole stage and (4) mature stage. The age at which each stage occurs depends on species and location. Fast growing species attain and transit from one stage to the other earlier than slow growing ones. However, each stage of the sequence may be disrupted by management interventions or natural disturbance (Kimmins 2004; Johnson and Miyanishi 2007). At the seedling stage, tree density is often very dense, usually in thousand(s) per hectare depending on the purpose of management. As individual trees become larger and transit to the other stage, the number of trees decreases due to the natural mortality (self thinning) of trees that had become weaker and overtopped from competition between the trees.

In even-aged stands, the differences in the performance of individual trees quickly sort them into crown classes, which apart from competition, could be caused by genetic differences between seedlings, site productive capacity, initial planting density, growth rate, differences in establishment after planting, and damage to seedlings (Kental 1988; Naidu et al. 1998; Evans and Turnbull 2004). Onyekwelu (2001) grouped trees in *Gmelina arborea* and *Nauclea diderrichii* plantations in Nigeria into two broad crown classes: suppressed and dominant classes. After crown closure in even-aged stands, growth and vigour of trees are strongly correlated with crown class, with trees in the dominant crown class performing better than those in the suppressed crown class (Fig. 27.3). This is because dominant trees compete better and capture more site resources than suppressed trees. Thus, crown expansion early in the life of the plantation and its subsequent restriction are critical in the growth dynamics of individual trees and their competitive advantage (Evans and Turnbull 2004).

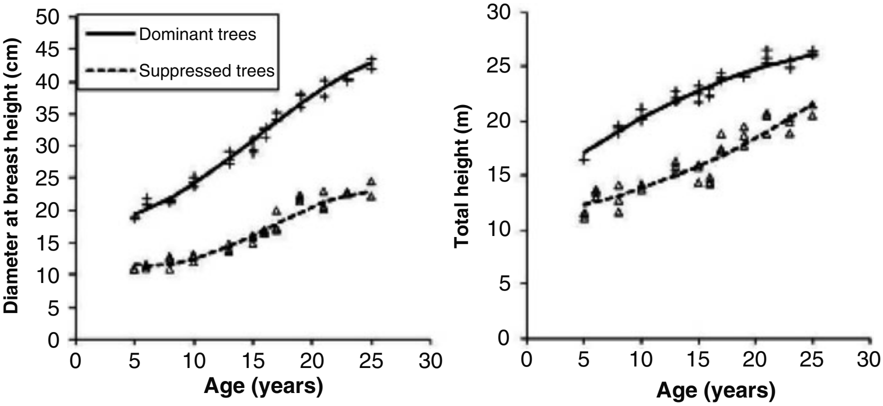

Fig. 27.3 Development of stand diameter at breast height and total height for two crown classes in Gmelina arborea plantations

27.12 Growth and Yield of Important Tropical Plantation Species

Fast-growing tropical plantation species such as *Eucalyptus* spp., *Albizia procera*, *Pinus caribaea* [*not radiata*], *Acacia mangium*, *Gmelina arborea*, etc. have high diameter growth which begins early in the life of the stand. Some of these species can attain a diameter (at breast height) of about 7 cm during the first year of growth (Lugo et al. 1990), but on an average, they are capable of maintaining rapid DBH growth rate of 3–5 cm/year (Hopmans et al. 1990; Panitz and Yaacob 1992; Onyekwelu et al. 2003a, b).

Rapid height growth has also been reported for species, like *A. mangium, Casuarina equisetifolia, Eucalyptus* spp*, Gmelina arborea, Tectona grandis,* etc., which can maintain height increment of between 2.7 and 6.0 m/year (Lugo et al. 1990; Gonzalez and Fisher 1994; Mok et al. 1999; Morataya et al. 1999; Krishnapillay 2000; Henri 2001; Onyekwelu et al. 2003a, b). However, this vigorous diameter and height growth are only maintained by the species at young ages. It is evident from reports in literature (see Table 27.6) that both diameter and height growth of tropical plantation species are rapid during the early years of growth and plateaus with increase in age (Bettinger et al. 2009). For most of these species, the maximum diameter and height increment are attained before the age of 10 years.

The high and rapid growth rates of tropical plantation tree species, coupled with their high density and intensive silviculture, translates to high basal area and volume production (Table 27.6), which has contributed to making them very important in meeting the world's growing demand for wood products, especially timber. Tropical plantations have been noted to have the ability of producing high amount of biomass within a relatively short period of time. Tropical forest

Table 27.6 Summary of growth and yield of selected tropical forest plantation species

Age (years)	Density (per ha)	Mean height (m)	Mean DBH (cm)	Basal area (m^2/ha)	Volume (m^3/ha)	MAI (m^3/ha/year)	References
Acacia mangium							
3	–	8.1	19.3	63.0	187.5	–	Newaz et al. (2005)
6	–	19.7	27.1	80.5	376.3	–	
9	–	25.5	30.4	87.4	474.7	–	
12	–	28.9	32.1	91.0	533.1	–	
1.5	1,667	11.5	11.0	–	76.9	35.0	Mok et al. 1999
3.7	1,533	18.7	14.0	–	176.9	47.8	
5.2	1,200	23.8	16.8	–	246.1	46.4	
Tectona grandis							
4.5	1,324	13.8	12.29	19.00	119.24	26.52	FAO (2002a, b)
10.5	468	22.5	23.9	36.88	329.85	31.41	
15	288	24.6	30.00	48.16	438.05	29.20	
20	249	25.3	34.5	54.12	505.12	25.30	
Gmelina arborea							
5	1,232	14.1	15.1	24.3	242.7	–	Onyekwelu et al. (2003a)
10	1,147	17.0	18.0	33.8	418.0	–	
15	1,184	18.0	21.1	48.3	630.7	–	
21	864	22.9	30.2	71.5	1165.0	–	
25	874	23.2	33.6	89.5	1519.3	–	
Pinus caribaea[a]							
6	–	9.0	–	–	37.5	6.2	Adegbehin et al. (1988b)
16	–	20.1	–	–	275.3	17.2	
26	–	27.0	–	–	615.0	23.6	
34	–	29.7	–	–	814	23.9	
Eucalyptus tereticornis[a]							
4	–	14.0	–	–	47.9	12.0	Adegbehin et al. (1988b)
10	–	23.4	–	–	131.2	13.1	
15	–	24.8	–	–	215.9	14.4	
20	–	25.1	–	–	295.0	14.8	
22	–	25.2	–	–	322.5	14.7	
Nauclea diderrichii							
5	667	9.0	9.6	4.12	28.27	5.65	Onyekwelu et al. (2003b)
9	587	13.1	15.7	13.86	121.40	13.49	
15	443	15.0	19.0	15.23	156.46	10.43	
24	491	21.1	24.3	23.18	344.37	14.35	
30	496	23.6	29.3	40.08	475.52	15.85	

[a] Height of dominant tree

plantations possess the capacity of producing between 3 and 10 times greater commercial biomass or timber per unit area than natural forests (Pandey 1995; Evans 1999c; Evans and Turnbull 2004). For example, while the maximum mean annual volume increment (MAI) in a natural tropical forest in Nigeria is 5 m^3/ha/year, that of an adjacent *N. diderrichii* (indigenous species) and *G. arborea* (exotic species) plantations were 16.0 and 51.5 m^3/ha/year, respectively (Lowe 1997; Onyekwelu 2001). Some tropical plantation species such as *Eucalyptus* spp., *Acacia mangium*, *G. arborea, Pinus* spp*,* etc., have MAI between 30 and 55 m^3/ha/year (FAO 2001a; Onyekwelu 2001; Evans and Turnbull 2004).

Given proper planning, good management and application of tree breeding, much higher yield is possible. For example, the maximum MAI of genetically improved *Eucalyptus grandis* plantations in Brazil and Cameroon was reported to range between 70 and 89.5 m^3/ha/year (Betancourt 1987; Pandey 1995). However, this high productivity is not applicable to all tropical forest plantation species. Some species (Teak, *Casuarina equisetifolia,* etc.) could have MAI as low as 2–4 m^3/ha/year (Lamprecht 1990; Enters 2000; FAO 2001a). The poor performance is mainly a result of low inputs and poor management, coupled with yield-reducing factors such as illicit removal, fire, pest infestation and disease outbreaks (Enters 2000). Given a good combination of high quality reproductive material, rigorous site selection and application of known technologies, good planting materials adapted to the site, appropriate silvicultural practices and improved protection, the yield of tropical forest plantation species could be increased considerably.

27.13 Thinning

There is a dearth of information in literature on thinning in tropical forest plantations. This can be attributed to two main reasons: (1) extensive forest plantation development in the tropics is a recent phenomenon and thus, time has been insufficient to undertake long-term thinning experiments (Evans and Turnbull 2004), (2) a high percentage of early extensive plantations were established as fuelwood and pulpwood plantations in which thinning programmes were not intended (Onyekwelu et al. 2003a). In addition, the lack of permanent sample plots, coupled with rising deforestation and encroachment into forest plantation stands in the tropics have hindered long-term thinning investigations.

The practice of thinning is one of the important and common basic tools of silviculture administered to improve tree growth rates. It has received increased attention with the advent of intensive forest management. Although unthinned stands usually show the highest stand density (relative mean basal area = 100%), such stands respond to thinning with a significant increase in volume increment. Optimum stand density (optimum basal area) is reached when the annual volume increment is at a maximum. A further reduction of basal area leads to increment losses (Pretzsch 2009). In practice, foresters seek to develop thinning schedules for an optimization of management and productivity.

Thinning involves practices ranging from light removal of small understory trees to heavy removal of dominant overstory trees. By implication, thinning is a controlled process by which inferior trees are progressively eliminated and better ones are encouraged to develop so that only the best candidate trees will remain for final harvest. Thus, it promotes the growth of the best individual trees in a stand by removing damaged, diseased, or deformed trees and concentrating growth on fewer, high-quality trees. Thinning is a complex biotechnical measure that needs careful planning. Before administering thinning, account should be taken of the stand's biological characteristics, e.g., age, density, composition and productivity, as well as site conditions, e.g., topography, climate, soils, rate and character of anthropogenic disturbance and wildfire dynamics (Danilin 2006).

The main objectives of thinning in forest stand include (Evans and Turnbull 2004): (1) to reduce the number of trees in a stand so that the remaining ones will have more space for crown and root development, thereby encouraging stem diameter increment and reach utilizable size sooner, (2) to remove dead, dying, diseased and any other tree that may be a source of infection for or cause damage to the healthy trees, (3) to remove trees of poor stem form, e.g., crooked, forked, roughly or heavily branched trees, etc., so that all future increments are concentrated on trees with good stem form, (4) to favour the most vigorous trees with good stem form which are likely to be part of the final crop and (5) to provide an intermediate financial return from thinning. Other objectives of thinning include: maintaining light level beneath a forest stand to provide vegetation for grazing, reduction of wildfire risk, encouraging ground forest flora, providing poles and posts, increasing recreational and amenity value of the forest, etc.

The main benefit of thinning is to increase economic gain, which may be achieved through improving the value of the residual stems (products), offsetting the expense of carrying establishment costs to rotation age, and/or increasing stand utilization. Large trees are more valuable than small ones because the resulting products from large trees have a greater value than those from small trees, particularly ones below sawlog size. Other benefits of thinning being recognized are increase in light and nutrient availability, risk reduction for insect infestations, disease epidemics, and damage from abiotic agents, etc.

27.13.1 Timing of Thinning Operation

The timing of thinning operations in forest plantations, especially the age at which thinning should commence is one of the most important and critical management and silvicultural decision in a rotation. Making an early start will affect stem quality and crown development and trees respond too slowly when stands are thinned too late (Lewis et al. 1976; Nwoboshi 1982; Dean and Baldwin 1993; Morataya et al. 1999). Wrong or poor timing of thinning operation can have profound effect, not only on the current status of the forest (Geoff et al. 2006), but also on future forest conditions. Thinning operations are usually administered between canopy closure and culmination of Mean Annual Increment (MAI) (Strin 1990) and should be determined by management objectives, site quality, stand density, probability of subsequent thinning operations, rotation length, etc. If sawlogs, veneer logs or multiple products are the objective(s) of management, early thinning may be required to increase the proportion of large, high quality, merchantable stems at final harvest, especially in short rotation, high density stand and on good sites with high growth potential. It is recommended that the final thinning be carried out before the peak of MAI.

First thinning can be administered earlier on land with high site quality than land with low site quality. However, the significance of site in administering the first thinning is better understood when considered with stand density. Time of the first thinning, even on the best sites, can be delayed in stands with poor survival and low initial planting density. The first thinning is best administered as soon as the seedlings are well established (i.e., shortly after canopy closure), prior to overcrowding and competition, reduction in diameter growth, heavy mortality and before the live crown ratio is reduced to below 35% of total height. Depending on species and site condition, this is usually between ages of 2 and 5 in tropical forest plantations (Dupuy and Mille 1993; FORMECU 1999), before the trees experience severe intraspecific competition and while they are still small enough to permit thinning with relatively light equipment such as a rotary mower or light chopper. Hughell (1991), Dupuy and Mille (1993), FORMECU (1999) and Morataya et al. (1999) recommended 3–4 years after plantation establishment for the first thinning in Gmelina plantations while Onyekwelu et al. (2003a) recommended 5 years for first thinning in Gmelina stands. Florence (1996) recommended that *E. grandis* plantations should be thinned within 2–4 years of canopy closure. However, care must be taken not to administer first thinning too early, as it may encourage the development of epicormic branches and slow down self-pruning. On the other hand, delayed thinning may result in a decline in release potential, higher risk of windthrow due to the spindly nature of trees and volume lost to natural mortality.

Experience in thinning in forest plantations in Latin American countries has increased in recent years (e.g., Galloway et al. 1996; Morataya et al. 1999; Kanninen et al. 2004). Based on their findings, Morataya et al. (1999) concluded that a delay in thinning (especially first thinning) is not desirable in *Gmelina* and Teak plantations, since both attained the greater portion of their diameter growth during the first 6–8 years. Amakiri and Nwoboshi (1986) showed that thinning reduced nutrient uptake from the soil, their accumulation in the vegetative part as well as the rates of litter decomposition and release of K and N in 25-year old *T. grandis* plantation. In Costa Rican teak plantation, highest individual tree growth was obtained when the plantations were thinned at 60% thinning intensity applied at the age of 4 years, and the two consecutive 25% thinnings at the ages of 8 and 12 years (Kanninen et al. 2004). Thinning had a significant effect on the diameter and height growth of individual trees in both pure and mixed plantations, with the thinned stands having greater diameter and height growth than unthinned ones. However, the unthinned stands generally had higher basal area and volume growth (Piotto et al. 2003). In a 24-year-old plantation of *Acacia koa*, thinning in combination with grass control and P fertilization significantly increased annual diameter increment by 118%; thinning alone did not produce a significant increase in diameter increment (Scowcroft et al. 2007). In a thinning experiment in Nigeria, Lowe (1976) found that 5 years after implementing a moderate and a heavy thinning operation, trees in a 20-year-old plantation reacted positively to thinning in terms of individual tree growth and concluded that thinning reinforced rather than changed the pattern of discriminative growth within the stand, even for the heaviest thinning. Other

Table 27.7 Thinning regimes used in the management of some tropical forest plantation species in various parts of the tropics

Tree species	Location	Age (years)	Stocking (tree/ha)	Reference(s)
Tectona grandis	Costa Rica	0	1,111	Pérez and Kanninen (2005)
		4	556	
		8	333	
		12	200	
		18	150	
		24	120	
		30	Clear-fell	
Gmelina arborea	Nigeria	0	1,300	Onyekwelu et al. (2003a)
		5	400	
		10	330	
		15	280	
		20	Clear-fell	
Eucalyptus grandis	South Africa	0	1,370	Schönau and Coetzee (1989)
		3–5	750	
		7–9	500	
		11–13	300	
		25–30	Clear-fell	
Nauclea diderrichii		0	1,111	Dupuy and Mille (1993)
		5	400–500	
		9	200–250	
		15	130–170	
		30–40	Clear-fell	
Pinus caribaea var. hondurensis	Queensland Australia	0	746	Evans and Turnbull (2004)
		2–3	700	
		22	500	
		30	300	
Araucaria cunninghamii	Queensland Australia	0	833	Hogg and Nester (1991)
		25	400	
		45–50	Clear-fell	

thinning experiences in tropical forest plantations include: Schönau and Coetzee (1989), Yahya (1993), Medhurst and Beadle (2001), Mabvurira and Pukkala (2002), Medhurst et al. (2003)and Pérez and Kanninen (2005). Based on experience and scenarios, thinning regimes have been recommended for various tropical forest plantation species. These recommendations are summarised in Table 27.7.

27.13.2 Positive Effects of Thinning

Most of the positive effects of thinning are obtained by increasing the amount of growing space available to residual trees. The temporary elimination of competition between individual trees for light, soil moisture and nutrient and the quick response of the trees to take advantage of the additional growing space result in the following main positive effects.

1. Trees in pure and mixed plantations of some tree species (*Terminalia amazonia*, *Vochysia guatemalensis*, *Jacaranda copaia*, *Virola koschnyi*, *Vochysia ferruginea*, *Calophyllum brasiliense* and *Genipa americana*) responded significantly to thinning with increased diameter growth (Piotto et al. 2003). Maximum diameter growth of residual trees is obtained when thinning is conducted early, at the onset of competition between trees. When plantations are thinned too late, there is little or no gain in diameter growth as residual trees respond too slowly to the additional growing space. For example, the diameter increment in *Gmelina arborea* and *Tectona grandis* plantations thinned late, which did not differ significantly with that of unthinned stands (Morataya et al. 1999).
2. Tree growth is strongly correlated with the amount of carbohydrate produced, which is a function of crown size and the ability of the root system to supply water and essential nutrients (Nwoboshi 1982). When trees are released from competition through thinning, their roots quickly respond with rapid lateral extension, thus most of the initial increases in growth following thinning are due mainly to increased moisture and nutrients supply. Also, the additional growing space surrounding a tree after thinning induces active growth of shoots and foliage, which results in outward crown expansion. Morataya et al. (1999) showed that early thinning in Gmelina and Teak plantations in Costa Rica resulted in more average foliage per tree than in unthinned stands.
3. Reduced susceptibility of stands to disease, insect and fire attack due to the removal of diseased and infested trees. Dense stands with slow growth and reduced tree vigour are more subsequently prone to insect and disease attack than healthy trees in well-spaced stands (Nebeker et al. 1985). Uninfested trees are generally larger, with thicker bark, greater crown/bole ratios, larger crowns, and faster growth rates. Good forest management with scheduled thinning has continued to be recognized as a means of maintaining healthy stands and promoting resistance to insect and disease attack (Nebeker et al. 1985).
4. Some genetic improvement may be achieved through thinning (Nebeker et al. 1985). The removal of diseased trees or trees with undesirable characters, e.g., bad growth form, by means of thinning prior to regeneration of the stand, can minimize undesirable traits in the following generation.
5. Other positive effects of thinning include:
   - Improved access for equipment
   - Enhanced wildlife habitat through increased herbaceous ground cover

27.13.3 Negative Effects of Thinning

1. Thinning facilitates the shaded low branches at the base of the live crown of residual trees to receive more light and remain alive longer, thus resulting in larger lower branches, higher live-crown ratio and delayed natural pruning. In addition, large epicormic branches may develop along the stem. While this may be desirable in maintaining rapid growth of individual trees, it adversely affects

wood quality and product value due to the increased size and number of knots in the timber (Geoff et al. 2006; West 2006).
2. Vulnerability of residual trees to wind or rainstorm may increase by increasing turbulence within the stand and allowing more sway of trees (Moore and Maguire 2005). Increased swaying increases the force exerted on the ground, thus increasing the risk of uprooting (Cameron 2002). The gravity of this damage varies with the rooting characteristics of the species. Shallow rooting species are more susceptible than deep rooting species. Also, tree swaying in a thinned stand substantially increases the amount of tension wood that develops in the stem of residual trees (West 2006).
3. There could be felling-related damages in the form of branch breakage, bole wounding, root breakage, bending and breakage of whole trees. The degree of felling-related damage is influenced by method of felling, felling equipment and its configuration, tree species, density, stand age, site climatic conditions (e.g., wind or rainstorm).
4. Reduction of the photosynthetic surface area in a stand, and thus an immediate drop in production per unit area occurs. Since thinning does not raise site production potentials, it does not raise the overall or total production of a stand. When compared to an unthinned stand, the total stand production of a thinned stand is less.

27.13.4 Methods of Thinning

Of the various thinning methods four have been widely accepted: (a) Low thinning (thinning from below), (b) Crown/high thinning (thinning from above), (c) Selection thinning, and (d) Mechanical (systematic) thinning (Fig. 27.4). Thinning methods that do not conform to any of the above are sometimes referred to as “free thinning” (Province of British Columbia 1999). Thinning method influences the diameter distribution of a stand (Fig. 27.4) as well as the subsequent growth of the residual stand because all the trees in the stand are not equally vigorous or able to respond equally to release. Thinning method also affects the revenue derived from the thinning. Methods that remove large trees will often be more profitable than those that remove smaller trees. For even-aged, single-species plantations, tree vigour is closely related to their position in the canopy. The method that results in the greatest growth response and best quality trees may also be the most expensive. Thus, the best choice of thinning method will often represent a compromise between cost and quality. Often a combination of thinning methods is used during a single operation due to the irregularity of the relative crown positions of the trees in most stands.

Details of the different methods and their characteristics are described comprehensively in e.g., Nyland (1996), Smith et al. (1997), Graham et al. (1999) and Evans and Turnbull (2004).

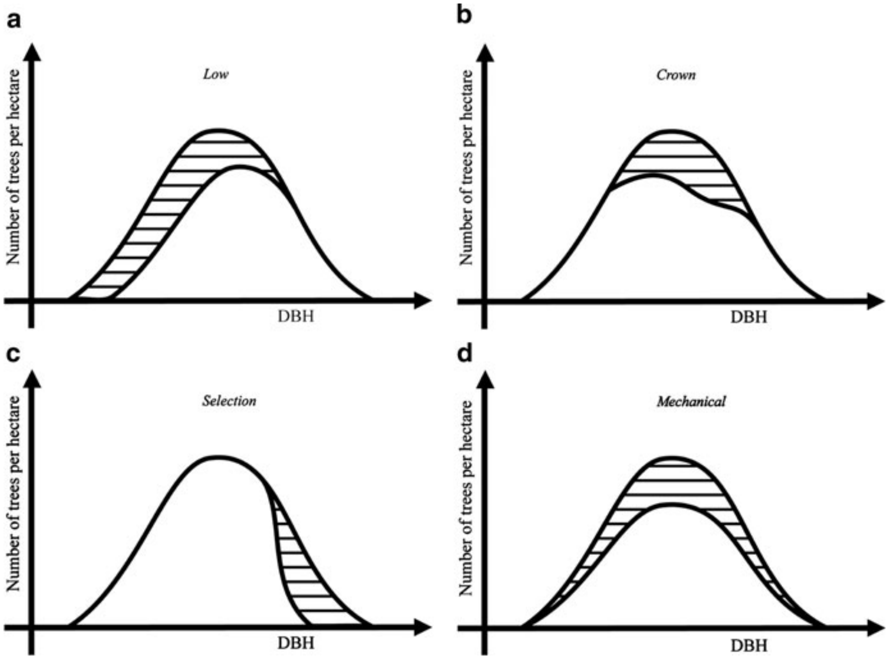

Fig. 27.4 The effect of different methods of thinning on diameter distribution of a forest stand (Evans 1982, Smith et al. 1997 – modified)

27.14 Pruning

Pruning is a silvicultural technique that removes branches in order to improve wood quality. It is often applied in two subsequent steps to different heights of the trunk, first in the so-called "low pruning" (3–4 m high) and second in a "high pruning" (6–8 m high) which takes place at the same time as selective thinning is carried out.

Montagu et al. (2003) comprehensively reviewed the biological and silvicultural basis for producing clear wood from planted eucalypts. Despite the "self-pruning" nature of many eucalypts species, the amount of clear wood able to be produced by untended stands is limited. Pruning increases the proportion of clear wood. The authors suggested that pruning should be undertaken while branches are small and alive. Minimising decay entry by ensuring branches are small when pruned is an important control measure. Pruning in conjunction with thinning can substantially increase the volume of clear wood produced.

27.15 Protection Against Pests and Diseases

Plantation forests should be managed with the objective of keeping them in a healthy, productive condition, one in which pests and disease are kept at low levels and do not interfere with management objectives.

Interestingly, the overall conclusion of a study by Nair (2001) is that while plantations are at greater risk of pest outbreaks than natural forests, plantations of exotic species are at no greater risk than plantations of indigenous tree species, because the exotic status is only one among the many determinants of pest outbreak.

Nair (2007) distinguishes three categories of insect pests associated with plantation tree species: nursery pests, sapling pests and pests of older, established plantations. Focussing on the latter serious pests include defoliators, sap suckers and stem borers. Leaf-feeding insects causing serious damage occur on *Dalbergia sissoo,* Eucalypts*, Falcataria moluccana* and *Tectona grandis*, among others. Nair (2007) mentioned, for example, the case of the caterpillar *Hyblaea puera,* which caused annual defoliation in some teak plantations in Asia that resulted in a 44% loss of the wood volume increment. According to Nair, this pest is becoming increasingly important in exotic teak plantations in Latin America.

Efforts to establish plantations of the African mahoganies *Khaya anthotheca* and *K. ivorensis* to sustain timber supply have been discouraged by the shoot borer *Hypsipyla robusta* moore. It was hypothesized that there is a shade level at which *Hypsipyla* attack and branching are reduced, but height growth is adequate (Opuni-Frimpong et al. 2008). The authors reported on the growth of these African mahoganies and *Hypsipyla* attack under three different forest canopy shade levels: open, medium shade, and deep shade. *Hypsipyla* attack on *K. anthotheca* was 85%, 11% and 0% attack in the open, medium and deep shade treatments, respectively. However, growth in medium and deep shade was slow, which limited the use of this strategy for controlling *Hypsipyla* attack. In Latin America, attack by *Hypsipyla grandella* Zeller is the main reason for the limited success of plantations of *Cedrela odorata* or *Swietenia macrophylla*, two other prominent and highly valuable representatives of the Meliaceae family. Because there is no viable method of pest control currently available, provenance and progenies of the two species are assessed for genetic variation in susceptibility to pest attack, especially for high foliar proanthocyanidin content, which may provide scope for selection for the ability to tolerate attack (Newton et al. 1999).

Wingfield and Robison (2004) reviewed insect pests and diseases of *Gmelina arborea*. In plantations within the natural range of the species, insects have caused substantial damage. Among these, the defoliator *Calopepla leayana (Chrysomelidae)* appears to be most important. No serious insect pest problems have been recorded where *G. arborea* is grown as an exotic, but some fungal pathogens have been introduced into those areas. Among these, leaf spot caused by *Pseudocercospora ranjita* is most widespread although it has not caused any substantial damage. A serious vascular wilt disease caused by *Ceratocystis* *fimbriata* in Brazil has caused the most significant failure of *G. arborea* in plantations.

The expansion of acacia plantations in Southeast Asia has also increased concern regarding the threats posed by diseases, such as those caused by fungal pathogens (Rimbawanto 2002; Lee 2004). Surveys between 1995 and 1996 resulted in a review of the current knowledge of the pathology of *A. mangium, A. auriculiformis, A. crassicarpa* and *A. aulacocarpa* in tropical areas of Southeast Asia, India and Australia (Old et al. 2000). The five most significant diseases according to the survey are root rot (*Ganoderma* complex), stem canker (e.g., *Lasiodiplodia theobromae*, *Botryosphaeria* spp.), pink disease (*Corticum salmonicolor*), heart rot (wood decay fungi) (Barry 2002; Lee 2002) and phyllode rust (*Atelocauda digitata*).

Old et al. (2003) presented a comprehensive review on Eucalyptus diseases in Southeast Asia. Barber (2004) mentioned Cryphonectria canker, Eucalyptus rust, Mycosphaerella leaf disease and the diseases caused by *Phaeophleospora destructans* and *P. epicoccoides* as major threats for Indonesia's eucalypt plantation forests. Eucalyptus rust is caused by *Puccinia psidii*, which occurs predominately in Latin America, and is a remarkable disease in that the pathogen is not known on eucalypts in their places of origin. It has apparently originated on native Myrtaceae in South America and is highly infective on some *Eucalyptus* spp. planted there. *P. psidii* causes one of the most serious forestry diseases in Brazil and is considered to be the most serious threat to eucalypt plantations worldwide (Coutinho et al. 1998).

A pantropically occurring pathogen is *Phellinus noxius* (Ramsden et al. 2002), which was also recently identified to cause a basal root rot in teak plantations and brown root disease in *Azadirachta excelsa* plantations in Malaysia (Mohd et al. 2005, 2006).

For further information, we recommend consulting region-specific reviews on insect pests and fungal diseases of plantation forests, e.g., for the Asia-Pacific region (Ramsden et al. 2002), Tanzania (Nsolomo and Venn 1994) or Uganda (Nyeko and Nakabonge 2008).

Integrated pest management (IPM) is a framework of decision making and action tools designed to maintain and improve forest health. Pest and disease-monitoring ensures early detection of potential problems. Combined with analysis of the economic, social and ecological impacts of pests, a sound basis on which to decide for or against control is derived. Two basic strategies, prevention or direct suppression, each with a range of tactics, can be applied. Prevention consists of actions taken to make trees and forests less hospitable to the build-up of pests and diseases and/or preventing new introductions. Direct suppression consists of biological, chemical or mechanical tactics designed to reduce pest and disease populations and subsequent losses. IPM systems consist of a combination of monitoring and action tools designed to reduce pest-induced losses. These systems are continuously evolving and are capable of accepting new technologies, as they become available (FAO 2001a, b, c, d).

27.16 Rotation

Rotation length is the total number of years between establishment and final felling that a forest plantation is allowed to grow. It is an important tool for controlling tree size: the longer the rotation, the larger the tree (Evans and Turnbull 2004). Rotation length (age) in forest plantation management is determined by various factors including: species, site quality and environmental conditions, rate of wood and fibre production, desired wood and fibre properties, regeneration method, tending operations, profitability. Profitability is the over-riding factor that determines rotation age. A properly designed forest plantation investment should encompass growth rates and wood properties in an investment equation that marries costs and prices to determine the optimal length of time that the plantation investment should be "held" (FAO 2001b).

A wide array of rotation ages are used in tropical plantations (Table 27.8). As low as 3–5 years, rotation is used in the establishment of high density, short rotation plantations, especially energy plantations. Up to 100 years rotation may occur for very high-value timber plantations such as teak. Generally, management inputs under very long rotation regimes have to be very low to be financially viable and the forest may revert to semi-natural status long before it is harvested (FAO 2001b). The length of rotation is closely related to the tree species and the proposed end-use of the plantation products. Generally, the shortest rotation age is found in energy

Table 27.8 Common rotation ages for various tropical forest plantation species in different parts of the world

Tree species	Rotation length (years)		
	Energy plantation	Pulpwood/pole plantation	Timber/veneer plantation
Eucalyptus camaldulensis	4–8	7–15	–
Eucalyptus grandis	5–12	5–15	–
Eucalyptus globulus	3–5	8–19	–
Eucalyptus deglupta	–	10–12	–
Gliricidia sepium	5–8	–	–
Acacia auriculiformis	5–10	–	–
Acacia mangium	5–14	–	–
Gmelina arborea	8–15	8–15	15–25
Pinus caribaea	–	10–20	15–25
Pinus patula	–	12–20	15–30
Leucaena leucocephala	5–8	–	–
Paraserianthes falcataria	9	5–15	–
Tectona grandis	5–10	–	30–70
Araucaria angustifolia and *A. cunninghamii*	10–12	–	40–90
Casuarina equisetifolia	8–15	–	30–50
Swietenia macrophylla	–	–	40–60
Terminalia ivorensis and *T. superba*	–	20–25	30–60
Nauclea diderrichii	–	15	35–40

plantations while the longest rotation is common in timber/veneer log plantations (Table 27.8).

Several types of rotations have been identified by foresters (Fenton 1967), which include:

1. Physical rotations
2. Silvicultural rotations
3. Technical rotations
4. Financial rotations
5. Rotation of maximum volume production

In the tropics, technical rotation and rotation of maximum volume production are most common. Technical rotation yields the most out-turn of a specified size and type to satisfy a particular end-use (Evans and Turnbull 2004). Applying technical rotation requires that a certain limit (usually lower limit) be set and that the trees be harvested when the limit is attained. However, upper limits may sometimes be used. It may sometimes be necessary to extend technical rotation to obtain better product. For example, it is important to allow pruned stands to grow beyond a specified technical rotation so that a worthwhile layer of clear, knot-free wood is laid down.

It has been shown that the culmination age of mean annual increment (MAI) in forest plantation corresponds to the age of maximum volume production. This MAI culmination age can be used as the biological basis for determining rotation age, if the objective of management is timber production (Clutter et al. 1983; Adegbehin et al. 1988a, b; Smith et al. 1997). This rotation is complete when the current annual increment (CAI) of the stand falls to MAI level (Fig. 27.5). It is the rotation of maximum volume production if the stand is felled at this point

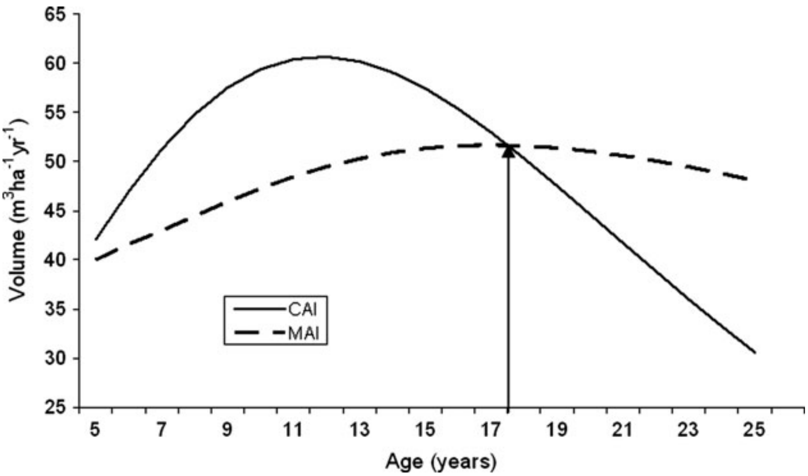

Fig. 27.5 Relationship between current (CAI) and mean annual volume increment (MAI) in Gmelina arborea plantations in Nigeria

(Evans and Turnbull 2004). The rotation age based on the culmination of MAI is attractive because it tends to realize the maximum growth potentials of a species on a particular site. This is because for a given parcel of land, it is the harvest age that will maximize total volume production from a series of rotations. Due to the high yield occasioned by the fast growth rate and early culmination of MAI, rotation age for many tropical forest plantation species are set to coincide with culmination age of MAI, especially for management objectives (e.g., pulpwood) that target maximum volume production. The culmination of MAI has been used to set rotation ages for plantation species such as *Pinus* spp., *Araucaria cunninghamii*, *Eucalyptus* spp., *Tectona grandis*, *Gmelina arborea* and *Nauclea diderrichii* (Sharma 1979; Adegbehin et al. 1988a, b; FORMECU 1999; Onyekwelu et al. 2003b; Evans and Turnbull 2004).

For a number of tropical plantation tree species, the culmination of MAI occurs very early in the life of the stand. For these species, the use of rotation of maximum volume production may not be adequate if very large dimensions logs (e.g., for timber and veneer log) are desired. Consequently, rotation age will have to be extended beyond the age of MAI culmination to the age at which the desired dimension will be obtained. However, the extension of rotation age beyond the culmination age of MAI usually sacrifices volume growth for quality.

27.17 Regeneration of Subsequent Generations

Replanting is the commonest method for regenerating forest plantations in the tropics (FAO 1998). In most cases, plantations are clear-felled and seedlings (or cuttings) of the same or totally new species are reestablished on the same site. However, apart from replanting, forest plantations can be regenerated through other means, among which are regeneration from coppice and natural regeneration through seeds.

27.17.1 Coppice Regeneration and Management

The coppice method of regeneration differs from other methods in that it depends on the shoots re-sprouting from the cut stump of the previous crop. Coppice management is therefore the operation of felling trees and regenerating them through coppices. Coppice management is restricted to tree species that typically sprout vigorously and have sprouts capable of attaining commercial size (Young and Giese 2003). In coppice management, the originally planted trees are felled and the next crop develops from the vigorous shoots (coppice) that sprout from the stumps. To obtain good coppice sprouts, healthy and dominant trees should be used and felling should be conducted during the rainy season and should be low and clean, without tearing of the bark (Venter 1972; Xu et al. 1999; Evans and Turnbull 2004). This is because the quality of the mother tree affects the quality and growth rate of the coppices. The larger the diameter of the coppiced stump, the more the shoots that will grow (McKenna and Woeste 2006). However, there is evidence that shoots developing on old root stocks (after two or three cuttings) exhibit premature aging and earlier culmination of height growth (Evans 1999c). In addition, some stumps die with each coppice cutting and yield declines as site occupancy diminishes (Evans 1999c).

If properly managed, coppices could be used to regenerate tropical forest plantations after harvest at the end of rotation. Coppices could grow faster than planted seedlings. In an experiment with *Eucalyptus urophylla*, Xu et al. (1999) reported that without fertilisation, coppiced trees grew better than replanted trees. Fertilisation did not have a significant effect on the growth rate of coppice trees as against the significant effect of fertilisation of replanted trees. The high growth rate obtained with fertilisation in replanted trees was comparable to that obtained without fertilisation in coppice trees, which suggests that the well-developed root systems of the previous tree helped coppiced trees to take up soil nutrients better than the smaller root systems of the replanted trees. Since several coppices usually develop on a single stump, tending operations is essential, if viable tree products are to be obtained.

Though there is evidence that many tropical forest plantation species have good coppicing ability, coppice management is not popular in the regeneration of tropical forest plantations. Coppice stands are usually managed on short rotations for the production of fuelwood or pulpwood. The use of coppice methods was once popular in Europe but declined in the second half of the twentieth century as oil and gas became cheap (Young and Giese 2003). However, the surge of interest in bioenergy has revived interest in coppice management, especially in Europe, where it is used in the production of high density short rotation energy plantations, thus it is more popular in the regeneration of energy plantation. Coppice management was recommended for second rotation of Eucalyptus plantations unless better genetic material is available for planting. Second rotation of some tropical plantation species (e.g., *Gmelina arborea; Eucalyptus* spp, etc.) has been regenerated through coppice management (Lamb 1968; Suilaman and Lim 1989; Xu et al. 1999). Information on the coppicing ability as well as the growth performances of the coppices of many tropical forest plantation species is lacking, which makes research in this area a necessity.

27.17.2 Natural Regeneration

Profuse regeneration of some important indigenous tree species has been reported in the understory of both monoculture and mixed species plantations (Fahy and Gormally 1998; Wilkie 2002; Lee et al. 2005; Onyekwelu and Fuwape 2008). This regeneration under forest plantation is sometimes comparable to that under natural forest ecosystems. For example, seedlings of *Cola gigantea, Diospyros mespiliformis, Celtis zenkeri, Drypetes* spp*, Hunteria umbellata, Bridellia* spp*, Lophira alata, Ricinodendron heudelotii, etc.* were reported in the understory of

Gmelina arborea monoculture plantations in tropical rainforest ecosystem of Nigeria (Onyekwelu and Fuwape 2008). This is mostly from the seeds buried in the ground prior to forest plantation establishment as well as seeds dispersed from trees in neighbouring natural forests. The implication of the regeneration of indigenous tree species under the canopies of forest plantation is that when current plantations are harvested, the plantation site has the potentials of returning to multi-species ecosystems akin to tropical secondary forests. This kind of regeneration can be adopted if the objective is to gradually revert the plantation site to natural forest site.

Also, many plantation species exhibit profuse natural regeneration (Evans and Turnbull 2004) from the seeds of the species after natural seed fall. *Cordia alliodora* is reported to exhibit profuse natural regeneration throughout Central America (Evans and Turnbull 2004). In Nigeria, naturally regenerated seedlings (also called wildlings) of *Gmelina arborea, Tectona grandis* and *Nauclea diderrichii* are used to augment seedling requirements and to replace planting failures. These seedlings have not been found to be genetically inferior (in terms of growth and yield) to seedlings raised in the nursery. For *Gmelina arborea*, naturally regenerated seedlings are mostly found at the edge of the plantation (Onyekwelu 2001). Although this regeneration system is not commonly used in industrial plantation establishment in the tropics, it has potentials that should be explored. The use of this regeneration method can present a cheaper means of plantation establishment and reestablishment.

27.18 Plantations and People

To meet the growing demands for wood in developing nations, there will have to be intensification of management of industrial plantations, combined with substantial increase in their productivity, and a greater reliance on wood produced by rural populations for their own use and for industry (Turnbull 1999; Mead 2005).

This review focuses on aspects of tree cultivation such as species choice, domestication and tree improvement, site selection and preparation, seed procurement and plant propagation (incl. biotechnology), nutrient management, management of pests and diseases, thinning, etc. However, social dimensions of silviculture, especially with regard to forest plantations, are rarely discussed. Endo (2003) argues that how trees are planted and managed and how a forest plantation project is organised have a long-term effect on people. However, as forest plantations become more widely established, the impacts of forestry programmes on people become substantial and need attention. According to Endo (2003) an important goal may be, “how to make a forestry project more people-friendly while maintaining or improving the project’s competitiveness.”

In developed countries there is growing concern not only on the sustainability of wood production but also on the environmental and social impacts of plantations in a regional and local context (e.g., Sawyer 1993; Kanowski 1997; Turnbull 1999).

For instance, social conflicts with local people have caused some unsuccessful timber plantation developments in Indonesia (Nawir and Santoso 2005) and elsewhere. Charnley (2005) reviewed cases which indicated that Intensively Managed Industrial Roundwood Plantations (IMPIRs) often bring about land ownership concentration, loss of customary rights of resource access, rural displacement, and socioeconomic decline in neighbouring communities. Often IMPIRs do not appear to provide enough quality jobs to stimulate community development, and rarely benefit people who are already politically and economically marginalized.

Successful beneficial partnership is based on commercial feasibility, equitable agreements, appropriate benefit and cost sharing and a shared understanding of co-management and participatory approaches. Essential for the debate are the questions of what forest practices are acceptable and appropriate on a given piece of land in a particular social context, and who decides. Decision makers concerned with plantation establishment and management should take into account their advantages to local people, and give local residents a voice in the decision-making process, which will help to reduce community resistance to industrial plantations and does not contribute to the decline of rural communities and indigenous cultures (Charnley 2005; Nielson and Evans 2009).

27.19 Conclusions

Forest plantations are established for a variety of reasons and vary in composition and structure, as well as in intensity of management. They are relatively simple production systems, usually even-aged monocultures, mostly managed to optimize the yield of wood from a site, protect or reclaim an environment and provide benefits and/or amenities that are important to the community.

It should be possible to grow most of the wood humans need in managed plantations, and hence eliminate the need to log wild forests (Sedjo 1999). Those wild forests are sinks of carbon and sources and resources of biodiversity. By establishment of plantation forests for the purpose of carbon sequestration, nations could be compliant with the Kyoto Protocol (Sedjo 1999). So plantations can offer a viable means of both conserving natural forests and reducing the amount of CO_2 in the atmosphere. The objective of managing plantations in an economic, ecological and socially responsible way has led to the formulation of criteria and indicators as well as guidelines for sustainable plantation forestry (e.g., Muhtaman et al. 2000; FAO 2006). Certification as an instrument for safeguarding was put into practice but sometimes criticized for being ineffective.

Successful plantation forestry will continue to depend on effective research, development and management, and on innovation and technological advances (Kanowski 1997). The appropriate form of technology will vary with social, environmental and economic circumstances. Kanowski (1997) postulates that the sustainability of plantation forestry will be enhanced, and the benefits of investments most fully realised, where plantation purpose and practice are embedded within the broader social and economic contexts: *In realising the considerable potential of plantation forestry to benefit society, one of the principal challenges to plantation forest owners, managers and scientists is to progress from a narrow focus, which Shiva (1993) has characterised as "monocultures of the mind", to a broader appreciation of plantation purpose and practice. We are well placed to do so, by building on the considerable body of experience and information we have gained relevant to plantation and other forms of forestry in many environments. It is in doing so that we shall sustain plantation forestry in the next century, and maximise its benefits.* As Evans (2009a, b) concludes concerning planted forests as a whole: *Planted forests are not a panacea but are a sustainable way of meeting the world's timber requirement from less than 7% of the world's forest area or a mere 2% of land.* Planted forests of the tropics are making a major and increasingly significant contribution.

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Chapter 28
Sustainability of Site Productivity in Tropical Forest Plantations: Case Study of *Gmelina arborea* Plantations in Tropical Rainforest, Nigeria

Jonathan C. Onyekwelu

Abstract The fast growth rate and high productivity of tropical plantation tree species and their high nutrient demand have raised site sustainability concern. The effects of fast-growing *Gmelina arborea* plantations in the tropical rainforest region of Nigeria were investigated. Soil physical, chemical, and biological properties of plantations of different ages as well as between old growth *Gmelina* plantation and adjacent degraded natural forest were comparable, suggesting that plantation development did not have adverse effect on soil nutrients. However, this only applies if the plantations are managed on long rotation, as short rotations would lead to massive site nutrient export. Sustainability of productivity during the next rotation is likely to be determined by factors such as harvesting methods of current stands and management practices during the next rotation.

Keywords Tropical forest plantations · Sustainability of site productivity · *Gmelina arborea* · Growth · Site and tree nutrients accumulation

28.1 Introduction

Most tropical forest plantation tree species are characterized by a fast growth rate and the ability to produce high amount of biomass within a short time. Research works have shown that some of these species (*Acacia mangium*, *Eucalyptus* spp., *Gmelina arborea*, etc.) can maintain average diameter growth of 3–5 cm/year. Tropical forest plantation species can produce much higher biomass per hectare (3–10 times higher) than trees in natural forests (Pandey 1995; Evans and Turnbull 2004). This fast growth rate and high productivity implies high demand on site nutrients, since actual stand productivity is determined by how well trees capture

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site resources. Consequently, concern about the physical, chemical, and biological sustainability of plantation sites has been raised (Trouve et al. 1994; Evans 1998; Kimmins 2004; Onyekwelu et al. 2006).

Site sustainability implies that long-term use of ecosystem is maximized to the intensity where the resource base, structure, or function is not degraded or adversely changed. For forest plantation to be sustainable implies no noticeable depletion in soil physical, chemical, and biological properties. Divergent views exist on the sustainability of forest plantation sites. Keeves (1966 cited in Evans 1999) reported 30% yield decline in second rotation *Pinus radiata* plantations. Repeated loss of nutrients from site during site preparation and in harvested Eucalyptus and Acacia wood adversely affected soil fertility and long-term site productivity (Khanna 1998). Stewart et al. (1985) noted no loss of productivity in Eucalyptus stands. There was improvement in the physical and chemical properties of *Eucalyptus tereticornis* site and stability of site nutrients in *G. arborea* plantations (Mishra et al. 2003; Onyekwelu et al. 2006). Thus, plantation sites are likely to be sustainable and increased productivity in successive rotations is expected provided that good practices are implemented and maintained (Evans 1998; FAO 2001a). Other factors that can influence site sustainability are as follows: species and provenance selection, nursery practices, climatic conditions, rotation age, harvesting practices, etc. (Evans 1998; FAO 2001a; Kimmins 2004). This chapter examines the sustainability of site production in tropical forest plantations using *G. arborea* plantations in tropical rainforest region of Nigeria as a case study.

28.2 Sustainability of *G. arborea* Plantation Site in Nigeria

28.2.1 Site Requirements, Establishment, Growth, and Productivity

G. arborea is among the first exotic tree species in Nigeria (Adegbehin et al. 1988; Onyekwelu 2005). Its optimal site requirements include extremes of temperatures of 18 and 35°C, annual rainfall of 1,778–2,286 mm, distinct dry season in which atmospheric humidity is not below 40% (Lamb 1968), and sandy loam soils. These requirements are met in Nigeria's rainforest region where most *Gmelina* plantations exist. Precipitation in this region exceeds potential evapotranspiration, with no prolonged period of drought, thus plant growth is continuous. The current *Gmelina* plantations are the first rotation on rich soils that once carried humid tropical rain forests.

Planting stocks for the current plantations were raised in the nursery for about 90 days, during which period an average height of 20–30 cm was attained. Initial planting spacing was 2.5 × 3 m, which is necessary to overcome stem form problems associated with the species. Tending operations were carried out during the early stages of the stands. Management objective was the provision of pulpwood at 8–10 years rotation. Thus, thinning was not intended and till date the bulk of the plantations have not been thinned, though management objective has changed to timber provision.

Data for this study were adapted from Onyekwelu (2004) and Onyekwelu et al. (2006). A total of 15 age classes spanning from young (5 years) to old plantations (28 years) were involved in the analysis of growth and productivity, whereas for the analysis of soil properties an age span between 10 and 28 years, and for soil microbial biomass an age span between age 13 and 28 was available. Detailed field data collection procedures as well as data analytical methods have been presented in Onyekwelu (2004) and Onyekwelu et al. (2006).

28.2.2 *Growth Characteristics of* G. arborea *Plantations*

Gmelina plantations are characterized by high growth rate and productivity, which agrees with reports in literature (FAO 2001b; Onyekwelu and Stimm 2002; Onyekwelu 2004; Swamy et al. 2004; Vanclay et al. 2008). Mean DBH and total height varied from 18.4 to 42.4 cm and from 15.2 to 25.3 m between 5 and 28 years (Table 28.1). Mean basal area and volume production were 24.3 m^2/ha and 242.7 m^3/ha for 10 years, and 80.7 m^2/ha and 1,023.4 m^3/ha for 25 years plantations. Mean annual increment (MAI) range is of 30–51.7 m^3/ha/year. Total above ground biomass production increased from 83.2 t/ha (5 years) to 392.1 t/ha (25 years). This high productivity demonstrated by *Gmelina* is characteristic of many tropical plantation species. The MAI of between 30 and 70 m^3/ha/year has

Table 28.1 Growth and productivity of *Gmelina arborea* plantations in south western Nigeria

Age (years)	Trees (per ha)	MHt[a] (m)	DBH (cm)			BA[b] (m^2/ha)	Volume (m^3/ha)	Biomass (t/ha)			
			Mean	Min.	Max.			Stem	Branch	Foliage	Total
5	1,221	15.2	18.4	7.7	31.4	24.3	242.7	70.0	10.6	2.6	83.2
6	1,275	19.0	21.7	8.7	33.0	30.4	370.3	106.1	15.6	3.7	125.4
8	1,264	16.4	19.4	7.5	32.0	33.8	350.7	116.9	18.0	4.2	139.1
10	1,125	18.3	21.9	9.0	42.6	38.8	418.0	135.2	21.0	5.3	161.6
11	1,012	19.2	22.5	6.0	40.5	46.4	467.1	164.0	26.3	6.6	196.9
12	1,000	19.2	23.1	7.1	52.4	50.7	492.0	179.7	29.0	7.7	216.4
14	933	20.4	25.3	6.5	48.5	52.5	572.0	197.9	32.3	8.0	238.3
15	1,184	19.7	26.2	9.0	53.0	58.3	630.7	220.6	35.9	9.2	265.6
16	1,024	21.0	25.4	8.9	50.8	60.5	675.6	231.9	38.1	9.5	279.5
17	992	22.0	28.5	10.2	50.0	65.3	815.9	270.0	43.8	10.2	324.0
19	837	22.4	31.8	12.0	52.8	67.8	859.5	267.2	44.2	11.0	322.4
21	800	24.8	34.3	12.6	54.0	72.9	899.2	302.2	49.8	12.7	364.8
23	742	23.5	34.5	10.8	65.5	77.9	963.6	308.7	54.0	13.5	376.1
25	683	24.2	36.6	15.5	67.8	80.7	1,023.4	321.5	56.5	14.1	392.1
28	492	25.3	42.4	16.1	75.2	74.7	973.3	301.9	53.4	13.5	368.8

[a] *MHt* Mean total height
[b] Basal area

been reported for species such as *Eucalyptus* spp., *Acacia* spp., *Leuceana leucocephala*, *Paraserianthes falcataria*, etc. (Hopmans et al. 1990; Mok et al. 1999).

Stem, branch, and foliage nutrient accumulation followed the order: Mg > N > Ca > P > K > Na. Mg accumulation ranged from 201.4 to 461.1 kg/ha (82.9%), 31.8 to 77.1 kg/ha (13.5%), and 8.6 to 20.0 (3.6%) kg/ha in the stem, branches, and foliage, respectively, while N in the respective components is 192.3–384.8 kg/ha (82.3%), 28.7–64.1 kg/ha (13.1%), and 9.8–22.7 kg/ha (4.6%). Branch and foliage accumulation of total P was 6.3–14.4 kg/ha (15.1%) and 1.8–4.4 kg/ha (4.8%), while it was 34.7–71.9 kg/ha (80.1%) for stem. Branch and foliage accumulation of total K was 15.8% (2.9–8.4 kg/ha) and 5.9% (1.0–2.9 kg/ha) while stem accounted for 78.3% (14.6–36.8 kg/ha). The share of stem, branch, and foliage of total Ca was 83.8, 12.4, and 3.8%, respectively.

28.2.3 Effect of Plantation Development on Site Physical, Chemical, and Biological Factors

The sand (63.9–71.7%) and clay (19.3–25) contents of *Gmelina* plantation sites in Nigeria indicate that texture is sandy loam to sandy clay loam, especially to 30 cm depth, beyond which texture tended toward sandy clay. Bulk density ranged from 1.41 to 1.59 mg/m^3 and was not significantly affected by stand development. The soils are neutral to slightly acidic, with a pH range of 7.2–5.7, which was not significantly affected by plantation development (Onyekwelu et al. 2006).

Soil chemical nutrients and organic matter had similar developmental trends (Fig. 28.1). Soil chemical nutrients of young and middle-age stands were either equal or slightly lower than that of old plantations (Fig. 28.1), except exchangeable potassium. However, these differences were not significant, indicating that *Gmelina* plantation development did not adversely affect soil chemical nutrient concentrations. The effect of *Gmelina* development on soil biological property (microbial population and biomass) was also not statistically significant (Fig. 28.2).

28.2.4 Indication of G. arborea Site Sustainability

Preplanting soil data for *Gmelina* sites in Nigeria does not exist. Consequently, degraded natural forest soil data were used as baseline data to evaluate the effect of *Gmelina* development on site nutrients. This was based on the assumption that if *Gmelina* had not been planted, the conditions of its current site will probably be the same as that of adjacent degraded forest, since they were planted by clearing these forests (Onyekwelu et al. 2006). Twenty-five-year-old *Gmelina* stand was used, since it is the recommended rotation age for its timber. Soil nutrient concentrations of adjacent degraded forest and 25-year-old *Gmelina* plantation are comparable

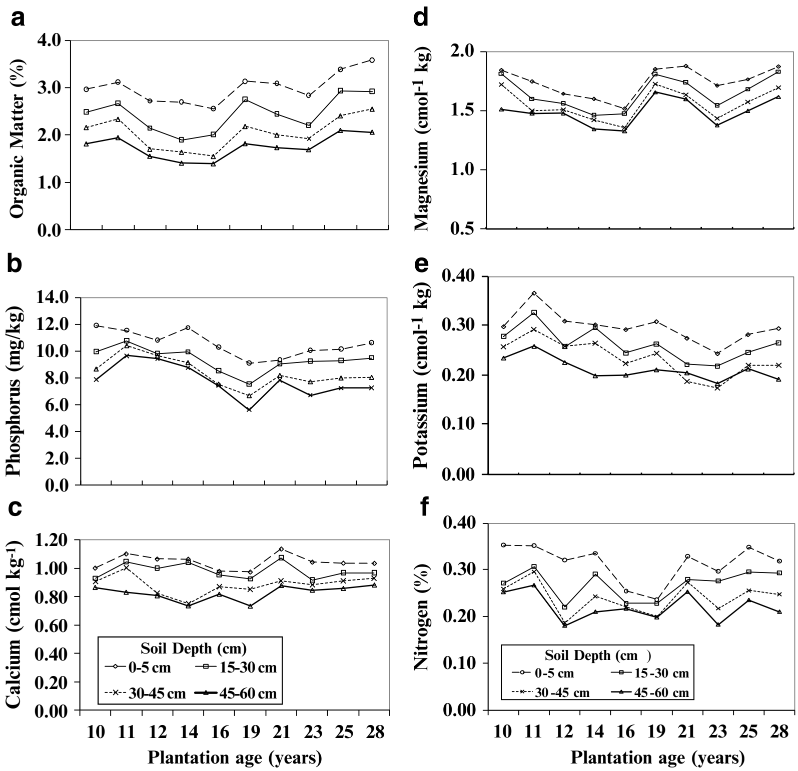

Fig. 28.1 Chemical properties of *Gmelina arborea* plantation soils in tropical rainforest region of Nigeria: (**a**) Organic matter, (**b**) Phosphorus, (**c**) Calcium, (**d**) Magnesium, (**e**) Potassium, (**f**) Nitrogen

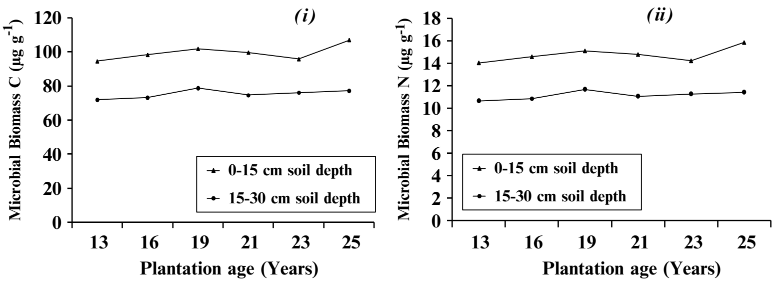

Fig. 28.2 Soil microbial biomass (i) Carbon and (ii) Nitrogen of age series *Gmelina arborea* plantations in tropical rainforest region of Nigeria

(Table 28.2). However, the nutrient concentrations and organic matter of young and middle-age sites were slightly lower than that of old plantations and by extension, degraded natural forest. Thus, it appears that the nutrients were slightly depleted in

Table 28.2 Comparison of soil physical, chemical, and biological properties of *Gmelina arborea* plantation and adjacent degraded natural forest in tropical rainforest region of Nigeria

Soil physical, chemical and biological properties	Soil depth			
	Gmelina plantation (25 years)		Degraded forest	
	0–15 cm	15–30 cm	0–15 cm	15–30 cm
Sand content (%)	66.8	61.1	70.6	68.5
Clay content (%)	23.0	26.0	21.9	22.9
Silt content (%)	10.2	12.9	7.5	8.6
Bulk density (mg/m^3)	1.42	–	1.36	–
pH	7.1	6.8	6.7	6.5
Total N (%)	0.41	0.33	0.36	0.32
Available P (mg/kg)	9.57	8.61	10.77	10.23
CEC (cmol/kg)	10.25	10.29	11.99	5.34
Exchangeable K (cmol/kg)	0.28	0.27	0.35	0.30
Exchangeable Mg (cmol/kg)	1.77	1.68	1.91	1.66
Exchangeable Ca (cmol/kg)	1.03	0.97	1.05	0.94
Exchangeable Na (cmol/kg)	0.21	0.25	0.25	0.27
Organic matter (%)	3.38	2.93	3.42	2.80
Microbial population (cfu/ml $\times$ 10^4)	43.51	20.0	37.42	14.47
Microbial biomass C (μg/g soil)	87.50	62.59	84.11	11.88
Microbial biomass N (μg/g soil)	12.98	9.29	12.42	9.35

young and middle-age stands and built up in old ones. This trend can be attributed to the high amount of nutrient uptake during the early stage of development to support the rapid growth rate, which slows down as the plantations advances in age. The period of slight depletion in soil nutrient concentrations coincides with the period of active growth (i.e., higher MAI) while the period of nutrient build-up coincides with that of growth recession in *Gmelina* plantations in study sites (Onyekwelu et al. 2003).

Foliage and branches are the main repository of aboveground nutrients in *Gmelina* trees in the study area (Onyekwelu et al. 2006). Since the foliage of the species are shed every year (deciduous), it implies that their high nutrient content are returned to the site every year, given the fast decomposition rate of *Gmelina* foliage (Nwoboshi 2000). Result shows that between 9.8 and 22.4 kg (N), 1.8–4.4 kg (P), 1.0–2.9 kg (K), and 8.6–20.0 kg (Mg) per hectare of foliage nutrients will be available for recycling every year, which will translate to average accumulation of 402.5 kg (N), 77.7 kg (P), 48.75 kg (K), and 375.5 kg (Mg) per hectare at the end of 25 years growth. In addition, the good amount of the branches shed every year will play a critical role in nutrient cycling.

Nutrient accumulation and export from forest plantation sites is an important consideration for long-term site quality and sustainability of production in short rotation, high-yielding plantation ecosystems. Although some researchers hold that the fast growth rate of the species depletes the nutrient base of the site and thus adversely affects long-term site sustainability, others opine that the decrease in productivity in successive rotations, where it exists, is due to inappropriate management practices such as topsoil and litter repositioning, burning of debris, harvesting method and management of harvest residues, etc. (Will 1992 cited in Evans 1999; Khanna 1998).

The nonsignificant difference of the nutrients of degraded forest site and that of various plantation sites implies that plantation development does not have adverse effect on site nutrients. It also implies that the nutrient status of the plantation sites has not been depleted to the extent that decrease in productivity during the next rotation would be anticipated, which agrees with report in literature. There was progressive increase in organic matter under Eucalyptus plantations in Congo DR and a significant improvement in soil nutrients under *Gmelina* plantations in Nigeria and India (Chijioke 1980; Trouve et al. 1994; Swamy et al. 2004). It has been shown that *Gmelina* does not exhaust its site nutrients in a single rotation. Of the total site nutrient stock of 2,771, 412, 5,782, and 2,124 kg/ha of N, P, K and Ca, respectively, average nutrient requirement for *Gmelina* in one rotation is 960, 371, 2,425, and 615 kg/ha of N, P, K and Ca, respectively (Nwoboshi 2000). However, the sustainability of site productivity in *Gmelina* plantations is only expected if the stands are managed on long rotation. Shorter rotations would lead to massive nutrient export from the site before replenishment and consequently depletion of site nutrients in subsequent rotations.

Since current *Gmelina* plantations in Nigeria do not have adverse effect on site nutrients, a decrease in productivity during the next rotation is not anticipated. Consequently, sustainability of site productivity in the next rotation is more likely to be determined by harvesting methods of current stands and management practices of the next rotation. If well managed, increase in productivity might result as was reported for second rotation stands of some species (Long 1997; Evans 1999). Kimmins (2004) demonstrated that if current stands are not harvested by whole tree method and if successive stands are managed on long rotations, site nutrients in successive rotations are likely to be maintained at the original level. In addition, management of soil organic matter is important as it contains the bulk of the nutrients. Maintaining the current organic matter status and retaining harvest residues on-site following harvest would play a critical role in maintaining long-term soil fertility and productivity in subsequent rotations.

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Chapter 29
Eucalyptus grandis and Other Important *Eucalyptus* Species: A Case Study from Sri Lanka

Thavananthan Sivananthawerl and Ralph Mitlöhner

Abstract Eucalypts are the most successful plantation species in the higher elevations of Sri Lanka. They were introduced to Sri Lanka in the latter part of the nineteenth century. The first plantation in Sri Lanka was established in 1890 at an altitude of 1,200–1,800 m. The major species planted in the up-country are *Eucalyptus grandis*, *Eucalyptus microcorys*, *Eucalyptus robusta*, *Eucalyptus pilularis* and *Eucalyptus torelliana*. *E. grandis* is more tolerant than *E. microcorys* to soils of low fertility. Also, compared with other species, *E. grandis* has a relatively fast growth rate, especially on marginal lands. The reference age is considered to be 30 years, at which clear felling usually starts. The diameter and the height growth in some areas are extremely high and the recorded total volume production and the maximum mean annual increment are also well above the standard scale. Therefore, compared to other *Eucalyptus* species, *E. grandis* is the most suitable species for rotation in the low-fertility and very acidic areas of the up-country of Sri Lanka with some additional silvicultural practices.

Keywords *Eucalyptus grandis* · Sri Lanka · Growth · Management

29.1 Background

Eucalyptus could be the most successfully and widely spread genus as a plantation crop in the tropics and subtropics. This is mainly due to its wide range of adaptability to both climate and soil. No tree genus has ever been so widely distributed over

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such wide geographic and environmental gradients throughout the world as the evergreen, emergent *Eucalyptus* (Zacharin 1978).

Eucalypt plantations are often established to provide firewood, charcoal, poles, pulp, shade and sometimes honey, eucalyptus oil and recycling nutrients. Also, the provision of wood from eucalypt plantations on previously cleared land contributes to some extent to the protection of remaining natural forests from exploitation for wood production (Davidson 1985). The increase of plantation areas occurred mainly in tropical Asia and the Pacific region at a rate of 2.11 million hectares per year (81% of the global increase in forest plantations) through species of *Eucalyptus* (Ball 1996).

Some researchers have very frequently put forward criticisms of and justifications for the establishment and management of *Eucalyptus* plantations, often without adequate reference to the type of plantation being referred to or its objectives. One criticism is that species in this genus decrease the soil pH. This may be true in situations where they have been planted on a site with a low buffering capacity. But this is not true for sites which have an alkaline soil with a high buffering capacity, or if the soil is already very acidic (Sawyer 1993).

Another argument is that *Eucalyptus* reduces the groundwater yield. In general, all tree species reduce water yield compared with shrubs and grasses. In one study, depths of up to 10 m in transects perpendicular to 4–7-year-old eucalypt belts (*Eucalyptus horistes*, *E. kochii* ssp. *plenissima*, *E. loxophleba* ssp. *lissophloia*, *E. polybractea*) indicated that the eucalypt species can exploit soil water to depths of at least 8–10 m within 7 years of planting (Robinson et al. 2006).

In a laboratory study from India with its native species, it was shown that *Eucalyptus* consumed 0.48 L of water to produce 1 g of wood, compared with 0.55, 0.77 and 0.50 L g^{-1} for *Albizia julibrissin*, *Dalbergia sissoo*, and *Syzygium jambolanum*, respectively. In another study it was claimed that on an 8-year rotation, the mean annual growth of *Eucalyptus* per hectare was as much as 40 m^3, whereas for indigenous trees, the average is 0.50 m^3. An overall high productivity therefore necessitates a greater overall water demand (Prabhakar 1998). However, the effect of groundwater reduction by eucalypts is less than that by pine and is greater than that by the other broad-leaved species (Poore and Fries 1985). In a study by Kanowski and Savill (1990), it was found that plantations of *E. tereticornis* and *E. camaldulensis* use no more water than degraded indigenous forest on adjacent sites.

However, to combat wood shortage in the future, one should accept some ecological effects on the environment as a compromise. A fast-growing tree needs more water than slow growers. This water deficit will become worse in drier areas, where evapotranspiration is much higher than precipitation. Greenwood et al. (1985, cited in Sargent 1998) measured and compared transpiration from two species of *Eucalyptus* and from grassland, and the annual average transpiration rates were found to be 2,700 and 390 mm, respectively. Most of these adverse effects can be minimized through proper plantation management.

29.2 Growth Habitat

E. grandis is the most widely grown species and is promoted by both the public and the private sector for its performance and suitability in the up-country of Sri Lanka. It is a light-demanding, shade-intolerant species which responds well to heavy thinning (Jacobs 1981). It has a white trunk bole which can grow up to 45–65 m in height and the bole is straight and clean up to two thirds of the total height (Luna 1996). It has an open-crown structure and the branches are mostly smooth white or grey–white in colour. Young leaves are stalked, opposite for several pairs and then alternating, ovate and green to dark green. Matured leaves are alternate, lanceolate or broad lanceolate, glossy dark green on the upper side and pale green underneath (Arnold 1998).

E. grandis in its natural habitat (Australia) occurs on flats or lower slopes of deep, fertile valleys, commonly on the fringes of, or sometimes within, rainforest stands (Boland et al. 1984). It can also occupy upper slopes and ridge-top sites (Eldridge et al. 1993) and can tolerate poor skeletal soils under adequate rainfall conditions. Even though *E. grandis* prefers moist, well-drained, deep, loamy soils of alluvial or volcanic origin (Streets 1962), it can adapt to conditions where the moisture content is high or to soils with poor drainage or short periods of water logging (Turnbull and Pryor 1978). Furthermore, Clemens et al. (1978) proved that seedlings of *E. grandis* were more resistant to water logging than those of *E. robusta* and *E. saligna*. *E. grandis* can tolerate periods of drought but, in general, it is not suitable for sites on dry, stony, skeletal soils or those that have relatively little soil depth (Arnold 1998).

E. grandis is preferred for its ease of nursery management, rapid early growth, good form and wood properties suitable for many uses (Hillis and Brown 1978). It may be the most widely planted eucalypt for industrial wood production, with an estimated plantation area of about two million hectares in 1987. The largest areas of plantations of *E. saligna* and its hybrids with other species are in Brazil (over one million hectares) and South Africa (300,000 ha) (Eldridge et al. 1993). It is also planted extensively in Argentina, Australia, India, Zambia, Zimbabwe, Tanzania, Uganda and Sri Lanka.

29.3 *Eucalyptus* in Sri Lanka

Eucalypts were introduced to Sri Lanka from Australia in the latter part of the nineteenth century. At the end of 1996, they covered about 31,300 ha of forestland (Bandaratillake 1997) and constituted a natural resource of high economic value. The first plantation in Sri Lanka was established in 1890 at an altitude of 1,200–1,800 m, where the annual rainfall ranges from 1,850 to 2,500 mm (FAO 1955). Eucalypts belong to the first of three exotic species introduced early in the nineteenth century to be raised as a forest plantation. In 1888, *E. saligna* was first planted in the montane zone (Nanayakkara 1996). However, there was confusion in differentiating between *E. saligna* and *E. grandis* until 1918. The other two species were teak (*Tectona grandis*) and mahogany (*Swietenia macrophylla*), which were introduced somewhat earlier than eucalypts. *Eucalyptus* species were originally established in the hilly areas to provide fuelwood for households and the tea industry. But later it was realized that most of these species were good for making railway sleepers and industrial timber (Bandaratillake 1996). According to past experience, these two species showed the best outcome than the other *Eucalyptus* species such as *E. cloeziana* and *E. pilularis* (Bandara *Eucalyptus grandis* seed orchard and *Eucalyptus microcorys* provenance trials and future genetic improvement strategies in the up-country of Sri Lanka: a report to the Australian Centre for International Agriculture Research as part of the "Seeds of Australian Trees" Project unpublished). Also, it was found that, compared with other species, *E. grandis* has relatively the fastest growth rate, especially on marginal lands.

Forest plantations, on a management scale, were begun in the twentieth century on abandoned so-called chenas (abandoned lands after shifting cultivation) and other unproductive lands. Afforestation of eucalypts was started in the 1930s in the "patana grasslands" in the up-country. *E. grandis*, *E. microcorys* and *E. robusta* were planted in compact blocks on the crests of ridges and hilltops due to windbreaks in some places within the Uva basin. This programme produced a positive result by protecting the villagers from the desiccating winds frequently reaching the area and improving their living conditions. After gaining experience from this programme, in 1954 the Forest Department extended its activity to other places in the Uva basin by planting windbreaks, mainly with eucalypts, which produced remarkable results within 10 years. In the 1950s *E. camaldulensis* and *E. tereticornis* were introduced to the lowlands of the wet and dry zones.

Before 1950 teak was the major species in reforestation programmes in the dry zone established under the so-called Taungya System. But because of teak's unsatisfactory performance, planting teak was abandoned on a large scale in 1970. Thereafter, *E. camaldulensis* and *E. tereticornis* took over its position in the dry zone and they have been very successful in this area. Eucalypts were planted in Sri Lanka mainly for production of fuelwood, sawn timber, railway sleepers, transmission poles and for the supply of essential oils, as soil stabilizers and shelterbelts. Beekeeping was another common practice in some areas of eucalypt plantations.

Presently, the major species planted in the up-country are *E. grandis*, *E. microcorys*, *E. robusta*, *E. pilularis* and *E. torelliana*. *E. citriodora*, *E. globulus* and *E. paniculata* are planted on a small scale. *E. grandis* and its hybrid with *E. robusta* grow very well in the up-country. *E. grandis* is more tolerant than *E. microcorys* to soils of low fertility. *E. robusta* grows very well and produces firewood and transmission poles with a rotation of 15–20 years and sawn wood with a rotation between 35 and 50 years. Thinning is practised at the age of 15 years with the removal of 15–30% of the basal area. It has a good form, makes a dense canopy and coppices very well (FAO 1979).

29.4 Present Management Regime

Techniques of plantation establishment in the older plantations of Sri Lanka are not well documented. The natural forest was exploited by the railway and neighbouring tea estates for fuelwood. Estates were allowed to exploit the natural forest, on the condition that they replanted the areas which they had cleared. However, this requirement was stopped with the realization of the importance of natural forest a few decades ago (in the 1970s). A wide variety of exotics were originally planted. But many of those failed and the lands were replanted principally with *Eucalyptus* spp. In some compartments (now termed "subblock") they were planted in strips in the natural forests, and later the intervening areas were also felled and planted with exotics. These practises have led to plantations mixed with different species and age classes. Some plantations have reverted to scrub and small indigenous trees, with a few scattered weakly regenerating stools of *E. robusta*. This was the result of poor establishment or an increasing mortality of stools over a large number of coppice rotations, or both (Forest Department 1994).

Since 1980, planting activities have concentrated on reestablishment through replanting of clear felled areas. In most cases such sites are ideal for the establishment of *Eucalyptus* species, being weed-free, fertile and with a favourable moisture regime (Forest Department 1994). Unfortunately, in many cases full advantage of these conditions has not been taken, mainly owing to poor-quality seedlings; not weeding sufficiently and at the correct time; too close spacing, resulting in plantations going into check within 2 years of planting; old stools from the previous crop not being bark-stripped, with the result that coppice growth repressed the new seedling crop; no fertilizer being applied; and species being mixed without any silvicultural or management rationale.

29.5 Distribution

The *Eucalyptus* plantations are distributed all over the island (Fig. 29.1). These distributions are based on the major division of the agroecological zones of Sri Lanka. Sri Lanka is divided into three major agroecological zones based on the average annual rainfall, namely the dry zone, the intermediate zone and the wet zone, with average annual rainfall of less than 1,250 mm, 1,250–1,900 mm and more than 1,900 mm, respectively.

The eucalypt species distribution depends on the climatic demarcations. *E. grandis* and *E. microcorys* are established in the wet zone, whereas *E. camaldulensis* and *E. tereticornis* are mainly distributed in the intermediate zone and the dry zone. The total extent of *Eucalyptus* species in Sri Lanka is around 25,000 ha (Table 29.1).

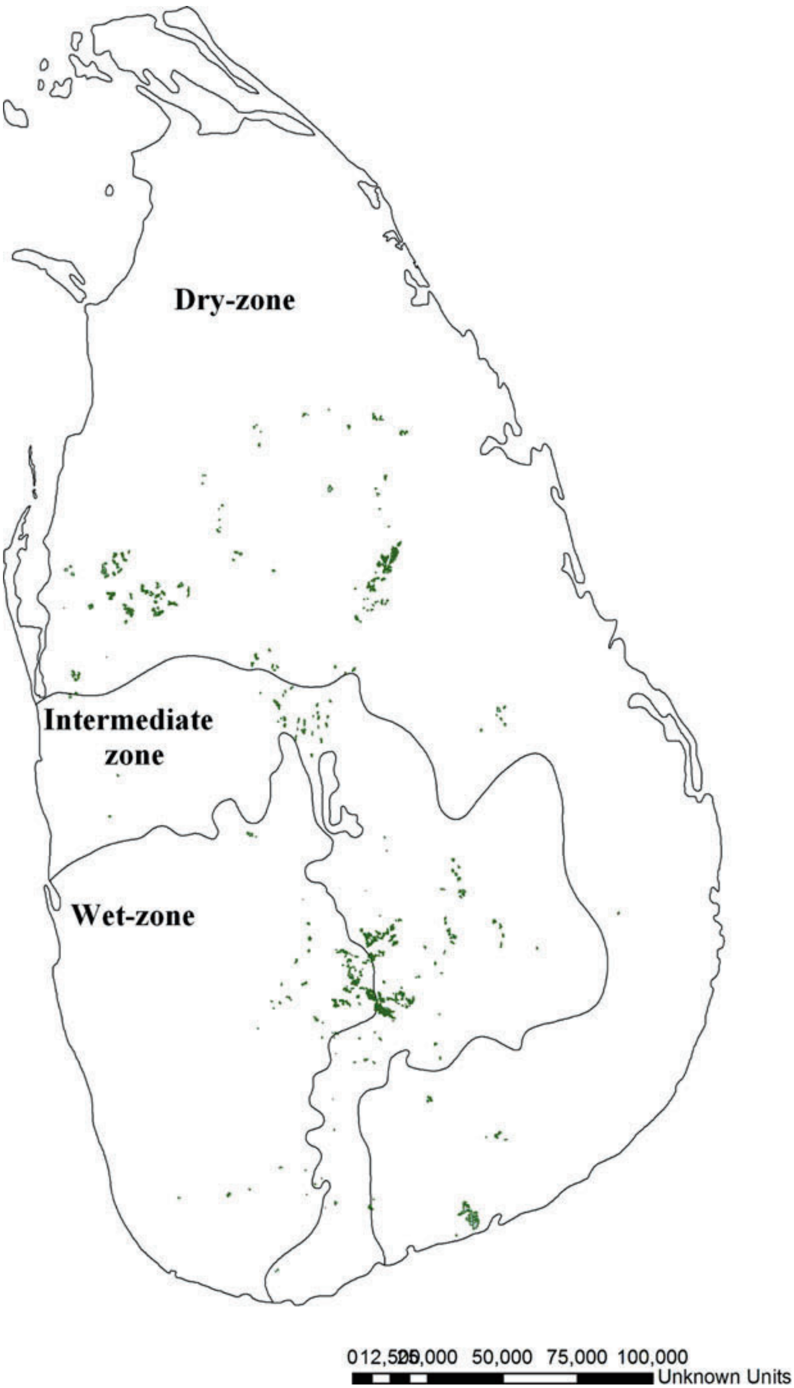

Fig. 29.1 Distribution of *Eucalyptus* plantations in Sri Lanka (Forest Department 2009)

Table 29.1 Extent of different types of *Eucalyptus* species in Sri Lanka

Species	Extent (ha)
E. deglupta	1
E. pularis	17
E. citridora	40
E. globulus	159
E. torelliana	159
Mixed	610
E. robusta	986
E. microcorys	1,072
E. tereticornis	1,240
E. grandis	6,583
E. camaldulensis	14,148

Source: Forest Department

29.6 Rotation Age

In addition to thinning and weeding, creeper cutting and cleaning are other management operations in the eucalypt plantations in Sri Lanka. Usually these operations are carried out at 1–2-year intervals as and when they are required, i.e. in a low-rainfall area the interval would be greater than in a high-rainfall area. The rotation age of eucalypt species depends on the management objectives. Since the demand for eucalypts for pulp is very limited in Sri Lanka, pulp wood production is not a major objective in eucalypt planting. *E. camaldulensis* and *E. tereticornis* are managed for small timber and fuelwood with a rotation age of 7 years. The regeneration of the second and third crops is through coppice, after which artificial regeneration is adopted to raise the new crop. All the other eucalypt species planted are managed primarily for production of transmission poles and sawn timber, including railway sleepers. Fuelwood is considered as a secondary product from these plantations. The rotation age of the plantations for fuelwood and pulp wood is 7 years, for transmission poles it is 15 years and for sawn timber it is 25 years (Bandaratillake 1996). The reference age was considered to be 30 years, at which age clear felling usually starts. The maximum observed top height for *E. grandis* was 63 m at the reference age of 30 years (Sivananthawerl and Mitlöhner 2002).

However, in practise, plantations are felled much later than the rotation age mentioned above. This is mainly because the clear felling operations have to work their way through the backlog of older (more than 30 years) plantations (Forest Department 1994). Owing to this delay in felling, the following consequences occur in the plantations:

- Loss of timber production, and thus revenue, because older, slow-growing stands are not being replaced and replanted in time by young, fast-growing plantations.
- Shake and splitting, resulting in a reduction of timber quality.

29.7 Success of *Eucalyptus grandis* in Sri Lanka

In Sri Lanka, *E. grandis* is distributed mainly in the Nuwara-Eliya and Badulla divisions. The annual precipitation varies among the beats (smallest land category for the management purpose) within the divisions. In the Nuwara-Eliya division, the annual rainfall ranges from 2,111 mm in Kandapola to 3,364 mm in Bambarakelle and from 1,000 to 1,375 mm in Badulla. A maximum of 3 months with rainfall of less than 75 mm has been experienced in several beats in Nuwara-Eliya, and Badulla has a maximum of 6 months with rainfall less than 75 mm per month. The plantations in the Nuwara-Eliya division are located at elevations from 950 to 2,250 m and in Badulla they are located at elevations from 500 to 1,500 m. The mean monthly maximum temperature for Nuwara-Eliya varies between 18° and 22°C and the mean monthly minimum temperature ranges from 6° to 12°C. The Badulla division has a mean annual temperature of 27°C and the mean daily range is 6°C (Forest Department 1994, 1995).

When the climatic conditions of *E. grandis*, grown naturally in Australia and planted as an exotic species in Sri Lanka, are compared, most of the climatic parameters such as rainfall, elevation and the mean temperature ranges overlap each other well. On the other hand, *E. grandis* grows on a range of soil types and can be grown in moderately fertile soils. It can also be accommodated on the upper slopes and ridge-top sites as a watershed (Eldridge et al. 1993). This shows the suitability of the crop for the hilly areas where most of the abandoned marginal lands formally used for tea production are found.

The maximum observed basal area was 74.6 m^2 ha^{-1} with 138 stems per hectare. Volume estimation plays a major role in the productive forest. The minimum volume recorded is 7.15 m^3 ha^{-1} at the age of 3 years (1,225 trees per hectare) and the maximum is 980 m^3 ha^{-1} at the age of 40 years (250 trees per hectare). The mean annual increment (MAI) is an important factor, and contributes to the thinning practises in a plantation. The observed maximum MAIs in beats Bopaththalawa, Kandapola and Dixon Corner were 36, 29 and 24 m^3 ha^{-1}, respectively. Beats Bopaththalawa and Dixon Corner have maximum MAIs at 18 and 20 years of age, whereas beat Kandapola has the maximum MAI at the age of 30 years. These figures differ from those for *E. grandis* plantations in South Africa and India (Kerala). For example, in Brazil, under good growth conditions, the MAI of eucalypts is 18–20 m^3 ha^{-1} $year^{-1}$ and the MAI of teak in India is only about 2.5 m^3 ha^{-1} $year^{-1}$ (FAO 1995). Compared with a poor site, a good site brings a stand to its maximum MAI at an earlier stage (beats Bopaththalawa, Kandapola and Dixon Corner are within the Nuwara-Eliya division).

The growth of *E. grandis* shows very good performance in the up-country of Sri Lanka. The diameter and the height growth in some beats are extremely high and comparable to those of plantations in other countries. The recorded total volume production and the maximum MAI are also well above the standard scale. Therefore, compared with other *Eucalyptus* species, *E. grandis* is the most suitable species for rotation in the low fertility and very acidic areas of the up-country of Sri Lanka with some additional silvicultural practises.

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Part IX
Planted Forests: Silvicultural Aspects for Restoration and Reforestation

Chapter 30
Review
Plantations for Protective Purposes and Rehabilitation

Michael Weber, Bernd Stimm, and Reinhard Mosandl

Abstract Planting trees for protective purposes is an increasingly important aspect in tropical forestry. Protective forest plantations are not only key elements in the protection of water resources, of soils from erosion, etc., but also they are elements in the mitigation of carbon emissions and minimizing loss of forest biodiversity. The area of tropical forest plantations for protection is 30.1 million ha (2005), with about two-thirds of the area in Asia. This chapter reviews various aspects of protective forest plantations in the tropics, including rehabilitation and restoration.

Keywords Global extent · Native species · Nonindustrial plantations · Protective plantations

30.1 Introduction

Land degradation and desertification are becoming an increasing challenge in wide areas of the tropics. The ongoing loss of natural vegetation cover, mostly forested land, predominantly caused by agricultural expansion, exploitation, and the growing extent of human settlements is one reason for the extreme reduction of indispensable forest functions such as soil and water conservation, slowing runoff from watersheds, and buffering climates.

In plantations for protective purposes, timber production is only of secondary importance, and the protective role of trees becomes the dominant consideration in all decisions (Evans and Turnbull 2004). On many degraded and inhospitable lands, protective plantations represent the only option for rehabilitation. Evans and Turnbull (2004) mention the example of China as one of the leading countries in establishing protective plantations, whether by shelterbelts or watershed protection.

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Combating desertification and dune fixation is the imperative motivation for plantations in many places in Africa, for example, countries bordering the Sahara or in Namibia (STCP 2009).

Unfortunately, there are no clear figures about the quantity and quality of protective forest plantations in the tropics. In forest statistics, for example, from FAO, these plantations are traditionally subsumed under "nonindustrial" plantations together with plantations grown for fuelwood, charcoal, domestic consumption, and nonwood products (FAO 2001). However, recently FAO embarked on a Global Thematic Study of Planted Forests (Del Lungo et al. 2006), which distinguished between two types of planted forests: productive and protective. Protective plantations are defined as "forest of native or introduced species, established through planting or seeding mainly for provision of services." In this context, the chapter is dealing with plantations for protective and rehabilitative purposes.

Although most protective plantations are established to protect, e.g. for conservation of soil and water (STCP 2009), others are established to reclaim, rehabilitate, or restore an environment (see Box 1). Lamb (1998) reviewed approaches to redesign timber plantations for the purpose of restoration of biodiversity. The degree of ecological restoration that can be achieved by using the proposed alternatives ranges from modest to significant, although none is likely to achieve complete restoration.

Protective plantations generally provide benefits and/or amenities that are important to the community (Kanowski 1997; Evans 1998; Carle et al. 2002). Funds for such plantations are most often derived from government programs and particularly from international cooperation. The future expansion of forest plantations for protection will be highly linked to the availability of financing from these sources (STCP 2009).

Box 1: Definitions

Restoration: Reestablishing the structure, productivity, and species diversity of the forest originally present. In time, ecological processes and functions will match with those of the original forest. The Society for Ecological Restoration defines it as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Lamb and Gilmour 2003).

Rehabilitation: Reestablishing the productivity and some, but not necessarily all, of the plant and animal species originally present. For ecological or economic reasons, the new forest may include species not originally present. In time, the original forest's protective function and ecological services may be reestablished (Lamb and Gilmour 2003).

Reclamation: Recovery of productivity at a degraded site using mostly exotic tree species. Species monocultures are often used. Original biodiversity is not recovered but protective function and many of the original ecological services may be reestablished (Lamb and Gilmour 2003).

(continued)

Restoration ecology: The science of restoring ecosystems (Sarr et al. 2004).

Ecological restoration: A practice that aspires to use ecological principles but does not fundamentally aim to expand our knowledge of ecosystems (Sarr et al. 2004).

30.2 Extent of Tropical Forest Plantations for Protective Purposes

In 2005, the global area of protective plantations was estimated as 30.1 million ha (0.8% of the total forest area), with about two-thirds of the area in Asia (see Fig. 30.1, FAO 2006). As shown in Table 30.1, the global area of protective forest plantations increased by 380,000 ha year^{-1} between 1990 and 2005, with varying rates in the different regions. Regarding the absolute change of area, Asia is still the region with the largest changes (187,000 ha year^{-1}) (FAO 2006).

For protective forest plantations in tropical and subtropical regions, no recent information is available. The area of "nonindustrial" forest plantations in the tropics and subtropics, which can be considered as an approximation to protective plantations, was estimated as 26.3 million ha for 1995 (Table 30.2), which represents about 37% of total tropical forest plantation area at that time.

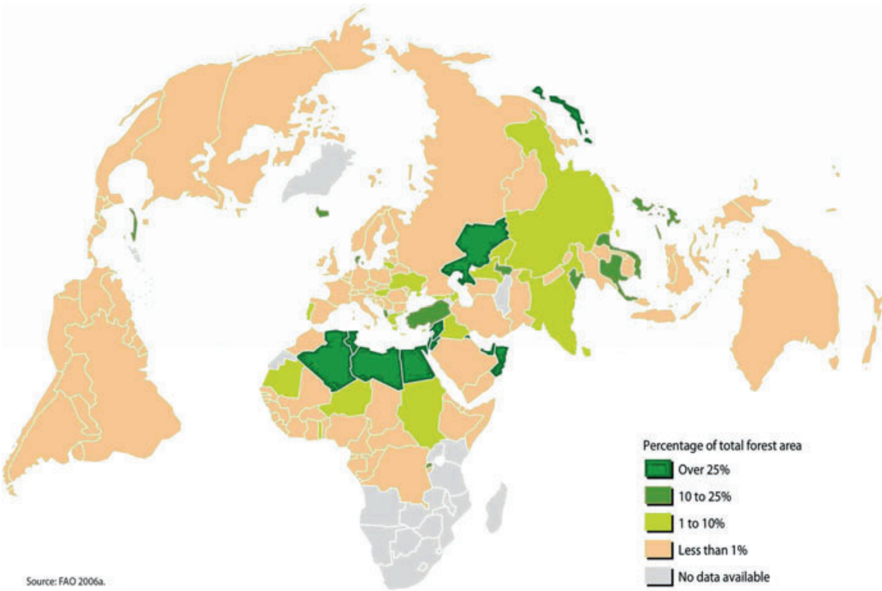

Fig. 30.1 Global forest plantations for protective purposes – percentage of total forest area [source: Vital Forest Graphics copyright 2009, United Nations Environment Program (UNEP), Food and Agriculture Organization of the United Nations (FAO) and the United Nations Forum on Forestry]
Source: UNEP, FAO, UNFF (2009)

Tropical and subtropical Asia had by far the largest area of nonindustrial plantations with 21.2 million ha (~80% of the total nonindustrial plantation area).

Table 30.3 gives an overview on the top ten countries with the largest area of protective plantations. Seven of them, i.e., India, Thailand, Mexico, Vietnam, Algeria, Sudan, and the southern regions of China, are part of the subtropics and tropics.

Table 30.1 Area and annual changes of protective plantations worldwide (FAO 2006)

	Area (1,000 ha)	Percentage of forest area	Annual change 1990–2005 (1,000 ha)	Annual change rate 1990–2005 (%)
Africa	2,370	0.55	26	1.22
Asia	20,474	3.58	187	0.99
Europe	6,027	0.61	97	1.86
North and Central America	1,190	0.17	67	13.14
Oceania	32	0.02	1.4	28.34
South America	31	n.s.	1	7.48
Global	30,125	0.82	380	1.41

Table 30.2 Forest plantation areas in tropical and subtropical countries in 1995 (FAO 2002)

Region (no. of countries)	Area (1,000 ha)		
	Industrial plantations	Nonindustrial plantations	Total plantations
Africa			
Tropical (37)	1,315	2,070	3,385
Subtropical (6)	2,482	945	3,427
Subtotal (43)	3,797	3,015	6,812
Latin America			
Tropical (25)	5,373	1,666	7,038
Subtropical (3)	2,458	467	2,925
Subtotal (28)	7,831	2,133	9,963
Asia and Pacific			
Tropical (16)	14,269	17,361	31,630
Subtropical (2)	18,586	3,855	22,440
Subtotal (18)	32,855	21,216	54,071
Total tropical (78)	20,957	21,097	42,054
Total subtropical (11)	23,526	5,267	28,793
Grand total (89)	44,483	26,363	70,846

Table 30.3 Top ten countries with largest area of protective forest plantations 2005 (FAO 2006)

Countries	Area of protective forest plantations (1,000 ha)	Annual change rate 2000–2005 (%)
Japan	10,321	n.s.
Russian Federation	5,075	1.8
China	2,839	5.6
India	2,173	2.8
Thailand	1,102	0.4
Mexico	986	−1.2
Kazakhstan	909	−3.0
Vietnam	903	6.3
Algeria	742	2.9
Sudan	675	−0.8
Total top ten	25,725	1.2

30.3 Effectiveness of Plantations for Protection

30.3.1 Protection from Desertification

A major reason for the establishment of plantations is the loss of vegetation cover with subsequent erosion of soil, and a qualitative and quantitative change of runoff from watersheds. In arid, semiarid, and dry regions, desertification often marks the endpoint of this sequence. Evans and Turnbull (2004) present a summary of prominent examples in their excellent textbook.

Planting of trees can stop the process of desertification. For instance, in countries bordering the Sahara or in China, the objective of plantation establishment is to reverse the degradation and to establish the so-called green belts. The specific role of planted trees is to stabilize soil, protect the bare ground, and start a process of soil development via litter fall and building up of organic material, soil fauna, and microflora.

Nitrogen fixing tree species are particularly useful, because they can grow on nitrogen deficient soils, tolerate harsh environments, and their leaf litter is more easily degradable in comparison to other species (Evans and Turnbull 2004).

The reason for failure of protective afforestation is often not a silvicultural one, because the techniques have been developed for planting trees on many sites and are proved for years. Poor tree survival in such projects usually results from a lack of aftercare. As it is mentioned by Evans and Turnbull (2004), a successful protection project needs the workout of an effective and proper silvicultural package, which is straightforward and easy to implement. Choice of species, planting methods, tending and protection needs, all need to be well established. For instance, typical criteria for the selection of species for erosion control and planting on impoverished soils are the following:

- Good survival and growth under conditions of water and nutrient limitation
- Ability to produce a large amount of litter
- Strong, wide-spread, and fibrous root system
- Ease of establishment and maintenance
- Capacity to form a dense crown and to retain foliage
- Resistance to pests and diseases
- Good capacity for soil improvement
- Provision of economic returns, e.g. by provision of NWFPs

30.3.2 Watershed Management

Watershed management aims at the sustainable provision of good quality water in a sufficient quantity. Besides the high potential of forests to mitigate the effects of soil erosion, e.g. soil relocation and sediment deposition, forest cover slows the movement of water through parts of the hydrological cycle (Evans and Turnbull 2004).

However, under drier conditions, planting of trees can also reduce the water yield from catchments. For instance, experiments in South Africa showed that planting grassland or spare vegetation with pines (*Pinus patula* and *P. radiata*) and eucalypt (*Eucalyptus grandis*) can reduce annual water yields. Plantations can lower ground water tables and thus conflict the access of people to water resources. The proportion of catchment planted influenced both the rate and amount of change in water flow.

The benefits of carefully established and operated plantations in water catchments to assist erosion control and prevent flooding may far outweigh the disadvantage of some reduction in total water yield. Sites with a long dry season are usually not planted (Evans and Turnbull 2004).

30.3.3 Provision of Shelter

Another objective for establishing plantations is the provision of shelter from rain, direct sun insolation, and strong winds. The so-called shelterbelts are planted to reduce the speed and intensity of wind in agricultural systems. The efficiency depends on the height and permeability of the shelterbelt. Trees for shelter are an important component in silvopastoral systems.

30.3.4 Plantations for Rehabilitation

In many tropical countries, there exists a substantial and often increasing amount of unproductive land, especially concerning over-used, degraded, or abandoned pastures. In developing countries, the loss of land productivity is particularly disastrous because forest and agricultural land-use is often the main generator of income, especially in rural areas. Degradation of land generates economic, social, and environmental concerns. Consequently, rehabilitation measures can have different objectives: rehabilitating or enhancing timber production, preventing further soil erosion, restoring scenic quality, or restoring the natural ecosystem (Vanclay 1994).

The integration of these aspects in silvicultural concepts requires a sound understanding of successional processes, good knowledge about the former land-use, and the resulting barriers for reforestation efforts as well as the consideration of the natural and environmental conditions at the planting sites. However, this is still neglected in many cases. For example, despite the exorbitant number of 2,736 native tree species in Ecuador (Jørgensen and León-Yánez 1999) of which 669 are endangered (240 of them critically to extinction) (FAO 2006), afforestation activities in Ecuador are almost exclusively based on exotic species, mainly pines and eucalypts.

One of the main reasons for this obvious neglect of the native species is the lack of adequate knowledge on their ecology and silvicultural treatment. Another reason is the absence (nonavailability) of forest reproductive material in nurseries, a fact that can be led back to the lack of knowledge about the biological basics (phenology, seed germination and storage, etc.). The above-mentioned example is not only true for Ecuador but also described for many tropical regions (e.g., Butterfield 1996; Butterfield and Fisher 1994; Davidson et al. 1998; Feyera et al. 2002; Holl 1998a).

A survey by FAO showed that tree species from only few genera are used for protective plantations. In Africa, the main tree species planted in protective plantations have been indigenous *Acacia* spp., for example, *Acacia senegal*, *A. seyal*, *A. nilotica*, and *A. mellifera*. Also some exotic *Eucalyptus* species have been planted in Northern Africa (Del Lungo et al. 2006). According to FAO (2002), *Casuarina* spp. occupied more than 1 million ha net area in plantations, 65.3% of which were located in tropical Asia, mostly in India, Vietnam, and Thailand. Most of the plantations of *Casuarina* spp. were for nonindustrial purposes – shelterbelt, coastal dune fixation, and fuelwood. For South America, no substantial information is available.

For tropical lowland and montane forests, a number of studies has been conducted during the last decade to provide a basis for the selection of native species suitable for the reforestation of degraded land and to improve the knowledge about their ecological and silvicultural characteristics (Günter et al. 2009; Haggar et al. 1998; Knowles and Parrotta 1995; Leopold et al. 2001; McDonald et al. 2003; Montagnini 2001; Montagnini et al. 1995, 2003; Weber et al. 2008). Most of these studies clearly showed that silvicultural interventions for the rehabilitation of degraded land must be very well adapted to the environmental conditions of the sites, which are closely linked to the intensity and duration of the former land-use. Consequently, Montagnini (2001) postulated the identification of the most important constraints of an area and the definition of the specific objectives as the initial steps of a reforestation project. Indeed the conditions in different categories of land available for reforestation can strongly vary with regard to the level of degradation, the size of the area, and the natural surrounding. As silvicultural interventions for reforestation represent investments into the land, it is expected from the measures that the resulting vegetation fulfills the objectives of the land owners or the demand of the society for products or services better than the previous ones or in a more sustainable way.

For the reestablishment of forests on degraded areas, several silvicultural options can be employed that are closely related to a gradient of disturbance:

1. Natural succession: On areas with minor disturbance where remnants of the original forest vegetation are still existing and able to recover to forests.

2. Enrichment planting/assisted regeneration: On areas with moderate disturbance where natural recovery is ensured, at least partially (in terms of area and productivity).
3. Plantations: On land where natural recovery cannot be guaranteed or takes too long time.

The minimal requirement to all measures is that they return the land to higher valuation in an adequate time.

30.3.4.1 Natural Succession

Historically, the most common pathway to reforest degraded land has been abandonment and reliance on natural succession. However, often natural regeneration is prevented by several adverse factors. As a consequence, natural regeneration on degraded land does not always operate on a time scale compatible with human needs (Günter et al. 2007; Parrotta et al. 1997).

The most common physical and chemical barriers for the establishment of natural regeneration are extreme climatic conditions (e.g., extreme temperatures, high insolation, seasonal drought) and soil nutrient limitations. These are highly influenced by the intensity and duration of the past land use. Typical biological barriers are missing seeds or rootstock, excessive seed or seedling predation, nonavailability of suitable microhabitats for plant establishment, absence of fungal or bacterial root symbionts, or root competition with grasses or ferns. These factors are well-documented in many studies (e.g., ITTO 2002; Parrotta et al. 1997; Holl 1998b, 1999; Nepstad et al. 1990; Wunderle 1997).

Natural regeneration depends on the seed bank in soil or the input of seeds from adjacent stands. The more the seed bank is reduced with increasing duration of disturbance (agricultural use or burning), the more the regeneration depends on recently dispersed seeds (Ewel et al. 1981, cited in Guariguata and Ostertag 2001; Aide and Cavelier 1994). Seed dispersal is a key factor in restoration of ecosystems (Palmer et al. 1997; Bond and Lake 2003; Wunderle 1997). Several authors discuss the remoteness of areas from existing forest edges as a main hindrance for fast regeneration (Cubina and Aide 2001; Greene and Johnson 1995; Guariguata and Ostertag 2001; Günter et al. 2007; Holl 1999; Lamb 1998; Pokorny 1997). Consequently, natural regeneration can be considered a promising option for reforestation of areas where the distance to remaining forest edges is not too far or where the seed bank in the soil has not been affected too much due to the agricultural use. It is not an economically acceptable solution in areas far away from forest remnants or with high fragmentation where a permanent seed flux into the area is interrupted. Silvicultural interventions that accelerate regeneration and quickly provide benefits to farmers are more likely to be adopted (Carpenter et al. 2004).

30.3.4.2 Enrichment Planting/Assisted Regeneration

Enrichment planting can be a promising rehabilitation option not only in exploited primary forests but also for secondary forests on abandoned land. When seedling establishment or the species composition in natural regeneration is insufficient, inter or underplanting with desired species can be used to speed-up the successional processes or to enhance the "quality" of a secondary forest by promoting economically or ecologically more valuable timber species. Species selection can not only be directed toward the increase of wood supply and enhancing the income-generating potential of the forest (Berish and Ewel 1988; Nigh 1995; Ramos and del Amo 1992) but also toward the biodiversity or conservation. In buffer zones of National Parks or conservation areas, planting of rare or highly exploited species can be a valuable contribution to conservation.

The selection of suitable species has to be in accordance with the objectives of the enrichment planting. Important silvicultural characteristics of suitable species are rapid growth, tolerance to adverse environmental conditions, and regular flowering and fruiting (ITTO 2002). Enrichment planting can be used to mimic natural gap dynamics and to protect soil by maintaining vegetation on the site.

If not properly planned and implemented, enrichment planting can be more costly and inefficient compared with plantations. Therefore, species selection has to be done very carefully and must consider the socio-economic value of the species as well as its ecological requirements and characteristics. For example, species should be well adapted to the conditions at a given site. As already mentioned, enrichment often fails because of poor maintenance and competition with existing vegetation (Butterfield 1995).

In open landscapes where seed rain is disrupted due to huge distance to seed sources, enrichment planting can also be used to stimulate or accelerate natural regeneration as trees represent perches or a food source for seed dispersers. In tropical regions, bats and birds are of fundamental importance for seed dispersal, therefore remnant or planted trees can be very important. Enrichment planting can further be used to improve the connectivity between fragmented forests. In contrast to Butterfield (1995) who concluded that planting of low-value species at low densities is not economically feasible, Janzen (1988) proposed to plant a relatively small number of trees in strategic locations to attract birds or other seed disperser, thus helping to improve natural regeneration. Martinez-Garza and Howe (2003) stated that planting disperser-limited trees may bypass 30–70 years of species attrition ("time tax") by attracting animals. They propose that the "time tax" can be beaten if natural succession of pioneers is actively enriched with plantings of late-successional and deep-forest animal-dispersed species.

30.3.4.3 Plantations

On intensively degraded sites, where seed sources are distant and the seed bank is exhausted, where soils require fast stabilization, or where the productivity of the land has to be restored within a short time, the establishment of plantations might be the best strategy to rehabilitate forest cover (Parrotta et al. 1997; Murcia 1997; Lamb et al. 2005). As pointed out by Sanchez et al. (1985, cited in Haggar et al. 1997), forest plantations on degraded land can also have a rehabilitative role in increasing soil organic matter accumulation and in improving nutrient cycling. One advantage of plantations is its economic productivity usually exceeding that of secondary forests very soon. Lamb (1998) considered timber plantations as one of the few means by which large areas of cleared or degraded landscapes can be reforested, and reviewed various approaches to redesign those plantations for the purpose of restoration of biodiversity. These approaches include using indigenous rather than exotic species, creating species mosaics by matching species to particular sites, and using species mixtures rather than monocultures. According to Guariguata and Ostertag (2001), plantations provide valuable contributions to biodiversity, because they can accelerate succession of forest understory communities through the elimination of weed competition while attracting seed dispersers. Carnevale and Montagnini (2002) and Camus et al. (2003) report that establishment of tree plantations can also facilitate regeneration of native species that could not otherwise grow in open micro-sites or in competition to herbaceous species. In addition, plantations can improve the structural diversity in the landscape and create corridors between fragmented forests, thus fostering seed dispersal across the area (Camus et al. 2003).

Exotic Species Plantations Versus Natives

The most prominent advantage of exotic species is the well-founded knowledge about their biology and silviculture and proven experiences. Lamb (1998) stated that choosing exotics means choosing a technological “package” with an apparent lower economic risk. Exotics usually have good growth rates, well-known wood properties, and their timber or products can be easily marketed (Lamb 1998). Likewise site requirements, nursery, and silvicultural methods are well established, and often they have passed tree breeding programs. Consequently, compared with native species plants forest reproductive material of exotics is usually easily accessible.

These advantages have to be balanced against many disadvantages: the most obvious is that in regions with high biological diversity, exotic species plantations are highly unnatural, and there is a high risk that they invade natural forests thus affecting not only the vegetation of the planted site but also adjacent natural habitats as described by Lorence and Sussman (1986) in Mauritius. Vitousek et al. (1997) call the biological invasion of nonnatives as one of the six human induced global changes because they do not only outcompete the natives but can also infect them with diseases to which they are not adapted to. They often make a return to the prior ecosystem more difficult and expensive. Especially in the vicinity of parks or reserves, exotics must be considered as a threat to the persistence of native assemblages (D’Antonio and Meyerson 2002). Exotics often build up large persistent seed banks and can thus affect a system even after their removal, preventing attainment of the desired restoration. Wunderle (1997) pointed out that exotic Pine and Eucalypt plantations are also unattractive to many organisms because they lack the requisite food resources for survival (e.g., nectar-producing flowers or fleshy fruits).

As several studies revealed, exotics can also alter the hydrological properties of soils. Farley et al. (2004), for instance, found that under plantations with *P. radiata* in the Ecuadorian Andes, the water retention declined drastically with stand age. Soils in the oldest pine stands retained 39, 55, and 63% (at 10, 33 and 1,500 kPa) less water than grassland soils. Soil organic carbon declined as well. Chacón-Vintimilla et al. (2003) found lower pH, lower cation concentrations, decreased N-mineralization rates, and lower microbial biomass as well as increased litter accumulation under *Eucalyptus globulus* and *P. patula* in the Sierra of South Ecuador. They even question the capability of soils under these species to support subsequent rotations or the growth of other species.

Native species have biological advantages that make them superior to exotics at many sites: As they are well adapted to the environment, they are considered to be less susceptible to local stresses. From the ecological point of view, they are valuable because they also allow to maintain the native flora and fauna and to conserve the genetic and biological diversity in situ. However, there is a huge and urgent need for information on the biological and silvicultural characteristics of native species that are suitable for reforestations of degraded land. Another problem is that there exists only little knowledge about their growth performance. However, this could be estimated based on their growth in the natural environment. On the contrary, the utilization characteristics of native species and its products are known by the local people and wood-using industries. Many native rainforest species have a higher market value than exotics or are socially valued by the local community (Leischner and Bussmann 2002). Therefore, native species are especially interesting for plantations, where requirements for ecological services or social benefits are higher than those for timber production.

Exotic Plantations as an Intermediate Step Toward More Native Stands

On severely degraded land, exotics are often the only species that are able to tolerate the extreme site conditions and to restore a forest cover within acceptable time (Sabogal 2005). As several authors already described (Carpenter et al. 2004; Feyera et al. 2002; Lugo 1997; Parrotta 1992), exotic plantations can facilitate the reestablishment of more natural stands and of species richness. Plantations provide shade and cover thus improving the microclimate and allowing the establishment of high concentrations of native tree species, including shade demanding, in their understory (Lugo 1997). The establishment of plantations with exotic species and the subsequent enrichment with native species can therefore also be considered a promising option to rehabilitate productivity or biodiversity of abandoned land.

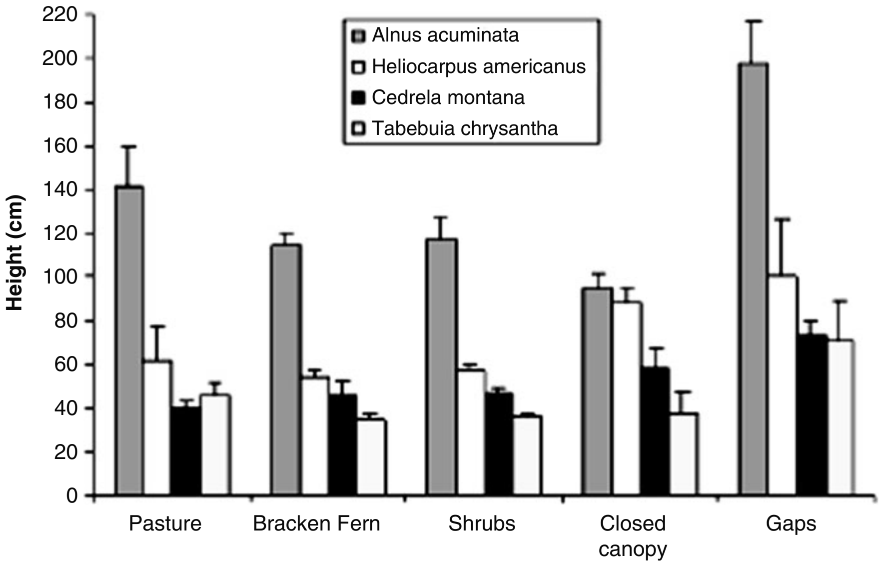

Fig. 30.2 Height of four native tree species 12 months after planting on different sites at the Reserva Biologico San Francisco, Ecuador. Shown are means and standard error (Weber et al. 2008)

To test whether exotic plantations can facilitate the establishment of planted native tree species, Aguirre et al. (2006) conducted an enrichment planting experiment in a *P. patula* plantation in Ecuador. They observed 648 individuals from nine native species under two different treatments: planting under closed canopy and in gaps of the pine plantation. The location adjacent to a reforestation experiment on abandoned pastures at different stages of succession allowed to compare the early development of four native species planted at all sites (Fig. 30.2).

The comparison of the height of the species revealed that the gaps in the pine stand offered the best conditions for all species. The authors argued that the good results in the gaps of the pine stand were due to improved microclimatic conditions and reduced competition of the ground vegetation, and hence concluded that underplanting of pine plantations represents a promising option for the conversion of the existing exotic plantations into more natural forest stands. Their findings support Carpenter et al. (2004), who pointed out that prepreparation of badly degraded sites by planting pines and subsequent intermixing of shade loving native trees may be more successful than planting such species directly onto the open sites.

Depending on the selected species, enrichment planting can serve different purposes: if the dominant interest is on economic objectives, species with high timber value can be used while selection of rare and endangered species could meet the demands for biodiversity or conservation aspects. However, as the presented results show very careful adjustment of the silvicultural methods to the environment at a given site is an absolute prerequisite for successful establishment of plantations with native species.

30.4 Conclusions

There exists a substantial variety of options to use forest plantations for rehabilitative purposes: industrial timber plantations, mixed species plantations, community forestry, and agroforestry.

The main silvicultural means in rehabilitation activities are as follows: assisted regeneration (cutting or pressing of weeds or competitive vegetation, protection from fires, interplanting), minimizing barriers to seed dispersal, germination and growth (pre- or postharvesting measures, fertilization, soil scarification, creation of perches), or accelerating soil improvement and natural seed dispersal (creation of corridors, nitrogen fixing trees, inoculation with mycorrhiza).

It is necessary to mention that for purposes of restoration and rehabilitation, practices used for the establishment of industrial plantations may need to be modified significantly, although much of the operational planning will be similar (Evans and Turnbull 2004). Because forest management frequently uses techniques similar to those used in restoration, restoration and forestry may simply be on the same continuum of management alternatives (Frelich and Puettman 1999). Sarr et al. (2004) suggested that the primary distinctions between restoration ecology and forestry lie in aim and scope. For instance, the principles of species selection and species–site interaction (which are already described by Onyekwelu et al. in Chap. 27) can be applied in restoration as well as in rehabilitation. In many cases, planting of native species is reasonably preferred to planting of exotics. Pioneer species or species with regular and heavy seeding are preferred to species that show periodically and intermittent reproduction events. Nitrogen-fixing ones, for example, *Alnus* or *Acacia*, are preferred to nonnitrogen fixing species. Nevertheless, it is important that local people are interested in the products and services provided by the species.

Silvicultural interventions can be a valuable tool to accelerate recovery of forest productivity and biodiversity on degraded land and thus to improve the socioeconomic situation. A prerequisite for success is that the intended measures do closely consider the actual status and the environmental conditions of the target area as well as the demands and financial conditions of the land owners and the society under consideration of the temporal scale. If fast recovery of the economic productivity is required, investment in planting may be the most effective option.

The following chapters provide examples for rehabilitation efforts from three different regions. Causes and consequences of degradation in the arid and semiarid Namibia, and possible silvicultural options and limitations for rehabilitation are introduced by Seely and Klintenberg. Montagnini and Piotto present efforts of restoring productivity on abandoned pasture lands at a humid tropical lowland site in Costa Rica, using mixed and pure plantations with native tree species. Aguirre et al. (Chap. 33) present a case study from a tropical mountain forest area in the Ecuadorian Andes. They report results from a reforestation experiment with exotic and native tree species within a gradient of three successional phases after abandonment of pastoral use.

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Chapter 31
Case Study Desertification: Central-Northern Namibia

Mary Seely and Patrik Klintenberg

Abstract This chapter presents a case study of use of forest products in central-northern Namibia and its implications for land degradation. Wood is mainly used for domestic fuel and construction. Population increase over the past 100 years has led to increased demand for wood products, which resulted in extensive deforestation, a major cause of desertification in central-northern Namibia. Tree planting projects on communal farms have been launched, both for timber wood and fruit trees, in response to increasing demands and decreasing availability. In addition, forests and woody vegetation are conserved under the Forest Act of 2001. The future of Namibian forest resources depends on the success of these conservation initiatives in empowering communities and alleviating poverty.

Desertification, defined as land degradation and loss of productivity, has been recognised and acknowledged in the Cuvelai basin of the central-northern dryland forest area of Namibia for decades (Erkkilä and Siiskonen 1992; Mendelsohn et al. 2000; Wolters 1994). This area was first permanently settled about 400 years ago in an ephemeral wetland flowing through stabilised Kalahari dunes (Marsh and Seely 1992). People settling the area established livelihoods extensively supported by woody vegetation. The original wood-intensive construction and cooking methods continued to be used as the population increased tenfold over the ensuing centuries (Mendelsohn and el Obeid 2005) (Fig. 31.1).

31.1 Situation: Forests, Species, Regeneration, Growth, Yields

Throughout Namibia, about 10% of all plant species are trees, growing to 1 m or more in height, whereas less than 10% of the northern parts of the country are occupied by forests. Geographically, these forests and woodlands are affected by

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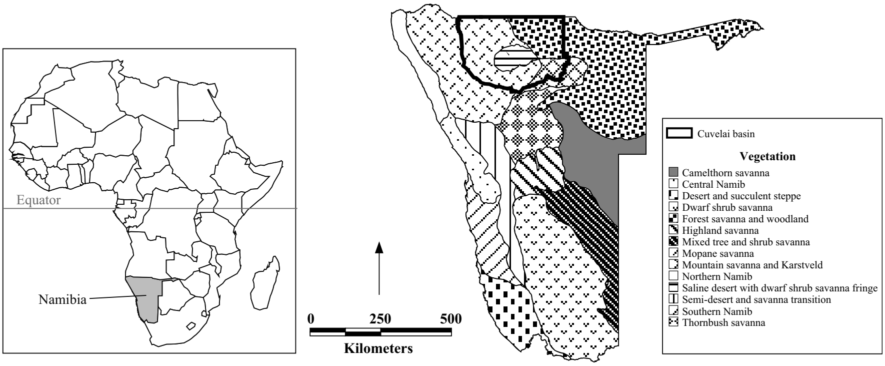

Fig. 31.1 Map of Africa indicating location of Namibia, and map showing vegetation types of Namibia (Giess 1971), and the extent of the Namibian part of the Cuvelai basin

Table 31.1 Commonly used indigenous trees in central-northern Namibia

Family	Genus and species
Anacardiacea	*Sclerocarya birrea*
Arecacae	*Hyphaene petersiana*
Bombaceae	*Adansonia digitata*
Burseraceae	*Commiphora africana*, *C. angolensis*, *C. glandulosus*, *C. mollis*, *C. pyracanthoides*
Capparaceae	*Boscia albitrunca*
Combretaceae	*Combretum collinum*, *C. imberbe*, *Terminalia pruinoides*, *T. sericea*
Ebenaceae	*Diospyros mespiliformis*
Fabaceae	*Acacia arenaria*, *A. erioloba*, *Albizia anthelmintica*, *Baikiaea plurijuga*, *Burkea africana*, *Colophospermum mopane*, *Dichrostachys cinerea*, *Guibourtia coleosperma*, *Philenoptera nelsii*, *Pterocarpus angolensis*
Euphorbiaceae	*Croton gratissimus*, *Schinziophyton rautanenii*
Moraceae	*Ficus sycomorus*
Rhamnaceae	*Berchemia discolor*
Tiliaceae	*Grewia bicolor*, *G. flava*, *G. flavescens*, *G. retinervis*

poor soils, generally shallow, low in nutrients or poor at retaining water, combined with aridity and fires. Although wood biomass may range up to 44 m^3 ha^{-1}, the percentage of timber, defined as *Baikia plurijuga*, *Pterocarpus angolensis*, *Guibourtia coleosperma* and *Burkea africana* with straight trunks longer than 2 m and diameter at breast height more than 45 cm, is never more than 0.5% (Mendelsohn and el Obeid 2005). The most commonly recognised trees used by the residents of central-northern Namibia include those listed in Table 31.1 (Marsh 1994).

Only *Colophospermum mopane* readily resprouts after harvesting but the shoots are not as valuable as the original tree. Regeneration of all species is limited by browsing livestock and fire (Marsh 1994). In a degraded area fenced off from livestock between 1967 and 1993, the estimated annual increment during 26 years of total protection was 0.8 tonnes ha^{-1} (Erkkilä 2001). Indigenous fruit trees are most commonly found in the cultivated fields of local farmers, where they are not specifically planted but are tended once they sprout from discarded seeds after the fruit is eaten (Aitana et al. 2003).

The relationship between the diameter and estimated ages of trees has been determined for a few species of commercial value (Worbes 1999 in Mendelsohn and el Obeid 2005). At an estimated 150 years, *B. plurijuga* has a diameter of 50 cm, *C. mopane* of 28 cm, *P. angolensis* of 52 cm, *C collinum* of 50 cm, *B. africana* 55 cm, whereas *T. sericea* is estimated to be over 100 years old at 40 cm diameter with no larger trees recorded. In general, most of the species are slow growing as a result, inter alia, of prevailing aridity.

Annual yield of indigenous wood used for commercial production is estimated as 100,000 m^3 for fuel wood, 240,000 m^3 for charcoal, 440 m^3 for carvings and 1,000 m^3 of mopane roots also used for carvings. Domestic, non-cash consumption is estimated as 983,000 m^3 and 316,000 m^3 for household construction and fencing (Mendelsohn and el Obeid 2005). These figures were assembled from a variety of different estimates undertaken during the Namibia–Finland Forestry Project 1997–2005. Timber production for export was discontinued in 2003.

31.2 Users and Uses

The direct users of forest products are mainly the rural farmers in northern Namibia. It is estimated that 81% of indigenous wood harvested in Namibia is used for energy production. As elaborated in Mendelsohn and el Obeid (2005), most fuel wood is collected close to rural homes, mostly from dead wood, and is used for domestic cooking (60% of the overall total). Another 6% represents commercially produced fuel wood, primarily from living trees, available for sale in urban centres. Charcoal, comprising 15% of wood production, is harvested from invader bush on degraded commercial rangelands and exported to South Africa or Europe.

Household construction and fencing in northern Namibia consumes an estimated 19% of the total annual harvest (Mendelsohn and el Obeid 2005). The required poles are primarily *C. mopane* and *T. sericea* chosen for their longevity, hardness and resistance to pests. This figure is decreasing as appropriate poles become scarcer and brick houses more common and more wire is used for fencing. The tourism trade consumes less than 1% of the total harvest, although the quality of wood selected may have a disproportionate impact on preferred species, for example *P. angolensis* and *B. plurijuga*.

An experienced farmer and forester in central-northern Namibia summed up the importance of trees as: “without trees there is no life” while local people in the same area described many uses for tree products as expanded on in this paragraph (Marsh 1994). A simple list of uses of wood includes: kraals, houses, furniture and utensils, fences, wood carvings, fuel, mortars and pestles, drinking troughs, protection of wells, ladders, ceremonial fires, firing pottery and ox-sledges. Products from branches, bark and leaves include: toys, twine, tooth brushes, fishing baskets, shelters for chickens and pigs, storage baskets, platforms for drying crops and fencing. Other goods include: fruit, seeds, oil, drinks, medicine for people and livestock, vitamins and nutrients. The trees themselves provide shade, maintain air quality, bring rain, protect the soil, indicate the location of shallow water, support nutritious insects, provide fodder for livestock and are the basis of culture and legends. The diversity of uses in the list above confirms that trees have a key role in the lives of rural residents.

31.3 Causes and Consequences of Degradation

People living in central-northern Namibia are fully aware of the changing landscape (Marsh 1994). One person commented: “When I was young there were trees just like the millet you see in my field, so thick was the forest. In those days we could replace fences whenever we wanted. In 1941, the trees I cut had never been cut before. There are no longer big trees in this area. Today, these trees produce smaller poles”. People ascribe such changes in tree cover to overuse, to an increase in population, to “people from elsewhere” cutting and removing trees for sale, and to decrease in rainfall (Marsh 1994). Grazing of establishing woody vegetation is another component of degradation (e.g. Jobst et al. 1995) and people actively protect young trees often using fencing of *Hyphaene petersiana* fronds (Marsh 1994). Fires are yet another cause of woodland degradation (Mendelsohn and el Obeid 2005).

Erkkilä (2001) ascribes almost all deforestation in two constituencies of Ohangwena Region, central-northern Namibia, to clearing of land for permanent agriculture. He estimates that a population increase of one person leads to about 1 ha of deforestation. Moreover, construction of one household requires about 45 tonnes of indigenous wood and about 3 tonnes per annum for maintenance. Overall, Erkkilä (2001) estimates an annual consumption of 600,000 tonnes, which he points out is lower than the sustained yield.

Based primarily on deforestation and woodland degradation, rangeland degradation, degradation of arable land and soil erosion, the annual economic loss in central-northern Namibia was estimated at a minimum of US$10 million per year (Quan et al. 1994). In terms of deforestation and woodland degradation, the decreased availability of construction materials and fuel wood exacerbated by the increased time required for their collection or the substitute costs of commercially purchased materials were identified as the major costs. The time required for fuel wood collection has increased an estimated 24-fold over 15 years and stands at an average of 1.33 h per day of women’s labour (Quan et al. 1994). In 1994, the cost to the nation of using mopane stakes for fencing a 3 ha field was calculated at N$640 while substitute fencing would cost N$2,168. At the same time, the cost to the nation of building a palisade fence surrounding a traditional homestead from indigenous poles was calculated to be N$18,480. With incomes below N$2,000 per year, availability of woody vegetation is important to household economies (Fig. 31.2).

Fig. 31.2 Livestock kraal in central-northern Namibia built with *C. mopane* poles

31.4 Possible Silvicultural Contributions and Limitations

Silviculture, with its traditional focus on growing and treating trees for poles or timber, started in Namibia in 1894 when trees were first grown experimentally in the central and southern areas by the German Administration (Mendelsohn and el Obeid 2005). By 1910, ten forest stations and nurseries supplied seedlings to commercial farmers for further testing. Attention shifted towards the indigenous forests of northern Namibia under South Africa rule and in 1926 commercial exploitation of natural timber expanded with three sawmills established. Although complete data is not available, logging declined from a peak in the late 1960s, evidently influenced by decreasing availability of indigenous timber. During this same period, large volumes of wood, primarily *Spirostachys africana*, were harvested for fuel and props at local copper mines around Tsumeb. This led to serious bush encroachment, a major component of desertification in Namibia (de Klerk 2004). Under the South African army, several plantations of *Eucalyptus* were established in the northern communal areas, although they are now almost entirely ignored. After 1990, it was realised that it is not sensible to clear indigenous forests with their diversity of products available under prevailing climatic conditions to plant monoculture forests (Director of Forestry, personal communication 2009). Forestry activities had thus shifted from “traditional silviculture” to a brief period of commercial harvesting of indigenous timber to its current position today that focuses on community-based management combined with sustainable use and conservation of forests and other woodland resources (Mendelsohn and el Obeid 2005).

Nevertheless, tree planting in the Cuvelai of central-northern Namibia has expanded since independence in 1990 along two different trajectories (e.g. DoF 2008a). One approach focuses on establishment of nurseries, government and non-government, supporting small-scale tree planting by people and institutions, such as schools in the settled areas of the Cuvelai. The second approach focuses on establishing larger plots, mainly in the hard-pan grazing areas of the Ombuga where population is sparse. This involves mechanised land preparation and provision of water infrastructure.

Government maintains five district offices and three nurseries in central-northern Namibia (Mendelsohn and el Obeid 2005). International sponsorship has led to four additional large nurseries serving settled communal areas (RAP 2000a). Promotion of tree planting has been directed towards communities, schools and individuals. As the director of forestry said more than a decade ago (Marsh 1994), "We believe that social forestry is the answer to the problem of deforestation in Owambo. Social forestry means that the people themselves must plant trees for their own food, fencing, shade, for soil enriching purposes and for fodder to feed their livestock". In such cases, it is difficult to assess the efficiency of tree planting or even to determine what parameters to be evaluated, for example trees planted per cost or per unit of human effort expended, or fruits produced per year or per quantity of water used.

High community demand from nurseries for fast-growing, exotic fruit trees such as guavas (*Psidium guajava*), pawpaw (*Carica papaya*), lemon (*Citrus limon*) and mangos (*Mangifera indica*) has been recorded as opposed to demand for slow-growing, indigenous grafted seedlings. Use of grey water for individual plantings around a homestead is encouraged as lack or cost of piped water is a major hindrance to establishment of community nurseries or homestead woodlots (RAP 2000a). Various woodlot projects have gained the temporary interest of communities to plant and look after trees as long as benefits such as food or cash are provided. In most instances, these community-based projects have not survived beyond the life of the project that initiated the plantation. Although statistics are not readily available, during the year 2007–2008, 327 seedlings were produced in community woodlots and 2,084 were produced in Directorate of Forestry nurseries. Of the latter, 968 seedlings of exotic fruit and shade trees were sold (and 236 were donated) to community members (DoF 2008b). Although not an adequate measure, the revenue generated from seedlings was N$8,259 – while expenditure excluding salaries was N$ 565,000 – indicating limited efficiency, if viewed from this perspective. Social forestry, however, considers that awareness, understanding and involvement in forestry issues are benefits hard to quantify.

This "social forestry" approach has been coupled with promotion of fuel efficient stoves and alternative energy and construction materials (RAP 2000b). Live fencing based on species that are thorny, easily coppiced, relatively unpalatable and fast growing, for example *Euphorbia balsamifera* and *Commiphora africana* has also gained interest (RAP 2000a).

Using the second approach, the Directorate of Forestry has established a major tree planting project in grasslands known as the Ombuga that obtains seedlings from these nurseries and other sources (DRFN 2001). This and other tree-planting projects have targeted areas where people do not live and tall trees do not normally grow due to the presence of hardpan or poor soils. Nevertheless, extensive water infrastructure has been supplied in the areas where these tree planting projects are taking place (Klintenberg 2007) and an establishment rate (over 4 years) of 85% has been recorded (DoF 2008b). Provision of water is an important component of tree planting, for watering during the first 2 years of growth, and a free supply is usually required. Irregular rain represents a key constraint for tree planting in the central-north and throughout the country. This focus on tree planting in the grasslands is partly related to the increased population density in the areas, where forests previously existed on the Kalahari sands (Seely et al. 2008) and the reluctance to dedicate cropland to trees.

To date, 83.5 ha in eight trial plots and community woodlots have been established, where maintenance includes watering, weeding, pruning and pest control (DoF 2008c). To capture interest of the community, vegetable growing for income generation was incorporated into the programme. The overall budget was slightly more than N$ 3 million, indicating a low efficiency in terms of almost 30,000 trees planted. Again, however, other benefits such as employment, awareness and involvement of communities in the activities are considered major benefits. The following species and numbers of seedlings were planted in the trial plots (DoF 2008b) (Table 31.2).

In addition to tree planting, community-based management and sustainable use and conservation of forests and other woodland resources are the major thrusts of current forestry in Namibia. The overall objectives are to empower people and reduce poverty while managing forests and woodland. This approach has benefited from two recent initiatives known as Community Forests and Community-Based Natural Resource Management (Mendelsohn and el Obeid 2005; NACSO 2008). As of early 2005, 13 community forests had been approved encompassing almost 400,000 ha and more than 35,000 people (Mendelsohn and el Obeid 2005). Management by local people focuses on natural resources but also fire management. This not only empowers people but also helps to satisfy a management vacuum not filled by government. Use of resources by local people is a second key point. This encompasses exclusive commercial rights over resources that

Table 31.2 Tree species planted in trial plots on the Ombuga grasslands and number of individual seedlings

Species	Numbers	Species	Numbers
Acacia erioloba	500	*Ficus sycomorus*	200
Acacia galpinii	700	*Grevillia robusta*	100
Acacia karoo	1,900	*Hyphaene petersiana*	700
Acacia nigrescens	1,100	*Kigelia africana*	3,200
Albizia versicolor	100	*Leucaena leucocephala*	3,600
Azadirachta indica	1,800	*Mangifera indica*	800
Berchemia discolor	400	*Olea europea*	300
Casuarina equisetifolia	2,200	*Parkinsonia aculeata*	500
Citrus limon	500	*Peltophorum africanum*	300
Colophospermum mopane	800	*Prosopis glandulosa*	1,500
Diospyros mespiliformis	300	*Psidium guajava*	2,200
Eucalyptus camaldulensis	2,600	*Rhus lancea*	200
Faidherbia albida	800	*Sclerocarya birrae*	2,000

provide material and financial benefits to the poorer rural people (Mendelsohn and el Obeid 2005). Particularly important to the communities involved in community forestry is the ongoing accommodation of livestock grazing within the forest area and accessibility of fuel wood.

Programmes are also in place, which promote farm forestry. Many projects are focused on non-timber forest products, especially those that can be harvested and marketed by poorer rural communities (Mendelsohn and el Obeid 2005). All these different approaches have contributed to the transition from traditional silviculture to harvesting of indigenous forests to community conservation, management and use of forest and woodland products while reversing land degradation, loss of productivity and ongoing desertification.

31.5 Summary and Conclusions

A major element of desertification and loss of productivity in central-northern Namibia is the extensive deforestation that has taken place, predominantly in the past 100 years. This has resulted in the decrease of available fuel wood and the necessity of purchasing or travelling great distances to obtain the required resources. It has also engendered a shift in availability of construction materials. In terms of tree planting, fast-growing fruit trees are preferred by communities that, nevertheless, neither serve as fuel wood nor are they used for construction.

Traditional silviculture has not met with great success in arid to semi-arid Namibia and is not being actively pursued today. Nevertheless, forests and woody vegetation are being conserved, managed and used under the Forest Act of 2001 for benefit particularly of poorer rural communities. This approach is augmented by active planting of fruit and shade trees on communal farms. Future trends in terms of forest and woodland conservation, management and use will depend on the success of community forests and their contribution to empowering communities and alleviating poverty.

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Chapter 32
Mixed Plantations of Native Trees on Abandoned Pastures: Restoring Productivity, Ecosystem Properties, and Services on a Humid Tropical Site

Florencia Montagnini and Daniel Piotto

Abstract Twelve native tree species grown in mixed and pure plantations on degraded pasturelands at La Selva Biological Station in the Caribbean Lowlands of Costa Rica were evaluated at 15–16 years of age. Mixed plantations performed considerably better than pure plantations. The best performing species under both conditions were *Vochysia guatemalensis*, *Virola koschnyi*, *Jacaranda copaia*, *Terminalia amazonia*, and *Hieronyma alchorneoides*. Mixed plantations were among the most productive in terms of volume and carbon sequestered, performing as well as pure plots of fast-growing species. Thinning and pruning improved growth, quality, and stability. At 2–4 years of age, three species suffered less pest damage in mixed than pure stands, while other species suffered no damage or similar damage under both conditions. At 6–8 years, four species suffered total mortality in pure plots and limited survival in mixed plots due to pest damage. Mixed plantations sometimes improved soil conditions such as higher organic matter, and had intermediate values for the soil nutrients examined. Both pure and mixed plantations facilitated tree regeneration by attracting seed-dispersing birds and bats. Income from thinnings and final timber harvest seem to strongly exceed establishment and management costs, providing an economic incentive for similar plantations. These systems are productive options for restoring degraded agriculture or pasturelands of the region.

32.1 Introduction

Several factors can limit or delay natural tree regeneration on degraded tropical lands, including nutrient scarcity, soil compaction, insufficient or excessive soil humidity, high solar radiation, competition from grasses or other aggressive vegetation, and lack of seed availability, especially on sites where distance to seed

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sources may limit propagule dispersal (Montagnini 2008a). Single- and mixed-species tree plantations can play an important role in the recovery of soils, forest structure, and biological diversity, and contribute to catalyzing forest succession in degraded ecosystems (Lamb et al. 2005; Carnus et al. 2006). Here, we report results of experimental plantations with native tree species in mixed and pure designs at La Selva Biological Station in the Caribbean Lowlands of Costa Rica. The objective of the research was to examine the plantations' ability to recover soils, productivity, and ecosystem services on formerly degraded pasturelands. The results are useful in designing and implementing viable strategies to restore degraded pastures on similar sites (Montagnini 2008a).

32.2 Tree Species Choices and Plantation Design

The studies were performed at La Selva Biological Station (10°26′ North, 86°59′ West, 50 m a.s.l., flat terrain, 24°C mean annual temperature, 4,000 mm mean annual rainfall). In the 1950s, the experimental area was deforested, utilized for agriculture production and cattle pastures for about 15 years and then abandoned, in a land-use pattern typical of the region and period (Montagnini 2008a). Immediately prior to plantation, the area was dominated by shrubs and early successional trees interspersed with patches of grass and ferns. The area was surrounded by 15- to 18-year-old secondary forest, and was located about 1,000 m from the primary forest of La Selva Biological Station.

Soils are Fluventic Dystropepts derived from volcanic alluvium; they are deep, well drained, stone free, and acidic (pH in water <5.0) (Montagnini et al. 1995). Results of studies done prior to site preparation for planting showed that soils in the experimental site, apparently as a result of pasture degradation, had lower amounts of organic matter (2.5–4.5%), and lower fertility than small patches covered with young successional forest (Montagnini et al. 1993). According to standards of soil fertility set by the Ministry of Agriculture of Costa Rica, these low fertility soils would not support conventional agriculture (Montagnini 2008b).

Three plantations were established in 1991–1992 with four species each: Plantation 1: *Jacaranda copaia* (Aubl) D. Don (Bignoniaceae), *Vochysia guatemalensis* Donn. Sm. (Vochysiaceae), *Calophyllum brasiliense* Cambess (Clusiaceae), and *Stryphnodendron microstachyum* Poepp. and Endl. (Fabaceae–Mimosoideae); Plantation 2: *Terminalia amazonia* (J. Gmell) Exell (Combretaceae), *Dipteryx oleifera* (Pittier) Record and Mell (Fabaceae–Papilionoideae), *Virola koschnyi* Warb. (Myristicaceae), and *Paraserianthes guachapele* (Kunth) Harms. (Fabaceae–Mimosoideae); Plantation 3: *Hieronyma alchorneoides* Allemao (Phyllanthaceae), *Balizia elegans* (Ducke) Barnaby and Grimes (Fabaceae–Mimosoideae), *Genipa americana* L. (Rubiaceae) and *Vochysia ferruginea* Mart (Vochysiaceae). The plantations were in randomized blocks with four replicates and six treatments: four different species in pure plantations, one mixed-species plantation with four species, and one control area of degraded pasture in which no planting took place. Initial spacing was at 2 m × 2 m and each plot was 32 m × 32 m, with a total of 256 trees per plot.

Tree species selection was based on growth, form, potential recuperation of soil fertility, presence of root nodules in the leguminous species, economic value, and seedling availability. Each mixed-species plantation included trees of different branching patterns, crown shapes, and sizes. Each mixed-species plantation also included at least one legume, one relatively fast-growing species, and one relatively slow-growing species. Within each mixed-tree plot, trees of the four species were planted alternating two species per line. The sequential order of the species within the lines was systematically reversed every other line. In that manner, each row contained the four species of the mixture in a sequence:

1 3 2 4
2 4 1 3
1 3 2 4
2 4 1 3

This systematic design maximized species interactions in mixed plots, with each individual surrounded by individuals of all other species. The design also facilitated some management operations; for example, individuals of all four species could be thinned by removing a complete row of trees (Montagnini et al. 1995). However for attaining the desired final spacing of 4 m × 4 m, trees in the columns also had to be eliminated, with care of retaining similar numbers of individuals of each species.

32.3 Growth and Productivity

At 15–16 years of age, the best performing species were *Vochysia guatemalensis*, *Virola koschnyi*, *J. copaia*, *T. amazonia*, and *H. alchorneoides* (Table 32.1). *Vochysia guatemalensis* grew well in pure plantations, but performed considerably better in mixed plantations. This species is used commonly and successfully by farmers in the region (Redondo-Brenes 2007). *Terminalia amazonia* was a very promising species because of its good growth in pure and mixed plots. This species is an economically viable option for farmers because it has good timber, and it can also be planted in cattle pastures for shade (Nichols and Carpenter 2006), as agrosilvopastoral systems that give early returns to farmers in the form of cattle products, while the trees are too young to be harvested (Montagnini et al. 2003; Montagnini 2009). *Hieronyma alchorneoides* also has high-value, dense wood, although frequent problems with bole form and bifurcation make pruning of lower branches necessary (Piotto et al. 2003). *Hieronyma alchorneoides* is also frequently used in agrosilvopastoral systems in combination with beef cattle. Given the growing interest in native tree species in Costa Rica, currently genetic improvement programs have started to focus on improving the stem form of *H. alchorneoides* and *T. amazonia*

Table 32.1 Average stem density, basal area, total volume, total aboveground biomass, and carbon storage of mixed and pure plantations of native tree species at La Selva Biological Station, Costa Rica

Species	Stem density (# ha^{-1})	Basal area (m^2 ha^{-1})	Total volume (m^3 ha^{-1})	Aboveground biomass (Mg ha^{-1})	Carbon storage (Mg ha^{-1})
Plantation 1: 16.5 years old					
J. copaia	314.9	15.6	210.1	30.7	15.3
V. guatemalensis	380.9	25.0	405.8	86.3	43.1
Mixture of 2 species	454.1	30.3	519.4	99.4	49.7
Plantation 2: 16.5 years old					
D. oleifera	402.8	9.3	92.3	99.1	49.5
T. amazonia	261.2	14.3	203.3	117.1	58.5
V. koschnyi	380.9	21.3	262.8	56.7	28.3
Mixture of 3 species	344.2	18.1	247.2	134.9	67.4
Plantation 3: 15.5 years old					
B. elegans	295.4	10.0	101.8	37.6	18.8
H. alchorneoides	346.7	10.1	142.6	47.0	23.5
V. ferruginea	214.8	12.5	152.8	33.2	16.6
Mixture of 3 species	373.5	15.5	206.2	56.4	28.2

As explained in text, mortality from pests, diseases and an unidentified agent resulted in the elimination of two species in Plantation 1 (*S. microstachyum*, *C. brasiliense*), and of one species in Plantation 2 (*P. guachapele*) and Plantation 3 *(G. americana*). Modified from Piotto et al. (2010)

(Redondo-Brenes 2007). *Virola koschnyi* and *J. copaia* also exhibited good growth in pure and mixed plots, although both species have relatively low-density wood of lower economic value. The five best performing species in this research – *V. guatemalensis*, *V. koschnyi*, *J. copaia*, *T. amazonia*, and *H. alchorneoides* – showed significantly higher growth in mixed plots, indicating that mixed plantations may be a preferable production system for these species.

32.4 Effects of Thinning on Growth and Productivity

Thinning can improve crown development and tree growth in plantations intended to produce sawnwood, as is the case for most of the species used here. In this study, half of each plot was systematically thinned at 3 and 6 years of age to a spacing of 4 m × 4 m, and half of each plot was left unthinned with a spacing of 2 m × 2 m.

All species responded positively to the reduced competition on thinned subplots with higher diameter growth. On the contrary, height growth was not directly affected by thinning (Table 32.2). These results suggest that tight initial spacing and thinning with high extraction of stems can improve diameter growth and timber quality, shorten rotations, and generate mid-rotation income.

Table 32.2 Mean dbh, mean total height, and number of trees per hectare for native species plantations with and without thinning at La Selva Biological Station, Costa Rica

Species	dbh (cm)	Height (m)	Trees ha^{-1}
Plantation 1: 10 years old			
C. brasiliense NT	15.52 (0.34)	12.93 (0.30)	892.7 (24.3)
C. brasiliense T	19.43 (0.40)	12.95 (0.21)	550.7 (37.5)
V. guatemalensis NT	21.85 (0.92)	19.21 (0.70)	848.2 (56.3)
V. guatemalensis T	25.11 (1.00)	18.62 (0.63)	550.5 (188.7)
J. copaia NT	20.16 (0.15)	21.32 (0.29)	1,012 (68.7)
J. copaia T	23.40 (0.76)	22.75 (0.68)	595.5 (34.4)
Mixture NT	18.42 (0.79)	17.69 (0.75)	1,429 (34.5)
Mixture T	26.80 (0.49)	24.29 (0.89)	477.2 (42.0)
Plantation 2: 10 years old			
V. koschnyi NT	20.15 (0.58)	18.25 (0.54)	952.5 (34.4)
V. koschnyi T	23.50 (0.84)	18.89 (0.59)	610.0 (37.4)
D. oleifera NT	13.21 (0.40)	14.12 (0.33)	863.0 (17.3)
D. oleifera T	14.62 (0.36)	15.60 (0.43)	669.5 (65.9)
T. amazonia NT	21.27 (0.79)	19.06 (0.42)	625.0 (38.3)
T. amazonia T	23.11 (0.62)	17.71 (0.30)	461.5 (44.7)
Mixture NT	18.07 (1.84)	15.47 (1.43)	819.0 (28.4)
Mixture T	20.56 (0.98)	16.75 (0.52)	665.5 (24.3)
Plantation 3: 9 years old			
G. americana NT	10.72 (1.31)	10.04 (1.18)	640.0 (50.8)
G. americana T	11.30 (1.18)	9.81 (1.00)	565.5 (38.5)
V. ferruginea NT	18.17 (0.20)	14.70 (0.12)	848.3 (112.3)
V. ferruginea T	23.39 (0.61)	19.96 (0.20)	595.3 (42.0)
H. alchorneoides NT	14.75 (0.50)	13.93 (0.20)	892.5 (103.1)
H. alchorneoides T	15.93 (0.43)	14.72 (0.53)	580.3 (37.6)
B. elegans NT	16.09 (0.58)	13.53 (0.53)	1,012 (41.9)
B. elegans T	17.20 (0.33)	13.76 (0.29)	729.3 (81.9)
Mixture NT	14.80 (0.37)	13.07 (0.29)	1,102 (120.1)
Mixture T	18.48 (0.70)	16.12 (0.42)	715.0 (73.1)

Averages are followed by standard error for each species. Modified from Piotto et al. (2003)
T with thinning, *NT* without thinning

32.5 Biomass Accumulation and Carbon Sequestration

In all plantations, mixed plots produced more aboveground biomass than any species in pure plots (Table 32.1). Mixed plots in Plantations 1 and 2 produced over 100 tons of aboveground biomass ha^{-1} at 16 years of age, comparable to secondary forests of similar age established after swidden cultivation in the region (Brown and Lugo 1990).

In this study, mixed-species plantations sequestered carbon at rates comparable or superior to pure plantations of even the fastest-growing species studied (Table 32.1). Some of these species (*V. guatemalensis*, *T. amazonia*) are currently used for projects involving payment for environmental services made to farmers based on carbon sequestration (Redondo-Brenes 2007).

32.6 Pest Damage in Mixed and Pure Plantations

At 2–4 years of age, 3 of 12 species suffered less pest damage in mixed plantations than pure stands, while other species suffered similar damage or no damage in either stand type (Montagnini 2005a). At 4 years of age, anthracnosis caused by the fungus *Glomerella* spp. resulted in 100% mortality in pure plots of *S. microstachyum*, while 42.2% of *S. microstachyum* in mixed plots survived (Montagnini et al. 1995).

At 6 years of age, root damage by gophers (*Orthogeomys* spp.) caused almost 100% mortality of *P. guachapele*, with no significant differences between pure and mixed plots, although isolated individuals of *P. guachapele* survived in mixed plots at year 14. Similarly, *C. brasiliense* trees suffered almost 100% mortality in pure plots because of severe vascular damage caused by a *Fusarium* fungus at age 14, but isolated trees survived in mixed plantation. *Genipa americana* trees suffered substantial mortality at 6–8 years of age, with no differences in mortality between pure and mixed plantations; no specific agent of damage was identified.

Mixed plantations may reduce the severity of pest damage depending on tree species involved, the pest in question, and plantation age. Because pests and diseases can be an important bottleneck for reforestation, mixed stands may provide significant economic and ecological benefits by reducing losses.

32.7 Impact of Pure and Mixed Plantations on Recovery of Soil Properties

Plantation forests generally ameliorate soils immediately following canopy closure, during the fallow enrichment phase, while site quality may deteriorate during the maximum-production phase as mineral nutrients are taken up by the trees. In our studies, 5 years after planting, soil organic matter was higher under pure stands of *V. ferruginea* than other species, with values of 8.43% $\pm$ 1.29 at 0–5 cm depth. Similarly, increases in soil Ca were found in pure stands of *T. amazonia* and *V. koschnyi*, with values of 1.13 cmol/L $\pm$ 0.31 and 1.55 cmol/L $\pm$ 0.45 at 0–5 cm depth, respectively. The mixed-species plots had intermediate concentrations of Ca, Mg, and K, and higher concentrations of P (Montagnini 2000).

When the plantations were 11–12 years old, top soil (0–5 cm depth) under *V. guatemalensis* had higher pH (5.03 $\pm$ 0.001) and greater Mg (0.84 cmol/L $\pm$ 0.02) than other treatments, perhaps because of the species' high ability to recycle cations (Montagnini 2008b). Similarly, soils under *D. oleifera* had greater K than other treatments with 0.19 cmol/L $\pm$ 0.01. In contrast, the soils under *C. brasiliense* had lower values for the same parameters: pH (4.15 $\pm$ 0.06); Mg (0.26 cmol/L $\pm$ 0.03); K (0.11 cmol/L $\pm$ 0.01). Soils under mixed plantations had the greatest quantity of organic matter, with values at 0–5 cm depth of 10.8% $\pm$ 0.32 in Plantation 1; 8.40% $\pm$ 1.20 in Plantation 2, and 14.7% $\pm$ 1.3 in Plantation 3. Pure plots of *H. alchorneoides* and *V. ferruginea* followed with values at 0–5 cm depth of 12.7% ± 1.55, and 12.2% ± 0.16, respectively. Organic matter and N were greater at 11–12 years than at 5 years, apparently because of litter recycling as the plantations approached maturity. Soil parameters under other species showed no distinct temporal trend (Montagnini 2008b). The mixed plots showed intermediate values for the nutrients examined, and sometimes improved soil conditions such as organic matter. This suggests that the complementary effect on nutrient cycling of the different tree species in mixed plots may create a more balanced soil nutrient status (Montagnini 2005a, 2008b).

32.8 Role of the Plantations as Catalysts of Natural Regeneration

At 15–16 years of age, woody understory regeneration was greatest in pure stands of *V. guatemalensis*, *C. brasiliense*, *T. amazonia*, *V. koschnyi*, *H. alchorneoides* and *V. ferruginea*, and mixed-species plots (Fig. 32.1). In contrast, control plots (abandoned pasture in which no trees were planted) retained low vegetation dominated by grasses and ferns (Fig. 32.2). In Plantation 1, the greatest abundance of regeneration was found in the pure species plots of *J. copaia* (18.2 individuals/16 m^2 subplot), *V. guatemalensis* (17.0 individuals/subplot), and mixed-species plots

Fig. 32.1 Plantation 3, understory in a 15.5 years old mixed plantation plot and a large *Hieronyma alchorneoides* tree at the center. Photo: Daniel Piotto

Fig. 32.2 Plantation 1, control plots dominated by ferns and pure plantation of *Vochysia guatemalensis* at the background. Photo: Daniel Piotto

(17.4 individuals/subplot). The mean abundance of understory plants in the control plots (9.4 individuals/subplot) was significantly lower than in either pure or mixed-species plots. In Plantation 2, the pure plots of *D. oleifera* had the greatest mean number of individuals (23.1 individuals/subplot). The mixed-species plot, control plots, and pure plots of *V. koschnyi* and *T. amazonia* had significantly fewer individuals than the pure plots of *D. oleifera*. In Plantation 3, the mixed-species plot (23.8 individuals/subplot) and *V. ferruginea* (23.3 individuals/subplot) had the greatest abundance. *Balizia elegans* and *H. alchorneoides* pure plots had significantly lower abundance. The control plots had the fewest individuals (12.8 individuals/subplot) (Butler et al. 2008).

Plant species richness in the understory was the greatest in the pure plots of *J. copaia* and *V. koschnyi*, and in the mixed treatment of *H. alchorneoides* + *B. elegans* + *V. ferruginea*. These treatments' ability to generate high species richness and abundance shows that they might be optimal for planting in plantations for restorative purposes (Butler et al. 2008). Understory plant species richness was higher in the tree plantations than in control plots. Although differences in understory species richness were small between pure, mixed, and control plots, there did appear to be differences in the types of species present. While most control plots were dominated by ferns, grasses, and other early successional species, the mixed and pure plantation plots had many shrub and tree species in their understories.

In all cases, there was a significant positive correlation between percent canopy closure and abundance of woody regeneration in the understory (Butler et al. 2008).

Tree species increased shade and litter accumulation, which may have reduced competition with grass species, encouraging woody invasion in the newly established understory. Studies on seed rain and seed dispersal syndromes showed that the plantations facilitated tree regeneration by attracting seed-dispersing birds and bats, while in the nonplanted areas most of the arriving seeds were wind-dispersed (Orozco Zamora and Montagnini 2007). Most of the species found in the understory of the plantations were associated with early secondary forests of the region (Orozco Zamora and Montagnini 2007; Butler et al. 2008). These results suggest that native species plantations can catalyze secondary forest succession, although the extent to which they can promote the establishment of species of more advanced stages of succession is unknown (Lamb et al. 2005). Nevertheless, management interventions are needed to restore the floristic diversity and structural complexity of forests, especially where natural successional processes are impaired by soil degradation, habitat fragmentation, and the loss of native fauna (Montagnini 2005b).

32.9 Economic Aspects

The plantation establishment and maintenance costs at 1 year of age were US \$1,661 ha^{-1}. Maintenance and management costs during the second year were US \$203 ha^{-1} (Montagnini et al. 1995). For the slower growing species, the costs of plantation establishment were lower in mixed than in pure stands due to a reduced need for weeding, because the shade from the faster growing species reduced the growth of weeds in the mixtures. For example, cost of establishment of pure plots of *C. brasiliense*, a slower growing species that started closing canopy after 3 years, was 40% higher than that of pure plots of *J. copaia* that closed canopy after 1 year (Montagnini et al. 1995). Weeding is a major cost in early establishment years: up to 68% of establishment cost for both pure and mixed plots occurred in year 1, because of initial dominance by aggressive grasses and ferns (Montagnini et al. 1995).

Plantations were thinned three times, at 3, 6, and 8 years of age with costs of US \$255, 130, and 194 ha^{-1}, respectively. The income from the second and third thinning together was up to US \$5,000 ha^{-1}. Net present value (NPV), benefit/cost ratio (BCR), and internal rate of return (IRR) were calculated for pure and mixed plots in all plantations for all species. NPV was calculated separately using a discount rate of 5% (Montagnini and Mendelsohn 1997). Stumpage volume was estimated by assuming a milling efficiency of 67% of gross volume (Streed et al. 2006) and stumpage prices were taken from the local markets.

Based on tree measurements at 15–16 years of age, near the rotation time for most of the surviving species (20–25 years), the estimated income from timber sales provides a strong incentive for timber plantations. For instance, the estimated NPV for *V. guatemalensis* at 16 years of age was more than US \$6,000 ha^{-1}, with an IRR of more than 14% (Table 32.3). In terms of NPV, BCR, and IRR, mixed-species plots performed better than pure plots for most species (Table 32.3). With the

Table 32.3 Stumpage volume, net present value, benefit/cost ratio, and internal rate of return of mixed and pure plantations with native tree species at La Selva Biological Station, Costa Rica

Species	Stumpage volume (m^3 ha^{-1})	Net present value (US$ ha^{-1})	Benefit/cost ratio	Internal rate of return (%)
Plantation 1: 16.5 years old				
J. copaia	140.79	1039.83	1.47	7.53%
V. guatemalensis	271.89	6035.33	3.72	14.29%
Mix of 3 species	347.98	8155.09	4.68	15.64%
Plantation 2: 16.5 years old				
D. oleifera	61.83	−553.31	0.75	2.85%
T. amazonia	136.20	146.48	1.07	5.28%
V. koschnyi	176.08	1906.63	1.86	9.22%
Mix of 3 species	165.65	1124.37	1.51	7.71%
Plantation 3: 15.5 years old				
B. elegans	68.21	−290.29	0.87	3.78%
H. alchorneoides	95.53	2654.72	2.20	10.81%
V. ferruginea	102.36	1046.21	1.47	7.73%
Mix of 3 species	138.14	2940.80	2.33	11.25%

Modified from Piotto et al. (2010)

exception of pure plots of *V. guatemalensis*, the mixed plots in Plantations 1 and 3 were more economically viable than all species in pure plots for all variables considered (Piotto et al. 2010).

32.10 Conclusions: Pure and Mixed Plantations with Native Species as Strategies to Recover Degraded Lands

Our results show that several native tree species grow well in pure and mixed designs, providing high timber productivity, soil recovery, plant diversity, and carbon accumulation. The five best performing species of this research – *V. guatemalensis*, *V. koschnyi*, *J. copaia*, *T. amazonia*, and *H. alchorneoides* – showed significantly higher growth in mixed plots, indicating that mixed plantations may be a preferable production system for these species. Mixed plantations provided high carbon accumulation, generally improved soils, and increased understory plant regeneration. Some pests and diseases were less severe in mixed than in pure stands.

Revenue from thinnings and final harvest seem to greatly exceed establishment and management costs. Since much of the relatively high establishment cost is due to high labor inputs, these systems are more profitable when the farmer uses his or her own labor, and when opportunity cost of the land is low, as is the case in degraded/marginal land. Payments for environmental services, which are available in Costa Rica and elsewhere, can contribute to offsetting high initial establishment costs, thus serving as an incentive for reforestation projects. After timber is harvested, liberation of the advanced regeneration in the plantation understory is expected to result in a secondary forest that can be preserved or managed for production.

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Chapter 33
Reforestation and Natural Succession as Tools for Restoration on Abandoned Pastures in the Andes of South Ecuador

Nikolay Aguirre, Ximena Palomeque, Michael Weber, Bernd Stimm, and Sven Günter

Abstract Ecuador is one of the global hot spots of biodiversity. Nevertheless, it faces high deforestation rates and unsustainable land use resulting in a substantial and growing amount of degraded land, which needs to be rehabilitated for productivity and biodiversity purposes. We present the results of a reforestation experiment within a gradient of three successional phases after abandonment of pastoral use. Six native species were tested against two exotics. Furthermore, we analyzed the regeneration potential from the soil seed bank and monitored the development of the diversity of woody species in the natural succession at the different sites. Our results show that dependence on natural regeneration for forest recovery cannot be an acceptable solution for forest users, due to the low speed of recovery and the insufficient species composition of the regeneration. Planted seedlings of native species are able to cope with the harsh conditions if they are selected according to their adaptation to the environmental characteristics of the respective planting sites.

Keywords Biodiversity · Ecuador · Native species · Reforestation · Restoration

33.1 Introduction

Ecuador is part of the 17 mega-diverse countries of the planet and contains portions of two of the world's biodiversity "hot spots," the tropical Andes and the Tumbes-Choco-Magdalena (Mittermeier et al. 1997; Myers et al. 2000; Brummitt and

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Lughadha 2003). However, it also has the highest deforestation rate of South America (FAO 2006). As a result of high deforestation and nonsustainable land use, there now exists a substantial and growing reservoir of unproductive land in Ecuador, especially of previously over used, degraded, and abandoned fields and pastures (Weber et al. 2008). For instance, from 1972 to 1985, pastoral land in Ecuador increased from 2.2 to 4.4 million ha and by 1989 pastures covered an area of about 6 million ha (Wunder 2000; Mosandl et al. 2008).

Tree plantations play an important role in tropical forest restoration and rehabilitation as they can provide canopy cover within a short time (Parrotta 1992; Sampaio et al. 2007). Natural succession is also considered as a strategy to forest recovery providing further advantages such as enhancing floristic composition. Recovery of tropical forests in abandoned pastures is very complex and depends on many factors, including land use history (Pascarella et al. 2000; Chinea 2002), land use intensity (Guariguata and Ostertag 2001), the often limited availability of seeds (Wijdeven and Kuzee 2000), competition with introduced highly competitive pasture grasses (Holl et al. 2000), time since abandonment (Aide et al. 1995; Klanderud et al. 2009), and distance from the forest edge (Zimmermann et al. 2000; Günter et al. 2007).

In Ecuador, initiatives to develop sound concepts for assisting forest recovery are very limited. The few reforestation activities carried out to date are predominantly based on plantations with introduced species, such as *Pinus* and *Eucalyptus* spp. (Günter et al. 2007). Among the main reasons for the neglect of native species in reforestation activities is the lack of knowledge about their ecology and silvicultural characteristics (Alvarez-Aquino et al. 2004; Stimm et al. 2008). In fact, no national organizations exist in the country, which are capable of designing and implementing a structured national research and development program for the forest sector (IITO 2009).

Based on the two approaches mentioned above, and with the aim to identify ways to accelerate the restoration processes, an experiment was set up in 2003 along a gradient of three successional stages of abandoned pastures. The specific objectives of the study were (1) to explore the suitability of native species for reforestation of abandoned pastures, (2) to compare the performance of the native species with that of the commonly used exotics, (3) to identify species-specific reactions to different successional states of planting sites, and (4) to analyze the natural regeneration potential of different types of abandoned pastures.

33.2 Study Area

The study area is located in Southern Ecuador, in the province of Zamora Chinchipe. The test sites are located in the high valleys of the San Francisco River, on the flanks of the Andes, near the research station “Estacion Scientifica San Francisco” (ECSF) in the buffer zone of the Podocarpus National Park (Fig. 33.1).

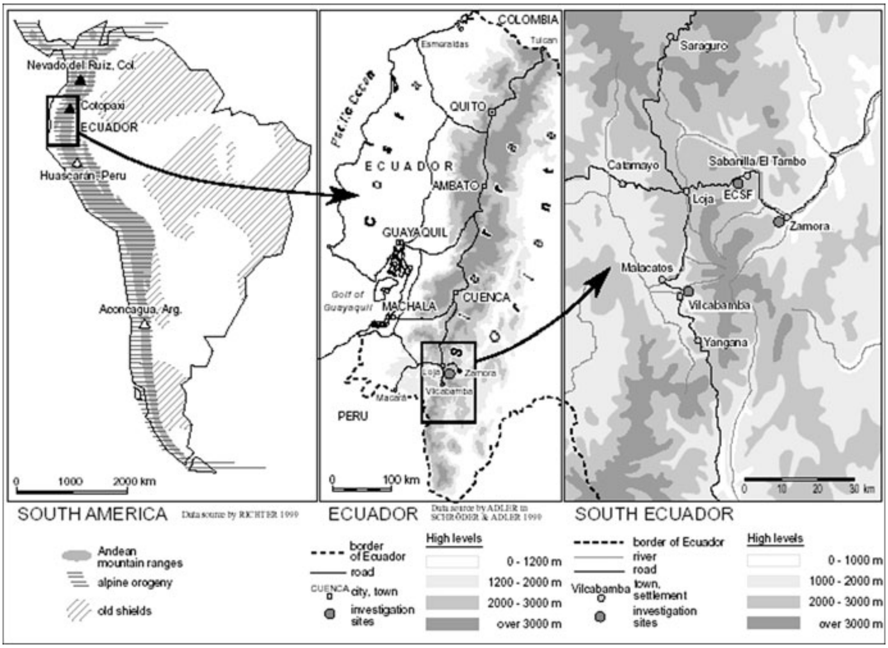

Fig. 33.1 Orientation map, showing the general location of Ecuador and the area under study (from Hagedorn 2001)

The study area occurs along an altitudinal range from 1,800 to 2,200 m asl (above sea level). The climate at ECSF is tropical humid and characterized by 11 humid months a year (Hilt and Fiedler 2005). The precipitation is strongly influenced by the altitude. At 1,950 m asl, mean annual rainfall amounts to 2,000 mm with an extremely wet season from April to July and a less humid period from September to December (Bendix et al. 2006). For the investigations of this study, three different stages of succession after abandonment of pastures have been selected (1) a most recently abandoned pasture used for livestock raising (hereafter: Pasture), (2) a shortly abandoned pasture actually covered by the invasive cosmopolitan fern *Pteridium arachnoideum* (hereafter: Fern), (3) an abandoned pasture where a young secondary forest with shrub vegetation could already become established (hereafter: Shrub).

33.3 Methods

33.3.1 Plantation Experiment

The reforestation experiment had been established in 2003 in a Generalized Randomized Block Factorial design with three successional stages of 4 ha each, six native and two exotic species, and two treatments of the competing herbaceous vegetation (removal only before planting vs. removal before planting and every 4 months after planting for 2 years). The experiment comprises a total of 480 plots with 25 seedlings each planted in pure and mixed species sets with a spacing of 1.8 m $\times$ 1.8 m, and eight replicates of each factor combination. A detailed description of the experimental design is given by Aguirre (2007). The species planted can be classified as follows:

- From the six native species, three (*Alnus acuminata*, *Heliocarpus americanus*, and *Morella pubescens*) are categorized as light-demanding species because they naturally regenerate especially in natural clearances and three as more shade tolerant species (*Cedrela montana*, *Juglans neotropica*, and *Tabebuia chrysantha*) due to their preference to regenerate under a closed canopy.
- The two exotic species (*Pinus patula* and *Eucalyptus saligna*) represent the species predominantly used for reforestation in the study area and more generally in the Andean region of Ecuador.

The data presented here refer to the situation of the control and pure plantations plots 48 months after planting. For the height analysis, the top height (best 20% of individuals in each successional stage) was used.

33.3.2 Soil Seed Bank

At each of the three sites of the reforestation experiment, ten plots were randomly selected for an analysis of the soil seed bank. In addition to these samples, ten plots were also selected in the natural forest of the ECSF to enable a comparison between the seed bank at the disturbed reforestation sites and that of an undisturbed forest. In each of these 40 plots, five soil subsamples were collected with a metallic cylinder of 10 cm^3. The 200 samples were placed in receptacles under controlled conditions of the greenhouse in the project nursery and the germination of all woody species was evaluated over a period of 210 days.

33.3.3 Monitoring of Natural Succession

In addition to the reforestation experiment, 16 randomly chosen plots for each successional stage were established to monitor the natural succession of woody species. In each plot, species richness and abundance were recorded every 2 years from 2003 to 2007. To test whether a reduction of the strong competition by the ground vegetation can accelerate the restoration processes, the herbaceous vegetation was removed in 50% of the plots, while the rest remained untreated. However, since the treatment effect was not significant, the corresponding results are presented as a single group.

33.4 Results and Discussion

33.4.1 Survival and Growth of the Native and Exotic Species Plantations

The results of the plantation experiment revealed species-specific patterns of behavior corresponding to their variable ecological needs. Forty-eight months after planting, the exotic species were more successfully established in terms of survival and growth performance in all successional stages than the native species (Fig. 33.2). Only *T. chrysantha* also showed excellent survival with a total mean of 91.1% over all three sites. This indicates that the species is able to cope with the harsh conditions on abandoned land and can tolerate a wide range of environmental conditions. However, the accumulative growth in height (cm), especially at the Pasture was very low, which is a typical behavior for late successional species.

At the Pasture, the native species *A. acuminata* showed the best growth of all species, including *P. patula* and *E. saligna*. This is considered to be an effect of its high demand for light and the ability to fix nitrogen making the species more competitive, while all other native species in this area suffer under the strong competition of the pasture grass *Setaria sphacelata*, which is known to reduce soil nitrogen and thus resists re-colonization of forest species (Rhoades et al. 1998). Surprisingly, the light demanding pioneer species *H. americanus* showed the lowest survival at the open Pasture and the highest at the Shrub site, but the height development of the species was low on all sites. This is in contrast to the observations in gaps of the natural forest of the ECSF where height growth of 2 m per year is observed. *J. neotropica* showed the poorest establishment of all natives in terms of survival and growth development at all successional sites. At the Fern site, *M. pubescens* showed excellent survival almost reaching that of *T. chrysantha* and the growth performance within the first 4 years was also good compared to

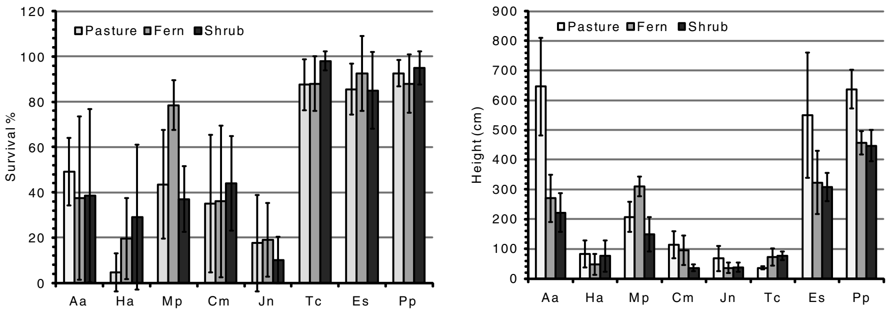

Fig. 33.2 Survival (%) and total height (cm) of seedlings at three different successional stages, 48 months after planting (means and standard deviation); *Aa Alnus acuminata*, *Ha Heliocarpus americanus*, *Mp Morella pubescens*, *Cm Cedrela montana*, *Jn Juglans neotropica*, *Tc Tabebuia chrysantha*, *Es Eucalyptus saligna*, *Pp Pinus patula*

other natives. At the Shrub site, survival of *C. montana* and *T. chrysantha* was higher than at the Pasture, but height growth of both species was quite low compared to *A. acuminata* and the two exotics and can hardly be considered sufficient for large-scale reforestation projects.

33.4.2 Regeneration Potential from the Soil Seed Bank

The results of the analysis of the soil seed bank showed that the number of germinated seeds as well as the species composition differed clearly among the four sites (Table 33.1). The number of germinated seeds increased gradually with each advanced successional site corresponding to about 38,000 N/ha at the pasture and more than 430,000 N/ha in the natural forest. In total, seeds of 15 species from 11 families germinated in the greenhouse. However, compared to the natural forest (=100%) the potential number of seedlings at the disturbed sites was much lower (Pasture = 9%, Fern = 17%, Shrub 35%).

The dominant families were Asteraceae and Melastomataceae, which comprised 54% of all individuals. Only two species (*Brachyotum and Rubus*) were present at all three sites of the plantation experiment. As expected, the proportion of tree species was highest in the samples from the natural forest in terms of individuals (422,100 N/ha) as well as of number of species. However, from the samples of the Shrub site, seeds of five tree species germinated, while at the Pasture and Fern, it was only one in each case (*H. americanus Miconia*). It was surprising that despite the high number of *Heliocarpus* that germinated from the samples of the Pasture

Table 33.1 Number of seedlings ($N \times 10^3$/ha) germinated in the nursery from soil samples of four different sites (Life form: *S* shrub, *T* tree; *N* = 200)

Family	Species	Life form	Pasture	Fern	Shrub	Natural forest
Asteraceae	*Ageratina dendroides*	S		11.5	35.7	
	Baccharis latifolia	S		17.8	40.7	
	Piptocoma discolor	T				15.3
Caprifoliaceae	*Viburnum pichinchensis*	T			10.2	
Clethraceae	*Clethra revoluta*	T			6.8	
Clusiaceae	*Vismia* sp.	T				10.2
Euphorbiaceae	*Hyeronima* sp.	T				10.2
Melastomataceae	*Brachyotum campanulare*	S	10.2	10.2	11.9	
	Meriania sp.	S				10.2
	Miconia sp.	T		10.2	10.2	50.9
Meliaceae	*Cedrela* sp.	T				5.1
Moraceae	*Rubus* sp.	S	10.2	25.5	25.5	
Rosaceae	*Hesperomeles* sp.	T			5.1	
Rubiaceae	*Palicourea* sp.	T			5.1	
Tiliacaeae	*Heliocarpus americanus*	T	18.3			330.4
Total shrubs			20.4	64.9	113.7	10.2
Total trees			18.3	10.2	37.3	422.1
Total			38.7	75.1	151.1	432.3

and the natural forest, no individuals of this species were present in the Fern and Shrub samples.

33.4.3 Development of Natural Succession

The monitoring of the natural succession plots revealed clearly that time since abandonment is an important driver of development of species richness and abundance at all successional sites. However, while species richness increased during the whole observation period at all sites, abundance at Pasture stagnated since 2005. This coincides with the findings of Holl et al. (2000) and Aide et al. (1995) that the dense growth and strong competition of grasses can inhibit the establishment of seedlings in abandoned pastures.

The Shrub site was the most diverse in terms of species richness and abundance followed by the Fern site (Fig. 33.3). Surprisingly, in the last 2 years of the monitoring, the Fern site was the most dynamic. This may be assigned to the fact that the shelter of the fern became stronger with time and provided more shade and protection against high irradiation, strong winds, and the resulting drought stress at the site. Although bracken fern is usually considered a weed species (Marrs et al. 2000) with phytotoxic effects (Dolling 1996; Marrs et al. 2000) that suppresses upcoming seedlings of other plants (Beck et al. 2008), this is not confirmed by our results. Our findings are in line with the results of Günter et al. (2009), who identified a facilitative effect of bracken on the development of planted seedlings of *C. montana* in our study area and Douterlungne et al. (2008), who revealed that *Ochroma pyramidale* (Cav. ex lam.) (Balsa) can also establish well in areas dominated by bracken fern. Consequently, we conclude that under certain conditions, bracken can mitigate extreme environmental factors and thus act as a facilitator for the development of natural succession and biodiversity.

The most dominant family at the Pasture and the Fern sites through time was Asteraceae (Table 33.2), which indicates the presence of many wind dispersed pioneer species. Table 33.3 shows that wind dispersed individuals clearly represent

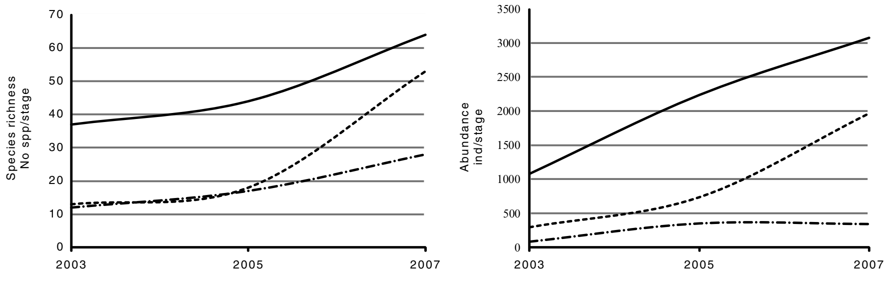

Fig. 33.3 Comparison of species richness (**a**) and abundance (**b**) of woody species at the different successional sites over time (sample area per site: 1,866 m^2)

Table 33.2 List of dominant woody species at the different successional sites (only species with >20 individuals/site), 0 and 48 months of monitoring (*P* pasture, *F* fern, *S* shrub)

Family	Species	Mode of dispersal	Life form	0 Months			48 Months		
				P	F	S	P	F	S
Asteraceae	*Ageratina dendroides* (Spreng) RM	Wind	Shrub	12	203	241	54	673	817
	Baccharis latifolia (R&P) Pers.	Wind	Shrub	20	3	1	55	6	1
	Baccharis tricuneata (L. f.) Pers.	Wind	Shrub				2	40	1
	Baccharis sp.	Wind	Shrub					379	
Caprifoliaceae	*Viburnum pichinchensis* Benth	Bird	Tree			5	4	1	92
Clethraceae	*Clethra fagifolia* Kunth	Wind	Tree					10	57
Cyatheaceae	*Cyathea* sp.	Wind	Treelet			20		1	69
Ericaceae	*Bejaria aestuans* L.	Bird	Shrub				1	74	4
	Bejaria resinosa Mutis ex L.F.	Bird	Shrub		23				
	Gaultheria erecta Vent.	Bird	Shrub				11	168	137
	Vaccinium floribundum H.B.K	Bird	Shrub			2			22
Gentianaceae	*Macrocarpaea* sp.	Bird and bats	Treelet					1	21
Grosilariaceae	*Escallonia paniculata* (Ruiz & Pav.) Roem. & Schult	Wind	Treelet				1	51	2
Melastomataceae	*Axinaea* sp.	Wind	Shrub					2	30
	Monochaetum lineatum (D. Don) Naudin.	Wind	Shrub		2	4	29	240	25
	Tibouchina laxa (Desv.) Cogn	Wind	Shrub	7	3	54	36	27	95
	Brachyotum campanulare (Bonpl) Triana	Bird	Shrub	34	4	16	48	34	394
Myrsinaceae	*Myrsine coriacea* (Sw.) R. Br. Ex Roem. & Schult.	Bird	Tree		36	581	2	142	1,016
Rosaceae	*Hesperomeles obtusifolia* Pers.	Bird	Tree			8		3	25
	Rubus floribundun HBK	Bird	Shrub	1	2	7	29	3	54
Rubiaceae	*Palicourea* sp. 1	Bird	Treelet						32
	Palicourea anceps Standl.	Bird	Treelet					1	54
	Palicourea sp. 2	Bird	Treelet			43			

Table 33.3 Number of individuals of woody species dispersed by animals and wind per successional stage (1,866 m^2) at 0 and 48 months (*P* pasture, *F* bracken, *S* shrub)

Dispersal mechanism	0 months			48 months		
	P N (%)	F N (%)	S N (%)	P N (%)	F N (%)	S N (%)
Animal	7 (8)	71 (24)	87 (20)	76 (22)	453 (23)	1,566 (51)
Wind	77 (92)	226 (76)	347 (80)	268 (78)	1,511 (77)	1,488 (49)
Total	84 (100)	297 (100)	434 (100)	344 (100)	1,964 (100)	3,054 (100)

the prevailing fraction at the Pasture and Fern, although the proportion of animal dispersed individuals at the Pasture increased from 8 to 22% within the observation period, a level which is equivalent to that of the Fern. In contrast, at 48 months of monitoring, the Shrub site was dominated by animal dispersed individuals (51%). The dominant family here was the Myrsinaceae with one single predominant species (*Myrsine coriacea*) whose life form is that of a small tree and whose seeds are dispersed by birds.

At all three sites, only a few species of late successional status were found, presumably due to the distance of the areas to a mature forest, which limits seed input. These results support also the findings of Günter et al. (2007) that the speed of the natural regeneration of abandoned pastures as well as its species composition is not satisfying from a user's point of view.

33.5 Conclusions

In conventional reforestation activities, trees are usually planted directly in open areas. In fact, areas where shrubs or bushes have already established are even manually or chemically cleared or burned before planting. Such conditions may be acceptable for pioneer species and many exotic species, but not for mid and late successional species, such as *C. montana*, *J. neotropica*, or *T. chrysantha*, that require slight shelter. However, it is not yet a well-established procedure in reforestation measures in the tropics to adapt tree species to the successional stage of the dominating vegetation at the reforestation site (Dobson et al. 1997; Lamb et al. 2005; Wishnie et al. 2007).

Our study confirms the well-known qualification of the exotic species Pinus and Eucalyptus in the reforestation of degraded land. However, as several studies revealed, these species have negative effects on the hydrological balance (Farley and Kelly 2004; Buytaert et al. 2007; Vanacker et al. 2007) and on biodiversity (van Wesenbeeck et al. 2003). Therefore, it is necessary to identify native species that are better adapted to the local environment. Our results show that native species, such as *A. acuminata*, *T. chrysantha*, or *C. montana*, are able to cope with the harsh conditions of degraded land. To ensure good survival and growth, it is necessary to choose the species according to their adaptation to the environmental characteristics at the respective planting site. For instance, in the Andean region of South Ecuador, *A. acuminata* could be a promising option for the reforestation of recently abandoned pastures, not only because of its good survival and growth rate but also its ability to improve the nitrogen status of the soil. *C. montana*, *T. chrysantha*, and *M. pubescens* could be valuable species for the reforestation of areas of advanced successional status or land dominated by *P. arachnoideum*. According to our results, the unilateral evaluation of bracken fern as a hindrance for tree seedling establishment has to be reassessed as well. Planting valuable tree species into areas of advanced natural succession (enrichment planting) could also provide a facilitating effect for the establishment of other plants (Vandermeer 1989; Carpenter et al. 2004), a more effective recovery of soil properties (Zheng et al. 2005), and a contribution to faster rehabilitation of biodiversity.

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Part X
Conclusions

Chapter 34
Five Recommendations to Improve Tropical Silviculture

Sven Günter, Michael Weber, Bernd Stimm, and Reinhard Mosandl

Abstract Ecological, technological, and socioeconomic aspects are the main pillars for sustainable forest management (SFM). Based on the suggestions of the authors of this book, and in accordance with these pillars and the main thematic elements of SFM, we identified five fields of actions to improve silviculture in the tropics. There is an increasing demand on timber and other forest products. As forest expansion is in strong competition with other aims of land use, intensification (1) aiming at higher output per area is a promising approach for overcoming this major problem, for example, by short-term rotation forestry, domestication, site improvement, and other measures. Diversification (2) as complementary strategy aims at providing additional products and services, reducing risks and contributing to sustainable management at the landscape level. Therefore, temporal and spatial scales for management (3) have to be adapted to both, needs of individual land owner (usually timber) and to collective needs of societies (protection, water, biodiversity, etc.). Better matching of operational units and ecological spatial scales (e.g., plant–site matching) is an important prerequisite for improving efficiency of silvicultural measures. SFM depends strongly on acceptance of all stakeholders involved. It is a common agreement in science and practice that participatory approaches (4) can contribute significantly to sustainability in this context. However, on the global scale participatory approaches are still in the stage of development. Finally, we illustrate how integration of silviculture and forest management (5) including all mentioned aspects can overcome frequently applied timber mining methods, and leads to a modern approach of silviculture in terms of adaptive ecosystem management.

Keywords Tropical silvculture · Intensification · Diversification · Forest management · Scaling · Participatory management · Adaptive management

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There is broad consensus among the authors of this book together with many other scientists (Bruenig 1996; Dawkins and Philips 1998; Puettmann et al. 2009; Smith et al. 1996; Weber-Blaschke et al. 2005) that ecological, technological, and socioeconomic aspects are the main pillars for silviculture and SFM. The United Nations Forum on Forests acknowledged seven main thematic elements as key components of SFM (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) socioeconomic functions of forests, and (7) legal, policy, and institutional framework. All of these aspects have to be considered to guarantee sustainability on the long-term perspective. However, they mainly describe a rather static framework for silviculture and do not necessarily imply management strategies or guidelines for silviculturists. Guidelines for SFM (e.g., ATO/ITTO, ATO/ITTO 2003; ITTO/IUCN, ITTO/IUCN 2009) usually comprise restrictions for local land users, which are accompanied by loss or omission of income, and are therefore not very attractive (Campos et al. 2001). Consequently, more positive motivation and pathways to overcome the "people-versus-parks" dilemma (Schwartman et al. 2000; Terborgh 2000; Knoke et al. 2009) are urgently needed. Bringing together the manifold suggestions and recommendations of the authors from all over the world contributing to specific silvicultural aspects in this book we can draw some general conclusions. In their case studies, the authors provide concrete examples for tackling specific problems. In accordance with the abovementioned pillars and main thematic elements of SFM, we identified five fields of actions to improve silviculture in the tropics.

34.1 Intensification of Management

"It should be possible to grow most of the wood humans need in managed plantations, and hence eliminate the need to log wild forests." This statement of Sedjo (1999) pinpoints two major aspects of intensive management (1) the extraordinary potential of efficient production of timber in managed plantations, and (2) the connectivity of the productive function of plantation forests with conservation issues in natural forests, which provides an opportunity to unburden natural forests for improved conservation of biodiversity. The more intensive the production of forest products at one site, the more effective can the protection of biodiversity be realized at other sites. In this sense, intensification of timber production in plantations can also be regarded as a conservation strategy. But in spite of great efforts in intensification of plantation forestry and on contrary to the statement of Sedjo, it may not be possible to exclude all natural forests from utilization. Also in natural forests, which could not be conserved in parks, an intensification of forest management will become necessary to fulfill growing demands for forest products, and avoid conversion into agricultural land. This management intensification should not be directed toward increased timber exploiting strategies but rather toward improved quality characteristics of the harvested timber by sophisticated thinning and regeneration techniques.

In general, to be compatible with conservation issues intensification in plantation and natural forests should comply with the following requirements:

1. Adequate spatial distribution of production and conservation areas, for example, maintenance of a complex landscape matrix, including a well-composed mixture of plantations of different type, size, and shape.
2. Delineation of buffer zones and corridors: plantations shall not disturb the connectivity among remaining natural forest patches; strict protection of conservation areas, or areas that are prone to disturbance, degradation, erosion; halting road building or commercial logging in centers of diversity and endemism.
3. Application of dynamic conservation strategies, for example, preference of indigenous species; maintenance or improvement of connectivity among forest fragments as well as of natural understory and genetic diversity.
4. Effective land use planning: Participation of stakeholders to mitigate conflicts; proper classification of soil types, zoning.
5. Stable environmental policies: Clear property rights; land devolution; establishment of land catasters; effective institutions (Chap. 17).
6. Effective control: apply certification; ensure proof of origin; chain of custody to stop illegal logging.

Since most tropical landscapes are located in developing countries, one or more of the abovementioned requirements may not be effectively fulfilled in practice. However, higher intensity of management in planted and natural forests is an important means for meeting the increasing demand for timber and offers also better opportunities for adaptation measures to climate change, at both industrial and smallholder levels.

Subsequently, we will summarize important aspects for intensification in plantations and in natural forests presented in previous chapters of this book.

34.1.1 Intensification in Plantation Forestry

In many tropical countries, plantations are still poorly managed and offer a huge variety of intensification options.

34.1.1.1 Domestication and Tree Improvement

Many plantation species are still in a very early stage of domestication (Chap. 8) limiting the yield and/or quality of the timber produced. For example, for the establishment of plantations very often noncertified or low quality reproductive material (seeds and seedlings) of unspecified regional origin is used (Stimm et al. 2008). Consequently, tree improvement or domestication programs offer huge opportunities for plantation forestry. As the example of Eucalyptus in Brazil shows such efforts can increase the yield of plantations by the factor of 5 (Campinhos Jr 1991 cited in Sawyer 1993).

Domestication integrates the key processes of identification, collection, production, evaluation, selection, and adoption of tree resources (Simons and Leakey 2004). The full domestication process involves the systematic sampling and characterization of genetic variation, development of optimal propagation and silvicultural techniques, and intensive breeding, including the use of molecular genetics technologies and sometimes hybridization (Finkeldey and Hattemer 2007). Domestication seeks to optimize the human benefit within a species as it is refined from a wild tree to a cultivated plant. Tropical forest ecosystems are rich in natural resources, particularly trees that provide food, fuel, fiber, medicine, and various other products, including construction and building materials. Especially, food tree species may become of greater importance within the next years, mainly for sustainable development of rural livelihoods that depend on them. Thus, domestication efforts should go beyond timber species and also involve nonwood forest product (NWFP) species, which can generate additional income and also enhance the diversity of production, thus reducing the risk of losses. Many NWFP species are already in a dynamic process of domestication moving from traditional gathering toward more intensive cultivation (Chap. 10).

34.1.1.2 Site Improvement and Forest Protection

For sustainable production in high yielding tropical timber plantations, adequate site conditions are a crucial prerequisite. On sites with insufficient nutrient availability, high variability of soil conditions, or high risk for pests and diseases the application of fertilizers and chemicals (herbicides, pesticides) may be adequate means to cope with these challenges. There exist a number of silvicultural techniques that can be applied to improve the site conditions for high production: homogenization of site conditions prior to planting (e.g., by plowing, harrowing, trenching), drainage, or irrigation. However, there is still a high need for research to ensure the ecological long-term integrity of those measures and their compatibility with conservation efforts and environmental services.

34.1.1.3 Better Adjustment of Species and Sites

A major concern of plantation silviculture is achieving the best match between the site requirements of a species and the conditions at the planting site, a task that is not always easy to meet given highly variable site conditions and numerous species in the tropics. Consequently, a major challenge for silvicultural science and practice is to develop decision support systems that are based on proper site classification and good knowledge of the site requirements of the tree species. Unfortunately, this information is only available for few tropical tree species with high timber value (e.g., Günter et al. 2009; Stimm et al. 2008). This also concerns the interactions between climate and site variables, which will become of particular importance before the background of adaptation to climate change. In this context, intensively managed plantations with short rotation cycles offer good opportunities to gradually adapt to detected changes by switching to other species, varieties, or provenances, irrigation, fertilization or phytosanitary measures (herbicides, pesticides).

34.1.2 Intensification in Natural Forest Management

In natural tropical forests the most frequent silvicultural system is creaming or exploitation, which usually does not ensure sustainability on the long run. Intensification in management of natural forests does not mean to intensify these systems, but to use systems which are more appropriate for managing these forests sustainably.

Silvicultural systems are normally based on generalizations of the shade tolerance of the key species. For example, Ashton and Hall (Chap. 12) state that shelterwoods with a short period of shelter can be considered appropriate for less shade-tolerant canopy trees or forest types that are driven by stronger episodic disturbance regimes. But they should not be applied to more shade-tolerant species in ecosystems with low disturbance regimes as they have to be considered rather critical for conservation aspects. However, they are usually positive for maximization of yields and cause less damages to the remaining stand and lower costs; in addition, they can be run in longer cutting cycles.

Selection systems in turn are more appropriate for shade-tolerant, slower growing trees where disturbance regimes are small and frequent. They are characterized by shorter cutting cycles, lower harvest volumes per area, slower tree growth in young phases, and more complex forest structures. Unfortunately, selective systems usually cause high damages to the remaining stand. However, reduced impact logging (RIL) offers new opportunities to intensify management in such forests without increasing damages. Especially in forests where conservation aims have to be considered, carefully applied, well-designed, and controlled selection systems may be proper options. Nevertheless, in terms of intensification of production, shelterwoods are much more appropriate.

Another measure to intensify management in natural forests is enrichment planting, which is a promising silvicultural method in the tropics (especially in Africa and the Neotropics) to accelerate achievement of higher densities of desired tree species to enable more constant and profitable yields in the future. Many authors agree that enrichment plantings in deed could enable many forests to be managed in a sustainable way (Kilroy and Gorchov 2010; Kuptz et al. 2010; Abdul Rahmam and Shashiah 2003; Schulze 2008). However, they are usually not applied because of the high initial investment, which has to be discounted for long time periods. The REDD+ mechanisms could offer an excellent opportunity to overcome this basic problem and thus initiate a renaissance of enrichment planting activities across the tropics, at least for forests which are endangered to be converted into agricultural land. In this context, intensification of natural forest management is a suitable example of applied conservation as mentioned above.

It is noteworthy that there is quite a spectrum of other silvicultural activities for intensification of management in natural forests, such as pre- and postharvesting activities, assisted regeneration, improvement thinning, sanitary cuts, etc. However, most of these activities are not applied. For example liana cutting or early liberation of potential crop trees (Günter et al. 2008; Villegas et al. 2009), consideration of plant–site matching aspects (Chaps. 12 and 15), or other pre- and postharvest measures all have an enormous potential for intensification and higher yields from natural forest management, but very few tropical forests are actually managed according to existing silvicultural concepts, mainly due to lacking investment in basic forest management practices but also because of insufficient knowledge about species-specific responses to treatments (Uhl et al. 1997). Thus, intensification in natural forest management is a very promising tool, but it depends largely on progress in research and financial considerations (see below "outlook").

Due to the increasing public awareness of the many ecological and social functions of forests, new financial instruments have been created during the last decennium, which can help to generate additional money for intensified silvicultural treatment in natural forests: for example payments for environmental services (PES), the REDD-mechanisms, ecotourism, or marketing of NWFPs. However, frequently the maximization of one product (e.g., timber) is conflicting with the maximization of another product (e.g., ecotourism). Optimization of the total economic return hence requires appropriate balancing between different aims, for example, by diversification.

34.2 Diversification of Forests Structures, Products, and Services

Since many years, the demand for services and products from forests is steadily increasing and expanding. To meet all these demands in terms of quantity, quality, and type of products and services, forest production cannot be directed toward few products any more but must be highly diverse considering a huge multitude of products and services in different quantities and qualities. Furthermore, the changing climatic and environmental conditions require highly adaptive forests, which are considered to be best guaranteed by diverse forest structures. Thus, besides intensification also diversification should become an essential objective.

Both intensification and diversification have to be regarded as complementary elements of sustainable forest management (SFM) on larger spatial scales. The more intensive and specific the management is on one site, the more important is diversification to mitigate possible environmental damages and shortages in the provision of other goods and services. Another important aspect of diversification besides fulfilling different demands, and improving the conservation value of forests is reduction of risks that are linked with monostructured production.

Diversification can be applied in different aspects, for example diversification of forest structures and diversification of products and services.

34.2.1 Diversification of Forest Structures

There is increasing evidence that the combination of species in mixtures can result in higher growth and production compared to monocultures (Piotto 2008; Richards and Schmidt 2010), which can be attributed to better utilization of the full site potential. Thus, mixtures can also improve economic returns (see also Chap. 32). Additional benefits of mixtures can be provided by positive plant–plant interactions, for example by nutrient input via litter fall and optimum shading conditions.

A key component of mixtures is the possibility to apply risk management. Single products (or services) underlie uncontrollable market price fluctuations. Just like stock markets, a diverse portfolio of land uses or plantations offering different products (with noncorrelated prizes) or different species provides lower risks than highly specialized management with only one single species or product (Knoke et al. 2005). To identify, establish, and maintain the optimal combination of species mixtures, products and the corresponding silvicultural treatment will be one of the key challenges for the future.

Besides mixing species, there are several other possibilities of achieving diversification of forest structures. For instance, all-aged forests are receiving a high attention in the scientific community, especially in the temperate zones. The major advantages of such forests are higher resilience against environmental stress, and permanent forest cover. Therefore, these forests are better compatible with aims for provision of environmental services. However, usually they show lower yields, higher requirements for management, and higher costs for harvesting and tending operations.

From a silvicultural point of view, it is important to reflect on the spatial patterns of species mixtures, ages, and strata, that is, whether to establish small areas of monocultures of different species of a specific age, that is, "coarse grained" mixture or "landscape mosaic," or whether a plantation should be established as a "fine grained" or "intimate" stand mixture of different and intermingled tree species.

34.2.2 Diversification of Products and Services

Several authors in the book indicated that demands of human society for forest goods and services are shifting from pure maximization of timber production to multipurpose management (Weber, Ashton and Hall, Kotru and Sharma, and Günter). They show that in tropical forests there is an increasing demand for NWFPs, environmental services, ecotourism, provision of clean water on the local scale and for mitigation of climate effects and conservation of biodiversity on the global scale (Chaps. 7 and 10). However, not all products and services are really compatible to each other. The success of diversified management depends largely on proper spatial scaling and arrangement of the components. For instance, NWFPs and timber extraction may be compatible at the same site (or same spatial dimension), while timber extraction and ecotourism can hardly be harmonized at the same site and thus depend on spatial segregation.

A good example for the potential of diversification in tropical silviculture could be the management of dry forests (Chap. 16). Implementing silvicultural systems for production of timber in dry forests is particularly challenging due to reduced tree growth, high carbon allocation into roots and high wood densities, which result in low harvestable volumes. Collection of firewood, fodder, foods, and other nontimber products is often more important than management for timber in these ecosystems. In theory, multipurpose management would match perfectly to the manifold demands of forest users in dry forests. Therefore, a combination of silvopastoral or agroforestry systems with coppicing or coppicing with standard methods offer a particular interesting potential for dry forests, especially regarding the high dependence of forest users from subsistence. As dry forests suffer extreme human pressure, many areas of dry forest ecosystems could benefit from PES or REDD+ approaches if it is possible to apply effective control mechanisms and mitigate possible losses of subsistence (due to degradation and deforestation) by additional income from carbon credits.

Conservation objectives have long been considered as conflicting with productive aims in natural forests and plantations. Consequently, for many forest users, conservation aspects have been recognized as restrictions for management. However, since PES are slowly percolating into forest practice, the perception of conservation is changing, and it is becoming an additional objective of silviculture in production forests as well. Thus, when PES can be realized conservation aspects have not necessarily to be seen as restriction but rather as an opportunity. The main problem may be opening the access to potential markets. For example, ecotourism is a common instrument in buffer zone management (at least for areas with good touristic infrastructure), providing additional income but its promotion is not a task of silviculture. Compensation payments are promising instruments for achieving local compensation for implementing global responsibility. This can be realized in different spatial scales, for example, by municipalities for water shed protections on local scale or REDD mechanisms for mitigating climate effects on global scale. Ecuador may be a good example for possible compensation payments on national scale. There is currently intensive international debate for avoided exploitation of oil in the Yasuni National Park; however, the financing still is not clear. These are only some examples indicating the enormous potential of conservation for land owners. Forest managers should consequently analyze prospective markets, negotiate prizes for conservation aspects, calculate costs and tradeoffs with other forest products and incorporate conservation aspects into their silvicultural concepts. This can be achieved either by spatial segregation of protected areas and areas for extraction of products (including NWFPs) or by spatial integration of conservation aspects and low intensity logging (and additional products). Certification of forest products is an additional instrument for merchandizing of conservation aspects.

34.2.3 Consequences of Diversification for Silviculture

As a consequence from diversification, silvicultural concepts have to become broader and more complex than before. Segregation and integration strategies must be applied as complementary components to ensure best public welfare in the future. As a result, silvicultural practice and education must be adapted to the new requirements. Silvicultural management must be highly flexible and dynamic and closely linked with market and economic aspects. This requires that silviculture is accompanied by:

1. Permanent monitoring and controlling of environmental and economic conditions.
2. Looking for possible synergetic effects between classical silviculture for timber and alternative forest products and services.
3. Applying optimization techniques.
4. Dynamic risk management.

34.3 Consideration of Appropriate Scales for Management

Several authors in this book (e.g., Ashton and Hall, Putz, or Weber) stressed the need for better consideration of different temporal (short to long term) and spatial scales (landscape, ecosystems, community, species, and genetic level) in forest planning and silvicultural management. Small-scale forestry can supply a wide array of goods and services. However, the adequate provision of several services depends on bigger scales to become effective, as for instance CO_2 sequestration.

From the management point of view, spatial segregation of highly specialized forest functions bears great advantages: More homogeneous site conditions or lower number of tree species involved alleviate planning, forest operations, control, and merchandizing of products. Less complex silvicultural concepts are easier to implement in the field, an important argument for many tropical countries with lower educational level. Thus, for profit maximization it may be recommendable to specialize on one single product or service which leads to highest internal rates of return. While it may be possible for single land owners to supply few products on a sustainable basis, the provision of multiple services for local, national, or global societies (e.g., erosion, water, recreation, climate, conservation) usually have to be planned, monitored and controlled on higher spatial levels. If not properly integrated in a comprehensive silvicultural concept on a broader scale (e.g., landscape or region) segregated highly specialized production would hardly be able to satisfy the manifold demands of the private and public stakeholders. For example, even sustainable plantation forestry considering all functions, products, and services of forests (beyond timber or biomass) may require effective landscape management, and measures that can compensate local negative effects somewhere else in the landscape (e.g., by protected areas, buffer zones). Increasing claims on land use, for example, for biofuels or food production will complicate finding a sustainable balance between the manifold interests of stakeholders involved, especially in developing countries. Under sustainable multifunctional forest management, it is necessary to include explicit spatial structures and objectives into planning and monitoring. Furthermore, silvicultural activities need to be embedded in sustainable landscape management plans that consider responses of forest ecosystems as well as markets to different silvicultural treatments.

34.3.1 Better Matching of Operational Units and Ecological Spatial Scales

Silviculture in the tropics is usually strongly focussed on operational units, which are closely related to the size of the property of a land owner: for a small-scale farmer the whole property will be his operational unit, while a big concessionaire will divide his concession into different operational units according to forest structure, expected products, or infrastructural or logistic aspects. Furthermore, in practice the operational units are often just schematically adopted to the number of cutting cycles. Consequently, silvicultural treatment often is more determined by the scale of the operational unit than by the ecological dimensions, beginning with tiny spatial scales (e.g., genetic structures, Chap. 8) over intermediate scales (such as the consideration of the variability of sites or the spatial extent of populations) up to global aspects such as global warming (see also Toman and Ashton 1996). A prominent example for mismatching of operational and ecological spatial scales is plant–site matching. A major concern of plantation silviculture is achieving the best match between species requirements and planting site conditions. Given the highly variable sites and numerous species in the tropics with all possible combinations and the limited information from inventories, it was not easy to achieve a good congruency between the ecological and operational units. Thus, in natural forest management site-specific management has been largely disregarded in favor of large-scale and broadly applied management prescriptions (Appanah and Weinland 1993). A major challenge for science and practice is to develop affordable inventory-based decision support systems considering temporal and spatial variability to come to a better plant–site matching (Heinimann 2010). This is of particular importance considering the problems of climate change. To ensure sufficient adaptation of forests to the changes of the environmental conditions, a good adjustment between plants and sites is existential.

34.3.2 Better Matching of Operational Units and Societal Structures: One Step Prior to Participative Management

Irrespective of the deficiencies in plant–site matching, there is also a discrepancy in the scales of the operational units and the societal structures. Forestry is a good example of the potential conflicts between individual and collective decisions and benefits. Mostly, the measures for profit maximization of the forest owner differ from those needed to maximize the benefits for the society. For instance, for a forest owner it may still be highly profitable to exploit natural forest resources (e.g., with high value species) and convert them into more profitable agricultural land use types. Indeed, this is still the reality in large parts of the tropics (Achard et al. 2002). However, there is broad consensus in the scientific community that the individual decisions of land users to convert their forests into agricultural land will have negative consequences for the global society. The prisoner's dilemma (see Lorberbaum 1994) developed in the 1950s describes very well the mismatching of apparently best individual and objectively best collective decisions.[1] The best way out of the dilemma is improved communication and cooperation of the stakeholders. This conclusion can also be applied to SFM. First, conflicting aims between optimization of benefits on the local scale and on the regional or global scale have to be identified, second communicated, and third negotiated between stakeholders. Finally, any restriction for landowners and local land users which is necessary for optimization of forest products and environmental services for the society should be compensated to avoid "illegal" interventions. For this purpose, the scale of the operational unit for the provision of a specific product or service should be related to the corresponding scale of the "consumer." For instance, the mitigation of climate change by carbon sequestration cannot be achieved at one small area but require adequate management of huge areas, while the provision of clear water from a watershed may be assured by proper management of the forests in the catchment area. The operational scale must also be clear to the "consumers" as this affects the required financial resources for compensation of the management activities that go beyond the interest of the forest owner and also the number of forest owners/managers that have to be convinced (or compensated) to include the requested service into their management objectives. The economic valuation of ecosystem services is an important prerequisite for this step (e.g., the economics of ecosystems and biodiversity TEEB). If TEEB can be successfully realized, proper adaptation of management objectives to societal scales will set the stage for achieving maximum collective benefits instead of the sum of maximum individual benefits. However, the proper implementation requires active participation of all stakeholders involved.

[1]Two prisoners are suspected to have committed a crime together. Each prisoner can reduce the individual punishment by confessing the crime: freedom instead of 5 years prison when the other prisoner refuses to give evidence, and 4 years each when the other prisoner confesses, too. The prisoner, who refuses to give evidence when the other prisoner confesses, will receive 5 years. Maximum reduction of collective punishment (2 years each) in turn can only be achieved when both of them refuse to give evidence. However, this decision includes the risk of being "betrayed" by the other prisoner.

34.4 Participation of Stakeholders

Human dimensions have often been disregarded in textbooks about tropical silviculture (Lamprecht 1986; Dawkins and Philips 1998; Bruenig 1996). They rather focused on technical aspects and proper application of silvicultural treatments to produce timber. In recent years, there is an increasing amount of authors, who emphasize the importance of the manifold societal demands: "Forestry is not about trees, forestry is about people" (Westoby 1987). This slogan reflects the definition by Smith et al. (1996): "silviculture is designed to create and maintain the kind of forest that will best fulfill the objectives of the owner and the governing society." Today, the production of timber, though still the most common objective, is neither the only nor necessarily the dominant one. Kotru and Sharma (Chap. 2) depict the complexity of the forest users ranging from forest dwellers, subsistence users, over concessionaires to civil societies and states. Each stakeholder has its specific interests and demands for forests goods and services. At the latest since the global change debate, it is obvious that the global community has become additional stakeholder with specific interest in reducing emissions from deforestation and degradation to mitigate climate change. Since markets do not yet exist for all forest services (e.g., climate and biodiversity), the fair sharing of burdens and benefits from forests has to be negotiated between all stakeholders. At the community level, positive examples of participatory approaches and the transfer of results into silvicultural implementation in the field do already exist and are presented in this book (Chap. 3). On the global scale, participatory approaches are still in the stage of development, for example the access and benefit sharing approach as formulated in the Convention of Biodiversity or as actually under discussion for REDD+ activities (Chap. 4). An important step in this direction is also the formation of the Collaborative Partnership on Forests (CPFs), which can develop corresponding means and regulations on the international level. Many authors in this book indicate practical recommendations and examples for participative approaches in silviculture, for example, the appropriate use of spatiotemporal scaling according to ecological and social requirements. However, an open question that remains is how compensatory measures for potentially reduced internal rates of return for the involved land owners can be realized. Foresters have to recognize that their "competence"-monopole to managing forests is history. Today, multiple stakeholders such as conservationists, indigenous groups, NGOs, or private profit makers are having an impact on the management of forests. It must become a self-evident task for silviculturists to actively involve those stakeholders in their considerations and to communicate the possible silvicultural alternatives, that is which kind of interventions may fit best with the specific aims, but also explain the respective technological and ecological restrictions, limitations, and financial consequences. The decision, which aim(s) shall finally be implemented should be the result of the participatory process of negotiation.

34.5 Integration of Silviculture into SFM Concepts: Moving Ahead from Timber Mining to Adaptive Ecosystem Management

As mentioned in the introduction chapter to this book (Chap. 1), forest management and silviculture without considering impacts of interventions on ecosystems, as a kind of “one-way management” are as old as mankind. However, even today, silviculture in the tropics is quite often still applied in the meaning of “mining” a forest, for example concession forestry or exploitation of forests for subsistence (Chap. 5). In early times of mankind, the main aims were directed toward gathering NTFPs and hunting. The silvicultural concepts in the tropics, however, have been developed almost exclusively for timber production (Dawkins and Philips 1998; Lamprecht 1986). Today, there is an increasing pressure on forest management and silviculture to fulfil both, the demand for products such as NTFPs and timber on a local or regional scale and the additional needs for environmental services such as conservation of biodiversity and climate on a global scale (Benskin and Bedford 1995). It can be summarized that (1) one-way management led to exploitation, and (2) human needs toward forests are variable through time and spatial scales. Thus, identifying the effects of silvicultural treatments (or nontreatments) require permanent monitoring and adaptation to avoid endangering the ecological basis and mismatching with societal needs.

Silviculture has the daunting yet essential responsibility of providing the biological and technical options to achieve management objectives (Dembner 1995). Therefore, control mechanisms, communication between stakeholders and the development of silvicultural concepts adapted to ecological and societal dimensions (temporally and spatially, as mentioned above) require an overlap between forest management and silviculture. Without appropriate silviculture, SFM cannot be implemented. Silviculture without close linkage to societal requirements in turn is unoriented and ineffective. In modern tropical silviculture, forest management and silviculture can hardly be separated.

Figure 34.1 indicates schematically the role of silviculture as mediating discipline between natural and societal dimensions (in accordance to Thomasius & Schmidt 1996; Mosandl and Felbermeier 2001; Puettmann et al. 2009). Silvicultural interventions, monitoring and adaption of treatments according to present situation are concrete activities of practical silviculture with manifestations in the physical landscape. The scientific discipline silviculture, in turn, is deeply rooted in basic ecological disciplines and oriented toward social disciplines and management. Considering the multiple demands and the complexity of the processes in nature, silviculturists are requested to cooperate with scientists and practitioners from ecological, technical, and socioeconomic disciplines. Thus, scientific silviculture has to be understood as an interdisciplinary issue, bringing together ecological and human dimensions at different scales.

Figure 34.2 presents exemplarily different stages of silviculture along a gradient of increasing complexity from forest exploitation to SFM. Figure 34.2a indicates,

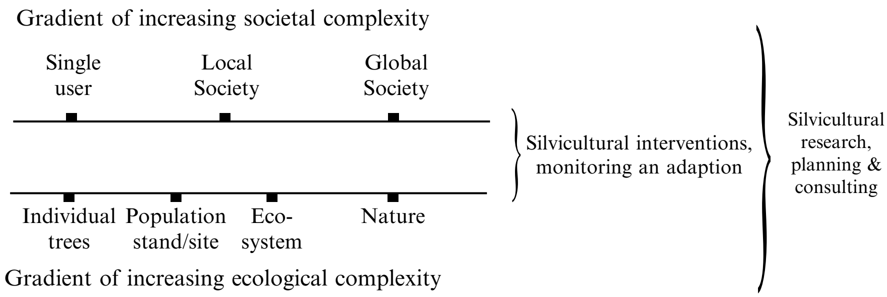

Fig. 34.1 Role of silviculture as mediating discipline between natural and societal dimensions (in adaptation of Thomasius & Schmidt 1996; Mosandl and Felbermeier 2001; Puettmann et al. 2009). Silvicultural activities can be directed toward single users, communities, countries and finally also to the global society. Affected ecological dimensions have to be considered in several scales of complexity, from individual trees, over populations, plant–site interactions, site conditions, and ecosystem functions. While silvicultural interventions, monitoring and adaptation of treatments are concrete activities of practical silviculture with manifestations in the physical landscape, the scientific discipline silviculture goes far beyond these limits; they are deeply rooted in basic ecological disciplines and closely linked to social disciplines and management

for example, the typical situation of basic exploitative silviculture in the meaning of "mining." The local land user defines the requirements toward forests (mostly timber) and implies simple silvicultural operations without any natural monitoring. Neither societal needs are considered nor any ecosystem function. Figure 34.2b includes already first elements of sustainability. The forest user implies a (rather simple) monitoring system at least for his specific product of interest (in this case timber) and adapts his interventions when long-term provision is endangered. Long-term concession forestry would match to this situation. Figure 34.2c illustrates a situation of higher social complexity. Additionally to the demands of the local forest user, local societies define requirements toward the specific forest, for example wood for communal buildings or regulations for communal hunting, ecotourism, etc. This requires an intensive socioeconomic dialog and negotiations, and possibly (but not necessarily) compensations for the land owner. Silvicultural concepts have to include these additional societal requirements and adapt them in case if long-term provision of any service or product is endangered. Model Fig. 34.2D indicates situations where silvicultural concepts do consider population dynamics of target species and their interactions with site conditions. However, the local forest user decides almost exclusively about the products and services required from the forest. Model 2E can almost be considered as a model of advanced approximation to sustainability on a higher level of social complexity. For example, many industrial countries have developed good forest services considering sustainable provision of all main products and services of interest. However, ecosystem aspects are still not completely included. Model 2F represents a (still rather theoretical) case, where one user considers complex ecological interactions (abiotic–biotic, biotic–biotic), static and dynamic ecosystem aspects

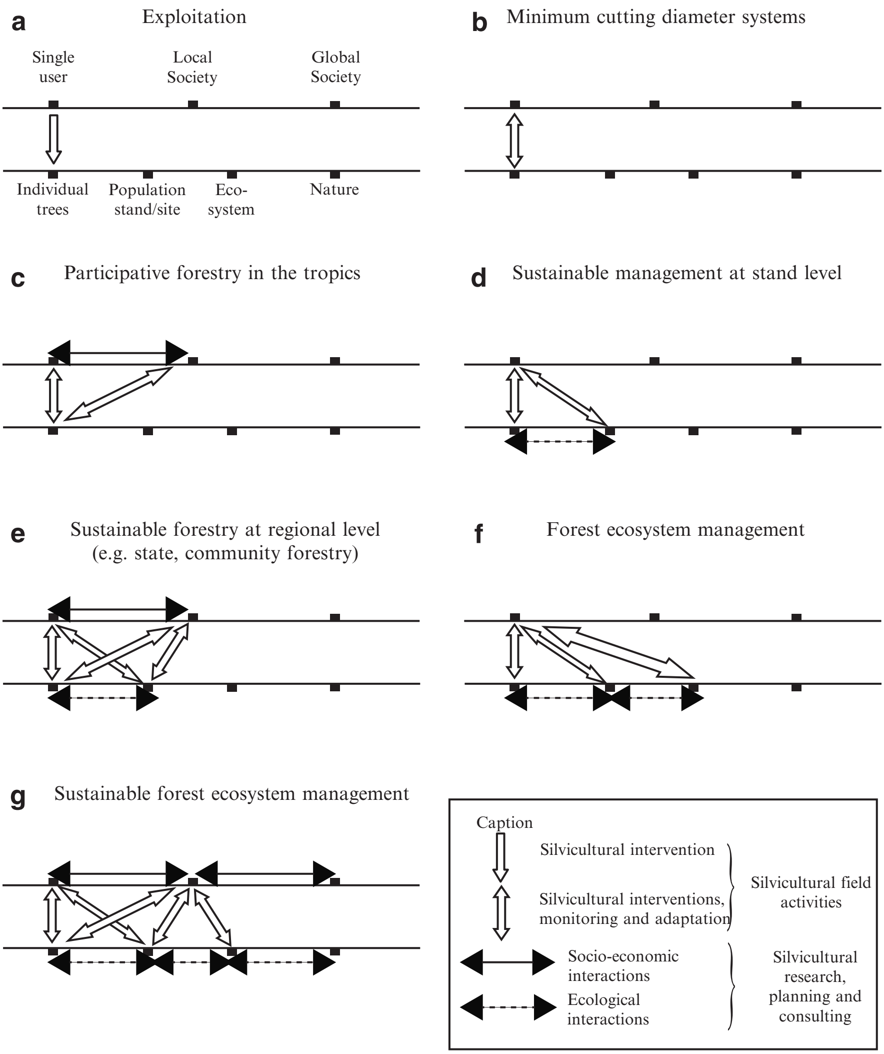

Fig. 34.2 Examples for silvicultural systems along a gradient of increasing complexity, from forest exploitation to sustainable forest ecosystem management. *Simple white arrows* indicate directions of human interventions. *Two-headed arrows* in *white* indicate silvicultural systems including monitoring and adaptation in case of nonsustainable results. *Black arrows* represent ecological (*dashed line*) and socioeconomic interactions (*continuous line*) which have to be considered in silvicultural research, planning, and consulting. Case (**a**) indicates a typical situation of exploitation. Simple monitoring systems for timber volume are applied in case (**b**), interventions and treatments are adapted when mid- or long-term provision is endangered. (**c**) Represents a case with higher social complexity with direct benefits to the land owner and additional requirements from the corresponding local community. Silvicultural concepts have to consider both societal dimensions, usually as results from participatory approaches. (**d** and **e**) Already include more complex societal and ecological dimensions representing "good" silvicultural practice in the tropics today. However, they do not yet consider complex ecological interactions

of all species, products, and services involved (Mosandl and Felbermeier 2001). A consequent further development of this approach under integration of societal and ecological aspects of all spatial and all principles for modern silviculture, presented in this paper leads to model 2G. Silvicultural activities should comprise complex ecosystem aspects (Heinimann 2010), and consider the human needs of all spatial levels. However, the organization of socioeconomic interactions, which are necessary to reach the optimum societal compromise are outside the silvicultural sphere of competence of silviculture and rather related to forest management discipline.

Several authors document the above delineated development of forest management and silviculture from simple exploitation to complex adaptive ecosystem management. First, the sustained yield approach was applied for timber only for decades and almost centuries as baseline for sustainability (Hans Carl von Carlowitz 1713). In the last decades, the term "sustainability" has evolved quickly, in particular since the Brundtland Report. The broad interpretation of this term led to increasing requirements in forest management. For example, foresters in the USA tried to cope with these requirements by renaming their business as "forest ecosystem management" (Kerr 1995). Mosandl and Felbermeier (2001) used this labeling to characterize silvicultural planning and activities, which are oriented to include all ecosystem functions. Heinimann (2010) claimed for "adaptive ecosystem management" instead of classical, and rather static and stand-oriented silviculture. Puettmann et al. (2009) emphasized "management forests as complex adaptive systems." However, it is important to note that introducing "new" terms does not necessarily claim the creation of a new scientific concept. They rather describe the progress of silviculture during time, always putting emphasis on the current state of the art. At least for the English language area, "silviculture" is a well-proven term, indicating clearly the disciplines' unique feature of bringing together ecological and human dimensions. Probably, it is not necessary to give new names to an "old-fashioned" silviculture, but rather give the "old-fashioned" term silviculture, a more modern interpretation. Some definitions of silviculture fit very well to the modern interpretation of sustainability, for example, those of Mosandl and Felbermeier (2001), Puettmann et al. (2009), and Heinimann (2010), or that of Smith et al. (1996) (see above Sect. 34.4) or the definition from Knoke (2010): "silviculture investigates the consequences of decisions about the treatment of forest ecosystems in order to fulfill present and future human needs." Both latter definitions include adaptive elements, ecosystem functions, all forest goods and services, and they imply sustainability as leitmotif for silviculture.

Perfect sustainability is an ideal without a chance of realization (Weber-Blaschke et al. 2005). However, scientific silviculture should aim at achieving sustainability in a process of scientific monitoring, application, and adaptation. In

Fig. 34.2 (Continued) (biotic–biotic, abiotic–biotic), dynamic aspects of all species, products and services involved, as indicated by (**f**). (**g**) Finally represents the consequent development of this approach under integration of all societal and ecological aspects, for example global change issues, and economic valuation and local compensation of providing ecosystem services theory, reaching perfect sustainability would end up in a never-ending cyclic approximation process implying infinite costs and infinite manpower. Thus, in practice the decision maker has to decide, at which cycle to stop this process, which scientific information already provides acceptable accuracy for the prediction of consequences of silvicultural treatments and which grade of scientific accuracy is still affordable.

34.6 Outlook: Open Questions for Tropical Silviculture

Many case studies and review chapters of this book claim the incredible lack of basic ecological data. Even for the supposedly best known tree species such as mahogany or teak, our knowledge is still limited (Chap. 15). The differences in growth performance on different sites are almost unknown for most species, even the most valuable ones. Also the light–plant and plant–soil interfaces and interactions as well as the differences between ecotypes are not sufficiently understood up to now.

Most empirical studies are based on a per plot basis, which implies a relatively low number of individuals on the species level. Especially for silvicultural target species additional approaches and data about growth and mortality covering larger areas, bigger populations and larger time scales under variable environmental conditions are urgently needed. A promising contribution to the solution can be provided by the increasing number of permanent plots either on a scientific basis (e.g., like Yasuni NP or Estación Científica San Francisco Ecuador, or Barro Colorado Island) or for forest services (IBIF, Bolivia). By combining these data with modeling approaches, we can expect great advances in this field. Models can help silviculturists to better understand the complex effects and interactions and to find a balance among the ecological functions and the different needs of the people; however, they will still need the imaginative skill needed for interpreting scientific knowledge, working with nature and within societal expectations (Shepherd 1986; Nyland 1996).

Ashton and Hall (Chap. 12) argue, for example, that selective systems are methods that are often corrupted and applied to the wrong forest autecology. Solutions how to integrate the autecology of various target species into spatially and temporally flexible silvicultural systems are not in sight. The greatest challenge for forest managers and silviculturists is to put together the puzzle of 1,001 ecological publications into a sound concept toward sustainable management. However, institutional links being able to mediate between the international scientific community (mostly publishing in English) and practical forestry in developing countries (mostly working in non-English speaking countries) are largely missing in the tropics. Some promising examples such as CATIE in Costa Rica, or CIFOR in Indonesia may indicate how to bridge this gap.

High sophisticated and innovative approaches for gaining better understanding of ecosystem functions are probably of same importance as calibration of results with simple parameters in the field (e.g., dbh) for easier implementation and control of anthropogenic interventions. While international publications are increasingly directed toward globally valid conclusions for achieving higher scores in citation indices, local application of SFM requires also local case studies for adjusting management practices to site-specific conditions. There is a specific need for a worldwide network of model and demonstration forests and silvicultural transfer projects where scientific knowledge is applied to local conditions and demands.

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