Seppo Kellomäki · Antti Kilpeläinen
Ashraful Alam *Editors*

Forest BioEnergy Production

Management, Carbon sequestration and Adaptation

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Preface

Throughout the world, forests and forest ecosystems provide timber, other raw materials, non-timber benefits, and protection against natural and human-induced threats. Forests are also an important energy source, providing fuel wood and energy biomass. In industrialized countries, fossil fuels have, however, replaced wood and become the dominant source of energy. Currently, coal, oil and natural gas provide cheap options for most human energy needs. Use of fossil fuels is increasing atmospheric concentrations of greenhouse gases (GHGs), especially carbon dioxide (CO_2), with the consequent warming of global climate and changes in precipitation. Global efforts are needed to mitigate the climate change and minimize the impacts of climate change. In this respect, the substitution of fossil fuels with renewable energy sources like forest biomass is among the ways to mitigate climate change. This option is attractive, because it has a direct effect on the global carbon cycle and allows it to be controlled through proper management of forest resources and forest ecosystems. Mitigating climate change through substituting fossil fuels is a new dimension of sustainable forestry and forest management.

This book summarizes recent experiences on how to manage forest land to produce woody biomass for energy use and what are the potentials to mitigate climate change by substituting fossil fuels in energy production. A key question is whether the energy based on forest biomass is carbon-neutral or not and what the possibilities are to reduce CO_2 emissions through proper management integrating timber and energy biomass in forestry. The book outlines the close interaction between the ecological systems and industrial systems, which controls the carbon cycle between the atmosphere and biosphere. In this respect, sustainable forest management is a key to understand and control carbon emissions due to the utilization of forest biomass (e.g. from management, harvesting and logistics, and ecosystem processes), which are often omitted from assessments of the carbon neutrality of energy systems based on forest biomass.

The focus in this book is on forests and forestry in the boreal and temperate zones, particularly in Northern Europe, where the use of woody biomass in the energy industry has increased rapidly in recent years. However, the global dimensions of forests and forestry place local findings in larger perspectives. This concerns especially the questions of the role of forest-based bioenergy in controlling the warming of global climate. Among many things, the book addresses how management can affect the supply of energy biomass using short-rotation forestry and the conventional forestry applying long rotations. In the latter case, there are many links between timber production and the supply of energy biomass, which require careful consideration in the management of forest resources.

We are grateful to all the persons who contributed to this book. Their role was most crucial to offer a wide and deep insight into some current issues which are affecting the use and acceptance of forest-based biomass in energy production. We also want to acknowledge Mr. Harri Strandman, University of Eastern Finland, for his help in preparing and editing the figures of this book. We are also grateful for the support from the "Motive" research program (EU Grant Agreement 226544) of the European Union, the ENERWOODS project of Nordic Energy Research, Kone Foundation and strategic funding from the University of Eastern Finland (SUBI project). The authors are supported by various organizations and/or funding agencies as specified in separate chapters. We gratefully acknowledge this support.

Joensuu, Finland
May 2013

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Chapter 1
Introduction

Seppo Kellomäki, Antti Kilpeläinen and Ashraful Alam

Abstract This textbook deals with the management of forest land for producing and harvesting energy biomass. Energy biomass refers to woody biomass originating from special plantations or forest biomass harvested in forestry primarily aiming at producing timber. The focus is on northern Europe, where there is now great interest in the use of woody biomass as a substitute for fossil fuels in producing energy, and thus in mitigating climate change. The chapters of the book address the potential of the main domestic and exotic tree species in producing energy biomass and the main principles of management to produce energy biomass in forestry in ecologically sustainable and cost-efficient ways. This provides the background for the discussions, which assess the potentials of using forest biomass in reducing carbon dioxide (CO_2) emissions by substituting it for fossil fuels. In this respect, the main focus is on how much CO_2 is taken up in growth and emitted in ecosystem processes, and in the management, harvesting and logistics of energy biomass and in combustion. The impacts of biomass production (forest growth) and energy biomass utilization are indicated by radiative forcing, which may be affected by proper management of forest ecosystems and substituting fossil fuels with energy biomass from forests. The environmental impacts of intensive management and harvesting of energy biomass are addressed in several chapters.

Keywords Climate change mitigation · Energy biomass · Forest biomass · Management · Nordic countries · Substitution · Thinning · Timber

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1.1 Background

For thousands of years, forest biomass or wood has been among the main energy sources of humans around the world. Since the industrial revolution, fossil fuels have replaced wood and become the dominant source of energy around the globe. Currently, coal, oil and natural gas dominate the energy sector, providing cheap and flexible options for most energy needs. The unrestricted use of fossil fuels is increasing the atmospheric concentrations of greenhouse gases (GHGs) at an alarming rate, especially those of carbon dioxide (CO_2) and dinitrogen oxide (N_2O). This increase traps more heat in the lower atmosphere, with the consequent warming of global climate and changes in precipitation in all parts of the world. Global efforts are needed to mitigate climate change and to minimize the impact of climate change. The substitution of fossil fuels with renewable energy sources like forest biomass is among the ways to mitigate climate change.

Currently, biomass (including forest biomass) accounts for over 10 % of the global primary energy supply. Wood-based fuels may comprise the woody biomass originating from special plantations established for producing energy biomass by using fast-growing species (e.g. willows) and intensive management (Fig. 1.1). Wood-based fuels may also comprise forest biomass harvested in forestry primarily aimed at producing timber. Forest biomass may include residues harvested in the tending of seedling stands and in thinning (precommercial and commercial thinning) and residues harvested from clear cut areas. In the latter case, stumps and coarse roots may also be harvested for energy biomass. Forest biomass is also used in the form of industrial residues including bark, saw dust etc. as such or in the form of other energy products, e.g. pellets. Furthermore, recycled wood used in construction and packages may finally be used in energy production.

On the European scale, the current potential to produce woody biomass is about 1,000 million m^3 a^{-1}, of which about 700 million m^3 a^{-1} is forest biomass. At the same time, the demand for woody biomass is about 700 million m^3 a^{-1}, of which slightly more than half is used in forest industry and the rest for producing energy. In the foreseeable future, the demand for forest biomass will substantially exceed the availability; i.e. in 2030 the demand is estimated to be slightly less than 1,400 million m^3 a^{-1}, but the potential supply is 1,100 million m^3 a^{-1}. The main part of the increase in demand represents the use of forest biomass for producing energy (Mantau et al. 2010; Röser et al. 2008).

In the Nordic countries, especially Finland and Sweden, the use of forest biomass has increased rapidly. In fact, energy biomass is a new variety of forest production, which is modifying the management and harvesting regimes used in forestry. This process is driven to a great extent by the commitment of the European Union (EU) to reduce CO_2 emissions in response to the Kyoto Protocol, thus mitigating climate change. Until now, the main part of energy biomass is that produced as a side product of timber production, which refers to the management of a forest ecosystem to produce saw logs and pulp wood. In this context, the energy biomass represents biomass originating from the tending of seedling stands and thinning of young stands

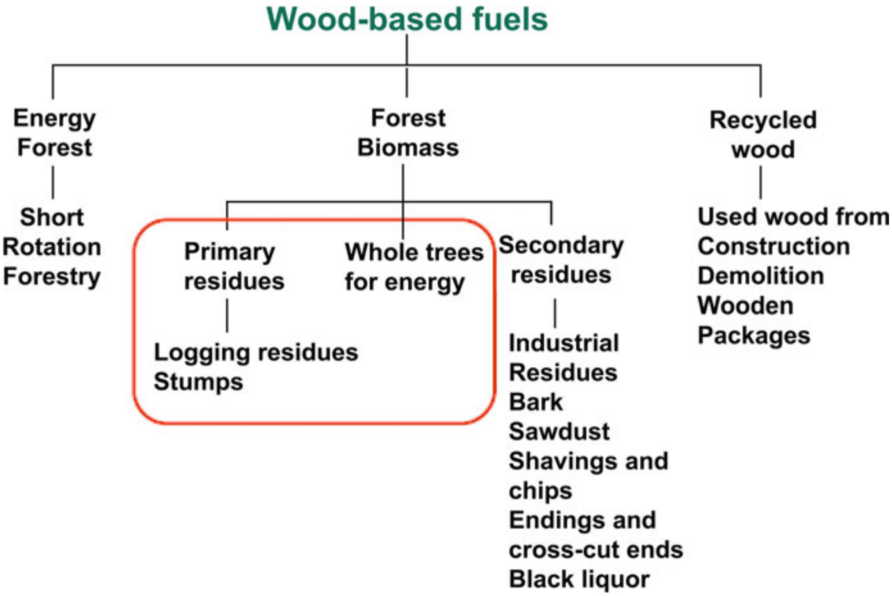

Fig. 1.1 Wood-based fuels and use of forest biomass in energy production in different phases of the life cycle of forest biomass. (Redrawn based on Röser et al. 2008)

not yet providing pulp wood. The growing need to use biomass in energy production has, however, enhanced interest in shortening the production cycle by utilizing fast-growing tree species and intensive management regimes, namely short-rotation forestry. In Sweden, in particular, fast-growing willows are cultivated intensively on agricultural land for energy purposes. This is an attractive option for using agricultural land to produce energy biomass, but it provides methodologies to enhance the production of biomass also in conventional forestry.

Today, the main part of energy biomass is produced in forestry by harvesting logging residues and biomass in precommercial thinnings and the tending of young stand. Appropriate choice of tree species, soil management, and control of spacing and rotation may substantially increase the potential to produce biomass on forest land. Sustainable and cost-efficient production of energy biomass on forest land is possible through integrated management, where the production of timber and energy biomass is balanced in an optimal way.

1.2 Objective and structure of the book

The general objective of this book is to summarize recent experiences on how to manage forest land to produce woody biomass for energy use and the potential to mitigate climate change by substituting forest biomass for fossil fuels in energy production. A key question is whether the energy generated from forest biomass is carbon neutral or not, and, in this respect, what are the possibilities to reduce CO_2 emissions through proper management by integrating the production of timber and energy biomass in forestry. The atmospheric impacts of energy biomass production and utilization are linked to the forest management which controls the sink/source dynamics in forest ecosystems. The concept of integration enables us to approach management strategies, including energy biomass, from the viewpoint of climate change mitigation. The focus is on northern Europe and the Nordic and Baltic countries, where woody biomass is widely used in generating energy.

The book is divided into four Parts.

Part I focuses on the main tree species available for biomass production and their management in biomass plantations (short-rotation forestry) and in forestry aimed at producing timber. In both cases, the impacts of management tools such as spacing in plantation and thinnings are addressed. Furthermore, nutrient management is discussed with the focus on how to avoid the detrimental effects of biomass removal on nutrient resources. On this basis, *Part II* focuses on the sequestration of carbon in the forest ecosystem and the mitigation of climate change by substituting fossil fuels with forest biomass. Uptake and emission of carbon in different phases of the production cycle are addressed in order to identify how the substitution of fossil fuels by biomass may have an effect on the atmospheric carbon and what are the potentials to mitigate the climate change in an efficient way by using forest biomass in generating energy. The importance of the overall analysis of carbon dynamics through the whole production chain of energy biomass "from cradle to grave" is emphasized, with the focus on life cycle assessment (LCA) in identifying direct and indirect emissions of carbon in order to assess the carbon neutrality of energy biomass and the role of energy biomass in mitigating climate change. These issues are further addressed in *Part III*, which focuses on adaptation in climate change and the role of energy biomass in adaptation. In this context, the impacts of climate change on the production potentials of energy biomass are addressed. Furthermore, the risks and uncertainties of future developments and their effects on decision-making are discussed, including the competitiveness of bioenergy in relation to fossil-fuel energy. Finally, *Part IV* summarizes some of the major issues affecting the role of forest energy biomass in mitigating climate change.

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Part I
Forest Management for Bioenergy Production

Chapter 2
Tree Species, Genetics and Regeneration for Bioenergy Feedstock in Northern Europe

Lars Rytter, Karin Johansson, Bo Karlsson and Lars-Göran Stener

Abstract In this chapter we discuss tree species that exhibit rapid growth in northern Europe, i.e. the Nordic and Baltic countries. These species include both common indigenous species and introduced species. We continue with an evaluation of current breeding work and the genetic potential of species that may be suitable for biomass production in this region. Because short rotation times are commonly desired in biomass production, fast, safe and cost-efficient establishment of stands is important. By carefully considering the conditions of the regeneration sites, selecting the most improved plant material from the tree species best suited to each site, and using the best available techniques for stand establishment, we offer guidance to successful growth and cultivation of various tree species to provide society with a renewable biomass supply for energy use.

Keywords Adaptation · Biomass production · Breeding · Clone · Management · Native and non-native species · Nordic and Baltic countries · Planting · Regeneration · Rotation time · Seedlings · Silviculture

2.1 Role of Forests in Supplying Energy Biomass

Large amounts of tree-derived biomass can be produced in regions where the environmental conditions are favourable for forest growth, and where large land areas are available for the cultivation of fast-growing tree species. In the European context, the Nordic (Denmark, Finland, Sweden and Norway) and Baltic countries

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(Estonia, Latvia and Lithuania) are extensively covered by forest land, which represents a potentially valuable source of energy biomass. At present, the supply of energy biomass from forests includes mainly small trees, stem tops and branches, and stumps, whereas stem wood is mainly used in the pulp and sawmill industries. However, there are several ways to enhance the supply of energy biomass available from forests: (1) to allocate more land for cultivating trees and other woody plants for energy biomass, (2) to utilize more efficiently existing forest stocks for energy biomass by exploiting assortments that are currently under-utilized and/or of small value in forest industry, and (3) to increase productivity through choice of tree species, tree breeding and proper management.

In this chapter, we discuss methods to increase forest growth for the supply of energy biomass, including the selection of appropriate tree species, the application of genetic knowledge and breeding improvements, and the efficient regeneration for given combinations of species and site conditions. The focus is on rapid initial growth and the use of short rotation times in biomass production in northern Europe, including the Nordic and the Baltic countries.

2.2 Forest Resources in Northern Europe

In the Nordic and Baltic countries, the total forest area is 69 million ha, of which around 54 million ha are in commercial use (Forest Europe 2011) (Fig. 2.1). The total growing stock of stem wood in this region is 8100 million m^3, of which almost 7000 million m^3 is available for commercial use. The available annual net increment of stem wood is over 237 million m^3 including bark.

Norway spruce (*Picea abies* (L.) Karst), Scots pine (*Pinus sylvestris* L.), birches (*Betula* spp.) and alders (*Alnus* spp.) are the most common tree species in the Nordic and Baltic countries, where they grow in both pure and mixed stands. Altogether, these species account for a growing stock of 7700 million m^3, representing almost 95 % of the total growing stock in the region (Forest Europe 2011) (Table 2.1). This huge volume emphasizes the dominance of these species in producing bioenergy in this region. In the southern parts of the region, oak and beech are also potential sources of energy biomass. Oak and beech currently account for a growing stock of slightly over 110 million m^3. These species are important in Denmark, Latvia, Lithuania and southern Sweden.

2.3 Tree Species Available for Biomass Supply in Northern Europe

2.3.1 Tree Species Available in the Nordic and Baltic countries

The main part of the forest resource in northern Europe is composed of native (or domestic) species, which are well adapted to the prevailing climatic and edaphic conditions (Table 2.1). Furthermore, there are several exotic (or non-native) species

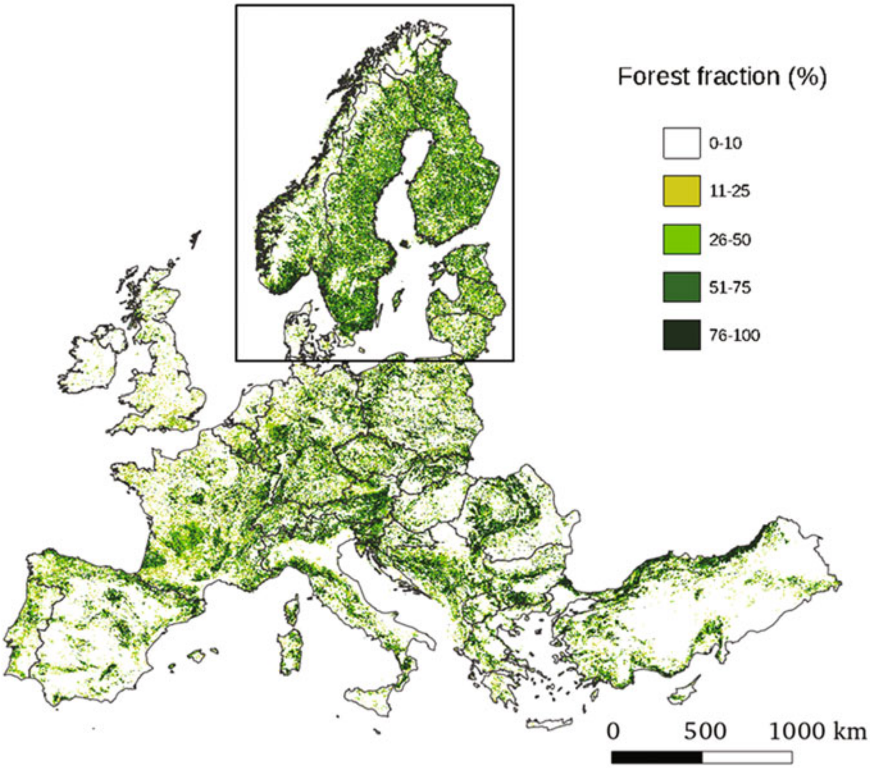

Fig. 2.1 Map showing the forest coverage of Western Europe and Turkey. The map was made for this book using methods presented in Kempeneers et al. (2011)

which grow successfully in these conditions as found in long-term experiments with tree species. Table 2.2 provides information on the regions in which the tree species can be grown, together with appropriate management regimes, while Table 2.3 gives information on productivity and wood density for the different tree species.

Native species include Norway spruce (*Picea abies* (L.) Karst), Scots pine (*Pinus sylvestris* L.), and silver and downy birch (*Betula pendula* Roth and *B. pubescens* Ehrh.), which grow in pure and/or mixed stands. Furthermore, aspen (*Populus tremula* L.) is common, but this species seldom grows in pure stands and even then in small patches, restricting the use of aspen biomass in energy use. Grey alder (*Alnus incana* L. (Moench)) is common in the Baltic countries, while the hardwoods common in Central Europe (e.g. oak, *Quercus robur* L. and beech, *Fagus sylvatica* L.) are abundant only in southern part of the region.

Exotic (or non-native) deciduous and coniferous species may have high growth potentials but their role in supplying energy biomass is still small in the Nordic and Baltic countries (cf. Table 2.1). The exotic deciduous species include, for example, several poplars (*Populus* spp.) and hybrid aspen (*P. tremula* L. × *P. tremuloides* Michx.). The exotic conifers include Sitka spruce (*Picea sitchensis* (Bong) Carrière),

Table 2.1 Growing stocks of native and exotic tree species in the Nordic and Baltic countries

Tree species	Growing stock, (million m^3)	Area as dominant tree species, (ha)
Native species		
Norway spruce (*Picea abies*)	$>$ 2,700	c. 18 million
Scots pine (*Pinus sylvestris*)	c. 3,300	$>$ 30 million
Birch (*Betula pendula*, *B. pubescens*)	c. 1,450	c. 8 million
Alder (*Alnus incana*, *A. glutinosa*)	$>$ 290	–
Aspen (*Populus tremula*)	$>$ 150	–
Oak (*Quercus robur*)	$>$ 70	–
Beech (*Fagus sylvatica*)	$>$ 40	–
Exotic species		
Lodgepole pine (*Pinus contorta*)	c. 30	c. 600,000
Sitka spruce (*Picea sitchensis*)	–	c. 85,000
Douglas fir (*Pseudotsuga menziesii*)	–	$>$ 6000
Grand fir (*Abies grandis*)	–	c. 3000
Hybrid larch (*Larix* × *eurolepis*)	c. 1.4	–
Siberian larch (*Larix sibirica*)	–	c. 30,000
Populus (excl. *P. tremula*)	–	c. 5000

Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), Grand fir (*Abies grandis* (Douglas ex D. Don)) and hybrid larch (*Larix* × *eurolepis* Henry), which are successful in the southern parts of northern Europe along with the lodgepole pine (*Pinus contorta* Douglas ex Loudon). Furthermore, the Siberian larch (*Larix sibirica* Ledeb.) is successful even at the Arctic timber line in the north, but its importance in biomass supply is still unexplored. The deciduous species also include fast-growing willows (*Salix* spp.). These are mainly grown in biomass plantations on agricultural land, thus they are classed as agricultural biomass rather than forest biomass.

2.3.2 Common Native Species

2.3.2.1 Norway Spruce

In northern Europe, Norway spruce is a dominant species, occupying 18 million ha of forest land, with a total growing stock of 2700 million m^3 (Keskkonnateabe Keskus 2010; Bekeris 2011; Finnish Forest Research Institute 2011; Statistics Norway 2011; Danmarks Statistik 2012; Directorate General of State Forests 2012; Swedish Forest Agency 2012). Norway spruce is also the most planted tree species in the region, with more than 350 million plants being produced annually (Finnish Forest Research Institute 2011; Swedish Forest Agency 2012).

Norway spruce is native throughout the Nordic and Baltic countries except Denmark (Hultén 1950) and is found here (Seppä et al. 2009). It is a shade-tolerant species with comparatively low initial growth but high growth during the later phases of the rotation period, and it can grow in stands of high density without losing vigour.

Table 2.2 Regions and sites suitable for different tree species used for the production of energy biomass in the Nordic and Baltic countries. The planting density, number of thinnings and rotation time included in this table are suitable for intensive management in conventional forestry

Tree species	Suitable region, sites	Planting density (plants ha^{-1})	Thinnings (number)	Rotation time (years)
Coniferous species				
Norway spruce	Whole region, preferably mesic and reasonably fertile soils	2,000–2,500	2–3	55+
Scots pine	Whole region, preferably mesic and dry sites, not the most fertile soils	2,000–2,500	2–3	70–100
Lodgepole pine	Above 60 °N on most sites except moist and very fertile	c. 2,300	2	60–100
Hybrid larch	Below 59 °N on medium fertile soils	2,000–2,500	up to 5	c. 40
Siberian larch	Above 59 °N on fertile soils	2,000–2,500	up to 5	80–100
Sitka spruce	Maritime climate, along the coasts in the southern part	c. 2,500	0–5	35–80
Douglas fir	Mainly below 59 °N on mild mesic to somewhat dry sites	750–1,500	3–4	80–100
Grand fir	Limited knowledge, probably like beech	2,500+	2–6	50–80
Deciduous species				
Birch	Whole region, silver birch on mesic sites, downy birch on moist sites	1,500–2,000	2–3	c. 40
Grey alder	Above 59 °N in the Nordic countries, in the Baltic area, mesic to moist soils	2,000–3,000	2–3	25–40
Aspen	Whole region, preferably mesic and fertile soils	c. 2,000[a]	2	c. 50
Oak	Below 60 °N on deep, fertile, mesic soils, also clay soils	400–800 grouped	5–10	100–150
Beech	Below 57 °N on deep, mild, fertile soils, preferably slopes	> 4,000, seeding	4–7	90–120
Poplar	Below 59 °N on mild, fertile sites, pref. agric. land. Future expansion	1,000–1,500	0–2	c. 20
Hybrid aspen	Below 59 °N on mild, fertile sites. Expanded in the future	c. 1,100	0–2	20–25

[a] Aspen is usually regenerated via root suckers rather than planted

Table 2.3 Productivity of stem wood of selected trees species representing natural populations and the genetic gain representing populations originating from genetically improved trees. The values represent the populations in recommended regions (Table 2.2)

Tree species	MAI for "natural" stands (m^3 ha^{-1} a^{-1})	Genetic gain (%)	MAI for improved plant material[g] (Mg ha^{-1} a^{-1})	Basic wood density (kg m^{-3})
Norway spruce	10–14	10–30	4–6	350
Scots pine	7–9	10–20	3–5	440
Birch[a]	7–10	10–30[e]	4–6	480
Grey alder	10–15	n.a.	4–5	360
Aspen	7–10	n.a.	3–4	380
Oak	4–6	n.a.	2–3	575
Beech	5–8	n.a.	3–5	580
Poplar	20–25[c]	n.a.	7–9	345
Hybrid aspen	15–20[d]	c. 25[f]	c. 9	360
Lodgepole pine[b]	9–13	10–20	4–7	430
Hybrid larch	10–14	n.a.	4–6	450[h]
Siberian larch	7–10	n.a.	4–6	600[h]
Sitka spruce	11–20	15–20	6–9	360
Douglas fir	15–20	n.a.	7–9	450
Grand fir	25–30	n.a.	9–10	350

MAI is the mean annual increment; *n.a.* not available. [a] Refers to Silver birch; [b] In the northern part of the region; [c] Result obtained with the OP42 clone; [d] Initial selection of clones; [e] With the Ekebo3 material; [f] Current commercial material for southern Sweden; [g] Where improved material was not available, the figures were based on the productivity in natural stands; [h] Density based on volume with 5 % moisture content, and thus resulting in an overestimation of the productivity in terms of mass

The productivity of Norway spruce on fertile sites (Table 2.3) is 10–14 m^3 ha^{-1} a^{-1} for stands generated from unimproved plant material (Eriksson 1976). The rotation used in managing Norway spruce is generally over 55 years (Table 2.2). The wood of Norway spruce is fairly light, with a basic density of 310–400 kg m^{-3} (Hakkila 1966; Brolin et al. 1995). Harvest residues from Norway spruce, consisting of branches and top parts of stem, are an important source of energy biomass in Finland and Sweden (Brunberg 2011; Parviainen and Västilä 2011).

2.3.2.2 Scots Pine

Scots pine is widely distributed in the Nordic and Baltic countries, and Scots pine forests cover more than 30 million ha in this region. Scots pine grows even on poor sites, where its ability to tolerate water shortages is of utmost importance. The total growing stock in the region is almost 3300 million m^3.

Scots pine is native to all of the Nordic and Baltic countries except Denmark (Hultén 1950). Its growth on fertile sites is 7–9 m^3 ha^{-1} a^{-1} (Persson 1992), and on sites of medium fertility 3–5 m^3 ha^{-1} a^{-1}. Rotation periods are usually 70–90 years on fertile and medium-fertile sites and more than 100 years on poor sites (e.g. Persson 1992) (Table 2.2). The wood density of Scots pine is higher than that of

Norway spruce; i.e. 410–475 kg m^{-3} (Hakkila 1966; Peltola et al. 2007). Scots pine yields less harvest residues (branches, top parts of stem) than Norway spruce per unit area (Marklund 1988). This is due to the lower production and shorter life span of branches and foliage relative to stem in Scots pine compared with Norway spruce.

2.3.2.3 Silver and Downy Birches

Silver and downy birches are the dominant deciduous tree species in the Nordic and Baltic countries. The combined growing stock of both species is about 1450 million m^3. They often grow mixed with Scots pine and Norway spruce, but birch-dominated stands cover almost 8 million ha. The productivity of both species is the highest on nutrient-rich sites with sufficient availability of water, but silver birch is more successful on drier sites than downy birch (Rytter et al. 2008). Both birch species tolerate pH levels below 4 (Cameron 1996), making them usable on most forest and agricultural sites. Downy birch also grows well on nutrient rich peatlands drained for forestry.

Both birch species are native in all the Nordic and Baltic countries. They are pioneer species that prefer non-shaded conditions, and each tree needs a relatively large space to grow fast (Rytter et al. 2008). In the southern parts of the Nordic countries, the average growth of silver birch is 9–10 m^3 ha^{-1} a^{-1} (Niemistö 1996; Rytter 2004) over the 40–50 year rotation (Table 2.2), whereas in more northerly areas, the growth is 5–8 m^3 ha^{-1} a^{-1}. Birch wood is heavier than that of most conifers, with a basic density of 430–520 kg m^{-3} (Rytter 2004). Until now, birches have seldom been planted for energy biomass alone due to their high establishment costs. In general, energy biomass based on birch trees is a side-product from naturally regenerated young stands, which are thinned in conventional forestry operations.

2.3.2.4 Black and Grey Alder

Black alder is common in Denmark, the southern parts of Finland and Sweden and along the southern coast of Norway, whereas grey alder is not native to Denmark. Both alders are common in the Baltic countries, where they account for a growing stock of 170 million m^3 (Latvia Forest Industry Federation 2008; Keskkonateabe Keskus 2010; Directorate General of State Forests 2012). Their combined growing stock in Sweden and Finland is 120 million m^3 (Finnish Forest Institute 2011; Swedish Forest Agency 2012).

Black alder grows best on nutrient-rich soils with a generous water supply, and it withstands periodic flooding. Grey alder prefers similar sites, but it is more tolerant of shortage of nutrients and water (e.g. Rytter 2004). Both alders have the unique ability to fix atmospheric nitrogen in symbiosis with the actinomycete *Frankia*. They can fix up to 100 kg N ha^{-1} a^{-1} (Binkley 1981; Rytter 1996), which facilitates the maintenance of site productivity when harvesting nutrient-rich tree residues.

In dense young stands used in short-rotation forestry, the annual mean growth of black and grey alders can be over 15 m^3 ha^{-1} a^{-1}, but remains around 10 m^3 ha^{-1} a^{-1} when applying conventional management (Rytter 2004). In biomass-oriented cultivation, a rotation less than 30 years is feasible (Table 2.2), whereas a rotation of 40–50 years is used for black alder in conventional forestry. The basic density of alder wood is 350–370 kg m^{-3} (Rytter 2004). The red colour of alder wood makes it less attractive for pulping (Rytter 1998), and therefore logs of small diameter may be used for energy. While black alder mainly regenerates from stump sprouts, grey alder effectively produces root suckers (Rytter et al. 2000). This could potentially be exploited in cultivating of grey alder for biomass production.

2.3.2.5 Aspen

Aspen is common throughout the Nordic and the Baltic countries (Hultén 1950). It grows mainly mixed with other species, which makes it difficult to estimate the total coverage of aspen. For example, Stener (1998) found that almost 60 % of the aspen volume in Sweden was in mixed stands with Norway spruce and Scots pine. This is also why aspen cannot be regarded as an important species for biomass harvest from existing forests. The total growing stock of aspen in the Nordic and Baltic countries is over 150 million m^3. The growth of aspen is generally in the range of 7–10 m^3 ha^{-1} a^{-1} on suitable high fertility sites. The basic density of aspen wood is 350–400 kg m^{-3} (Rytter 2004).

2.3.2.6 Oak and Beech

Oak grows in Denmark, southern Sweden, along the southwest coast of Norway, in the Baltic countries and in southern Finland, where it is a rare and endangered species (Hultén 1950). Beech is of economic importance only in Denmark and in the southernmost parts of Sweden. The total growing stock of oak is 70 million m^3 and of beech 40 million m^3. The growth of oak on fertile sites is commonly 4–6 m^3 ha^{-1} a^{-1}, while the growth of beech is 5–8 m^3 ha^{-1} a^{-1} (e.g. Rytter 1998). The basic density of wood of both species is 575–580 kg m^{-3}. Oak and beech are not normally planted specifically to produce energy biomass, but the branches and top part of stems of both species are used for energy.

2.3.3 Non-Native Species

2.3.3.1 Lodgepole Pine

Lodgepole pine is the most widely used non-native tree species in the Nordic countries. It has primarily been used in northern Sweden. The species is native to the north-western parts of North America. The variety of lodgepole pine (variety *latifolia*) used in the Nordic countries comes from the northern inland areas of this region. In Sweden, lodgepole pine was introduced on a large scale in the 1970s, and plantations cover almost 600,000 ha (Elfving et al. 2001). In other Nordic countries, lodgepole pine has not been this popular, and in Finland, for example, the plantations cover only 9000 ha (Finnish Forest Research Institute 2012). Currently, the total stocking of lodgepole pine in Sweden is 30 million m^3 (Swedish Forest Agency 2012). Its growth is 36–50 % larger than that of Scots pine regardless of site fertility, while its wood density is about 3 % lower (Elfving et al. 2001).

Lodgepole pine grows successfully over a wide range of sites, but less so on moist and highly fertile sites. The survival rate of lodgepole pine is higher than that of Scots pine in the establishment phase. This is because lodgepole pine is less sensitive to low temperatures, and is browsed less by moose than Scots pine. Furthermore, lodgepole pine suffers less from snow blight (*Phacidium infestans*) and twist rust (*Melampsora pinitorqua*) than Scots pine. However, lodgepole pine is more sensitive to wind and snow damage, and to attacks by Scleroderris canker (*Gremmeniella abietina*) (Elfving et al. 2001).

2.3.3.2 Larch Species

Hybrid larch is probably the most useful larch in the Nordic and Baltic areas. It is a cross between the European (*L. decidua*) and Japanese (*L. kaempferi*) larches. Currently, this hybrid accounts for the majority of the 1.4 million m^3 of stem wood stocking of larches in Sweden (Swedish Forest Agency 2012). The annual mean growth of hybrid larch is about 13 $m^3\ ha^{-1}\ a^{-1}$ on fertile sites over a 35–40 year rotation (Ekö et al. 2004). Thus, the productivity of hybrid larch is similar to that of Norway spruce, but the rotation is shorter. Wood density of hybrid larch is 410–490 $kg\ m^{-3}$ (Karlman et al. 2005; volume determined at 5 % moisture). Hybrid larch is sensitive to root rot (Rönnberg and Vollbrecht 1999), and it is vulnerable to browsing animals (Frisk 2011).

Siberian larch is used only marginally in forestry in the Nordic and Baltic countries. In Finland, for example, there are about 30,000 ha of Siberian larch plantations (Lukkarinen et al. 2010). Growth of Siberian larch on fertile sites is 7–10 $m^3\ ha^{-1}\ a^{-1}$ during a fairly long rotation (Karlman 2010). The wood density of Siberian larch is 535–670 $kg\ m^{-3}$ (Karlman et al. 2005; see the hybrid above), and larch wood is commonly used outdoors due to its high resistance to rot and decay. Neither hybrid larch nor Siberian larch is species primarily grown to produce biomass for energy use.

2.3.3.3 Sitka Spruce

Sitka spruce is native to western North America, from Alaska in the north to California in the south. This species is likely best used in the maritime parts of the Nordic and Baltic countries. Sitka spruce is most common in Denmark and Norway, where it has been planted on about 50,000 ha (Øyen 2005; Vadla 2007). In Denmark, the plantations of Sitka spruce cover around 34,000 ha (Danmarks Statistik 2012), which represents about 7 % of the total forested area in the country. Sitka spruce grows more rapidly than Norway spruce, and its total growth is up to 40 % higher than that of Norway spruce. Growth of Sitka spruce in western Norway will peak at an age of 70–115 years at a level of 20–33 m^3 ha^{-1} a^{-1} (Øyen 2005), while the growth of Norway spruce under similar conditions is 12–24 m^3 ha^{-1} a^{-1}. The basic density of Sitka spruce wood falls in the range 325–390 kg m^{-3}, which is somewhat lower than that for Norway spruce (Vadla 2007). Sitka spruce resembles Norway spruce in many respects. Its wood could be used for the same purposes and it will most probably be treated like spruce in terms of biomass production for energy.

2.3.3.4 Douglas Fir

There are two major subspecies of Douglas fir; i.e. the costal and interior ones. The coastal Douglas fir is found in northern British Columbia and along the Rocky Mountains in California. The interior Douglas fir is native to the eastern Rocky Mountains through Montana down to Mexico. The interior Douglas fir is preferable in the southern parts of the Nordic and Baltic region, where it is used in forest cultivations due its resistance to a harsh climate. However, the coastal Douglas fir is so far the more widely used subspecies in the Nordic and Baltic countries (Svensson 2011), even though frost damage is common. At present, Douglas fir plantations cover only 500 ha in Finland (Metla 2011), and they account for around 1 % ($\sim$ 5,000 ha) of the forested area in Denmark (Nord-Larsen et al. 2009). Growth of Douglas fir is probably superior to that of Norway spruce, and in Denmark its average annual growth is expected to be 20 m^3 $ha^{-1}a^{-1}$ (Henriksen 1988). Douglas fir is usually cultivated for the production of high quality timber, but tops and branches could be used for energy generation.

2.3.3.5 Grand Fir

Grand fir is only sparsely used in northern Europe, and its growth in these conditions is poorly known. Grand fir is most widely planted in Denmark, where it covers approximately 3,000 ha of forest land (Bergstedt and Jørgensen 1992). Under these conditions, its annual mean growth is 25–30 m^3 ha^{-1} a^{-1} (Bergstedt 2005) over a rotation of 50 years; i.e. the yields are 65–70 % higher than those achieved by Norway spruce. Furthermore, Grand fir seems to be less sensitive to root rot (*Heterobasidion* spp.) than Norway spruce (Swedjemark and Stenlid 1995). Grand fir can grow on a fairly wide range of site conditions. It is a secondary species with relatively high light demands capable of growing in multi-layered stands. Establishment can be tricky because the plants are sensitive to handling, low temperatures and browsing.

2.3.3.6 Poplars and Hybrid Aspen

Use of poplars (*Populus* species) in forestry is relatively new in the Nordic and Baltic countries, although poplars have been used in landscaping and shelter belts for a long time. Poplars belonging to the section balsam poplars (Tacamaha) seem best suited to the Nordic conditions. At present, about 5,000 ha of land has been planted with poplars, including hybrid aspen (Rytter et al. 2011a; Tullus et al. 2012). All poplars are highly productive and should be used on fertile sites. For example, the growth of hybrid aspen is over 20 m^3 ha^{-1} a^{-1} in 20–25 year rotations (Rytter and Stener 2005; Tullus et al. 2012). Regarding other poplars, there is less information available, but in the Nordic and Baltic countries the growth of some other poplars will probably be somewhat higher than that of hybrid aspen (e.g. Stener 2010; Rytter et al. 2011a). The wood of poplars is relatively light; i.e. the basic density is 300–420 kg m^{-3} for hybrid aspen and 300–390 kg m^{-3} for balsam poplars depending on species, clone and age (Rytter 2004; Stener 2010).

Currently, hybrid aspen is a most promising candidate for the effective supply of energy biomass. Hybrid aspen is the hybrid of European aspen and trembling aspen from North America. It is well adapted to the Nordic and Baltic conditions, because both parent species have boreal distributions. Hybrid aspen produces root suckers after the final felling, whereas other poplars mainly regenerate via stump sprouts. The root sucker stands of hybrid aspen quickly produce large amounts of biomass. In a few years, the average growth may reach 10 Mg ha^{-1} a^{-1} (about 30 m^3 ha^{-1} a^{-1}) (Rytter 2006; Tullus et al. 2012). Regarding the use of stump sprouts in poplar regeneration, some clones sprout vigorously while others are less inclined to sprout (McCarthy and Rytter 2012). Therefore, the natural regeneration of poplars is still an unreliable way to establish new poplar plantations, until the clonal performance of sprouting is better understood.

2.4 Potential to Enhance Biomass Supply Through Tree Breeding

2.4.1 Breeding Practices

2.4.1.1 Objectives of Breeding

Tree breeding refers to the genetic improvement of tree populations in order to enhance their survival, growth and wood properties by making use of the genetic variability (diversity) of trees and their ability to inherit specific traits. Breeding can be divided into long- and short-term breeding.

Long-term breeding combines intensive breeding, gene conservation and preparedness for future climatic changes (Danell 1993). Within species, the material is divided into multiple breeding populations (MPBS), where crossings, testing and

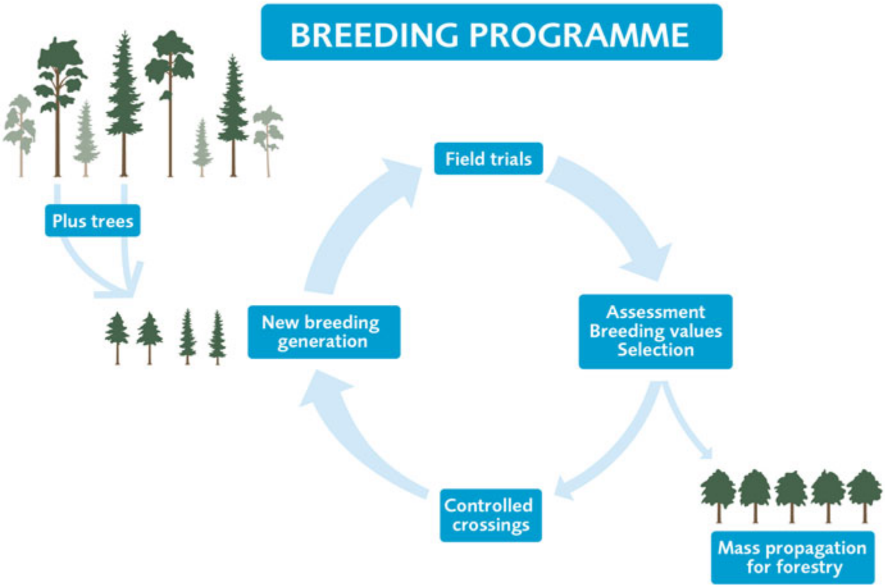

Fig. 2.2 Schematic outline of a long-term breeding program

selections for the next breeding generation are carried out (Fig. 2.2). This is repeated over several generations. The genetic gain in growth per breeding generation is 10–15 %. Breeding is performed in close cooperation with mass propagation, where the deployment of improved material across the plantation area is optimized. Long-term breeding strategies are planned to provide sustainable gains over the generations in order to produce regeneration material with high performance and a sufficient level of genetic variation.

Short-term breeding is an option for species of less importance. In such cases, the emphasis is on producing the maximum possible genetic gains over a short period of time. Sufficient genetic variation for commercial use is retained, but no attention is paid to long-term gene management. Short-term breeding normally involves phenotypic selection of plus-trees, field testing and finally selection of the best genotypes for commercial use.

2.4.1.2 Breeding for Enhancing Supply of Energy Biomass

To date, breeding programs for traditional forest purposes have focused mainly on traits of importance for the production of timber and pulp, and specific traits important for bioenergy production have received much less attention.

In parallel with the increasing demand for renewable energy sources, there is a growing interest in short-rotation forestry crops in many countries (Dickman 2006). The willow coppice is the most well known management system using short-rotation forestry in northern Europe, but the use of poplar plantations based on the *Salix* strategy is increasing (Weih 2004). Poplars are also popular elsewhere in Europe, e.g. in Italy poplar clones used in traditional cultivations for producing timber for veneer have been used in short-rotation forestry trials (Paris et al. 2011). However, many of these clones were not suitable as short-rotation forestry crops. Breeding traits important for using poplars in short-rotation forestry include:

1. high rooting capacity in field conditions when applying direct planting of cuttings in establishing plantations;
2. high survival of sprouts after frequent short rotation coppicing;
3. ability to tolerate high plant densities;
4. continuous high biomass production after repeated coppicing; and
5. stem properties (dimensions) suitable for mechanical harvesting.

A combination of high yielding clones and appropriate management is essential for a good economic outcome. Therefore, the testing of clones under different management regimes is recommended, e.g. some clones produce more biomass with non-coppicing management than they do under a coppice-based management regime (Afas et al. 2008; Ferm and Kauppi 1990).

Regarding non-native species such as poplars, Douglas fir, Grand fir, Sitka spruce and lodgepole pine, the climatic adaptation is particularly important, because the non-native species have not adapted naturally to the climates of northern European countries. In this respect, the optimal timings of budburst and cessation of growth are crucial in the boreal conditions. The main factors triggering phenological processes are temperature (e.g. the heat sum) in the spring and the photoperiod in the autumn (Hannerz 1998). Seedlings of a more southern origin often have a higher growth than seedlings of local origin, provided that extreme climatic events are avoided.

Pathogen resistance is another important selection trait in breeding (Newcombe 1996; Yanchuk and Allard 2009). The genetic impacts of such traits are primarily evaluated based on field performance. Furthermore, inoculation tests for stem canker and root rot, and laboratory procedures for testing leaf rust resistance in poplars (e.g. Steenackers et al. 1990; Swedjemark and Stenlid 1995), can be used in breeding.

Wood property traits, such as basic density and lignin content, can be of great importance when the supply of biomass for energy use is the preferred aim in breeding. Regarding the combustion or gasification of poplar, for example, it would be desirable to increase the density and lignin content of the wood, whereas a lower lignin content may be desired in ethanol production. There is considerable scope for selecting clones specifically for bioenergy production (Dinus et al. 2001; Poke et al. 2006).

2.4.1.3 Breeding for the Future Climate

In northern Europe, global warming may increase the annual mean temperature up to 3–5 °C by the end of this century (IPCC 2007). This would increase the growing season by 30–40 days in the northern parts and by 100 days in the southern parts

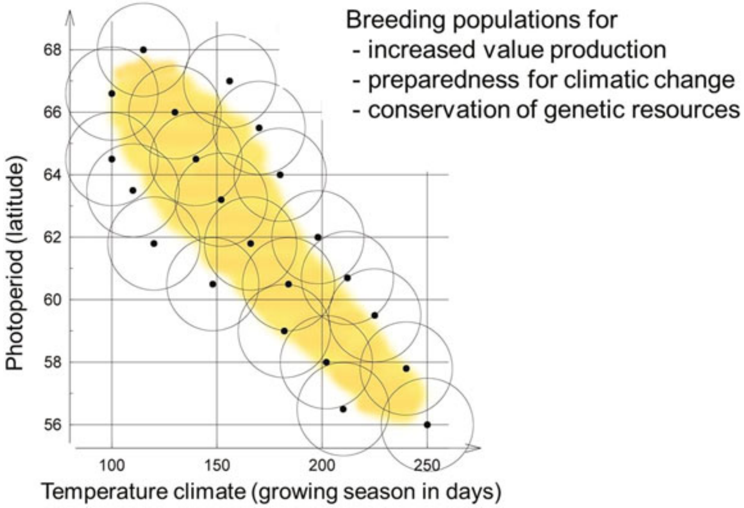

Fig. 2.3 Breeding populations of Norway spruce in Sweden are distributed across defined climatic gradients of light and temperature

(Brázdil et al. 2010; Jönsson and Bärring 2011). As a consequence, growth is estimated to increase by 20–40 %. On the other hand, there will be increasing risks of frost damage in the spring and early summer in the southern regions. Furthermore, the damages due to pathogens and pests may increase in a warmer climate.

In a very long-term scenario, trees will adapt to the changing climate by natural selection. However, adaptation occurs in response to the current conditions, and there is no way for natural adaptation to future climates to occur with the same efficiency as is possible in selective breeding. In this respect, the Multiple Population Breeding Strategy (MPBS) may provide preparedness for future climate change. In this strategy, the breeding population is divided into different sub-populations, each of which is bred against different adaption targets defined by light and temperature gradients (Danell 1993, Fig. 2.3). In the long run, each population will gradually adapt to the climate profile designed for it. In the short term, it will be possible to introduce plant material representing populations adapted to southerly climates to more northerly regions. At the same time, uncertainties regarding future climatic changes can be addressed by increasing the plasticity and adaptability of existing populations using field tests at multiple locations covering a wide range of climatic conditions.

2.4.1.4 Mass Propagation of Plant Material Improved in Breeding

Gains from breeding can only be realized when bred material is deployed to tree plantations. Currently, most improved material is produced in seed orchards (Fig. 2.4), which have been established for all the major species and some of the less common

Fig. 2.4 Seed orchard of Norway spruce at Ålbrunna, Sweden. (Photo: Olle Rosenberg)

species throughout the Nordic and Baltic countries. Seed orchards are usually established for producing open-pollinated seeds. However, it is possible to produce seeds using more controlled pollination, for example, when seed orchards are established under cover in greenhouses, as for silver birch in Finland and Sweden.

The main disadvantage of seed orchards is the long time before flowering and initiation of seed production. In fact, genetic gains from breeding can be realized more rapidly by using vegetative propagation methods such as rooted cuttings, tissue cultures or somatic embryogenesis (SE) in multiplying known plant materials. These methods can be used with either scarce seed lots (e.g. controlled crossings) or tested clones. Cutting is an inexpensive way to use vegetative propagation for species that regenerate through stem sprouts and root suckers. Furthermore, tissue culture can be used in producing large numbers of plants from a small initial quantity of plant material.

Box 2.1 Biotechnology and Breeding

Genetic markers for the identification of genetic relationships are widely used in breeding, e.g. in clonal verification in seed orchards, clone archives etc.

Indirect selection based on data from genetic markers has considerable potential for increasing the cost-efficiency of breeding; i.e. it could reduce the duration of breeding cycles, decrease costs by reducing the need for field testing, and increase the intensity of selection. On the other hand, the most important traits such as yield, adaptability and wood quality are complex and difficult to link to major genes. In addition, genes interact with each other and the environment; i.e. genes that sometimes seem to be important may not be significant for all trees or in different environments. Marker-assisted selection (MAS), based on quantitative trait loci (QTL) analysis, has long been a promising method but has not been able to identify specific genes for complex traits in forest trees with the same effectiveness as has been achieved in model systems (Neale and Kremer 2011). New techniques such as association mapping and genome-wide mapping seem to have a great potential for establishing relationships between phenotype and large number of gene markers.

Genetically modified organisms (GMOs) are another product of biotech methods. GMO crops are commonly used in agriculture, mainly due to their resistance to herbicides and insecticides. However, the development of GMO trees for forestry has been slow, and the application of GMO in forest tree breeding is still some way off. In applying GMO techniques in breeding, the stability of gene expression must be verified in long-term field trials and there must be no negative side effects. In addition, obtaining a licence for utilization of a GMO crop is a time-consuming, expensive and unpredictable process. Furthermore, GMOs are often perceived negatively by the general public. Consequently, GMOs are not expected to have a significant impact on operational tree breeding in the near future.

2.4.2 Breeding Gain in Selected Coniferous Species for Supply of Energy Biomass

2.4.2.1 Norway Spruce

Experiments in Sweden indicate that the use of seedlings originating from first-round (-generation) seed orchards increases the growth by about 10 % relative to seedlings based on no breeding (Rosvall et al. 2001) (Table 2.3). The gains in volume production of stem wood from the second round of seed orchards range from 10 % (phenotypically selected plus-trees) to 25 % (intense selection from tested plus-trees). Gains of up to 25 % are anticipated from the third round of seed orchards, which are currently being established using the best plus-trees from the second-generation orchards.

2.4.2.2 Scots Pine

Breeding programs for Scots pine have been on-going in many northern-European countries for several decades. The focus of these programs has been mainly on traits that are relevant to the traditional industrial uses of wood, i.e. the production of high quality logs for sawmills and the pulp industry. In breeding of Scots pine, there is increasing interest in the content of resin acid, which is the main component of crude pine oil usable in producing biodiesel. Resin acid content seems to be highly genetically controlled (Ericsson et al. 2001). It is also strongly genetically correlated with the content of phenolic compounds, which are essential for resistance to decay (Ericsson et al. 2001) and to biotic and abiotic stress factors (Chong et al. 2009).

2.4.2.3 Lodgepole Pine

An intensive breeding program for lodgepole pine exists for the northern part of Sweden (north of latitude 60) (Ericsson 1994). Currently, seed orchards produce improved seeds sufficient to satisfy the needs of the annual planting area. The breeding objectives of the program are primarily aimed at timber production, but high growth of this species using improved seeds is enhancing the production of biomass for energy use. The growth of unimproved lodgepole pine is 30–40 % greater than that of Scots pine (Elfving and Norgren 1993). The existing seed orchards provide a further genetic gain, which will increase the growth of lodgepole pine by 10 %.

2.4.2.4 Larch

The hybrid larch has received some attention in breeding in Denmark and southern Sweden. This is because its juvenile growth is fast, the proportion of heartwood is high (which makes it decay-resistant) and the basic density of the wood is high. In breeding, the growth and straightness of stem have been the major selection criteria. Adaptation traits, such as spring and autumn phenology, are also important, because larch can be severely damaged by frost. In addition, European larch is quite susceptible to canker (*Lachnellula willkommii*), whereas Japanese larch is resistant as is mostly the case for the hybrid. The properties of larch wood have received little attention in breeding until recently, since larch wood has inherently good properties.

2.4.2.5 Sitka Spruce

Great Britain runs the most ambitious tree improvement program for Sitka spruce in Europe. The most important economic traits in the British breeding program are volume production, stem form and wood density (Lee 2001). Lee and Matthews (2004) reported genetic gains in the volume growth of 20 % relative to the unimproved provenance material from Queen Charlotte Island (QCI). In Denmark, Hansen and Roulund (2011) found that the improved plant material will produce 30–50 % more dry matter than the unimproved alternative; i.e. an annual mean growth exceeding 20 Mg ha^{-1} a^{-1} of above-ground dry matter during a 40-year rotation. However, there is a strong negative genetic correlation between growth capacity and wood density (Lee 1999), which has to be considered when selecting material for mass propagation. Furthermore, the green spruce aphid (*Elatobium abietinum*) can reduce growth in Sitka spruce plantations, but it is possible to increase the resistance to this insect by breeding (Jensen et al. 1997).

2.4.2.6 Douglas Fir

Douglas fir has a large genetic variation across several traits between and within provenances and stands. In using Douglas fir in northern Europe, the most serious problem is its sensitivity to frost in the early phase of growth and development. However, St Clair (2006) found a large variation among populations with respect to autumn hardiness. It is evident that the variation within populations is sufficiently large to be exploited in selection. Hansen (2007) showed further that there are clear differences among seed sources as regards frost damage. Combined provenance and progeny trials based on the seeds collected from British Columbia show that survival is higher among interior provenances than among the coastal provenances (Martinsson 1985). However, even among coastal material the survival and growth are adequate. Regarding biomass production, the breeding gains are not known as far as the Nordic and Baltic countries are concerned.

2.4.3 Breeding Gain in Selected Deciduous Species for Supply of Energy Biomass

2.4.3.1 Birch

The most comprehensive breeding program for birch is in Finland, where birch breeding has continued since the 1960s (Koski and Rousi 2005). The primary focus is on silver birch (Fig. 2.5), but downy birch is bred too. The main selection criteria are the yield and stem quality (including straightness, thickness and number of branches, forking and number of ramicorns), with the aim to produce high quality timber. Climatic adaptation (spring and autumn phenology) is also high on the breeding agenda. Mass propagation of improved birch seeds is based on greenhouse nurseries. Genetic gains of 30 % with respect to yield have been obtained, along with improvements in stem quality, relative to unimproved plant materials (Hagquist and Hahl 1998).

2.4.3.2 Alder

Regarding supply of energy biomass, grey alder has received more attention than black alder in the Nordic and Baltic countries. According to Matthews (1987), the

Fig. 2.5 Progeny trial of silver birch in Järpås, Sweden, at the age of 15 years. (Photo: Lars-Göran Stener)

breeding potential of grey alder is high due to early flowering, high seed production and easy vegetative propagation, but very limited genetic improvements have been obtained to date. Studies on black alder show that there is substantial genetic variation within and among populations (stands) (Ljunger 1972). The selection effect is probably high, as shown in an experiment where the growth increased by 18 % after a 10 % selection from the total population of 125 plus-tree clones (Stener 2007). The gain is of the same magnitude as that of silver birch (Stener and Jansson 2005). It is most likely that grey alder has a similar potential for breeding as black alder. The hybrid between black alder and red alder (*A. rubra*) has shown a 50 % higher growth compared with black alder (Stener 2007).

2.4.3.3 Poplars and Aspen

A number of species and their hybrids within the *Populus* genus are among the highest yielding trees adapted to northern-European climatic conditions. Except European aspen, these poplars are introduced species. The poplar species most likely to be useful in northern Europe are *P. tremula*, *P. tremuloides*, *P. trichocarpa*, *P. maximowiczii*, *P. deltoides* and *P. nigra*. Inter-specific hybridization in combination with reciprocal recurrent selection of the parent species is frequently used in long-term breeding (Li et al. 1993; Bisoffi and Gullberg 1996) (Fig. 2.6). Backcrossing is a more short-term approach that is used for resistance breeding, such as increasing *Melampsora* leaf rust resistance by backcrossing *P.* × generosa F1-hybrids with *P. deltoides* (Pinon et al. 2006). Poplars are susceptible to a large number of pathogens, and this susceptibility is often under strong genetic control (e.g. Newcombe et al. 2001). At present, the genetic gain possible in poplars and aspens for the supply of energy biomass is poorly known.

Fig. 2.6 Poplars aged 18 years in a trial in southern Sweden. (Photo: Lars-Göran Stener)

2.5 Regeneration and Establishment of Tree Plantations for Supply of Energy Biomass

2.5.1 Factors Affecting Regeneration Success

The successful establishment of tree plantations is the critical phase in producing biomass for energy use. In short-rotation forestry, excess mortality of seedlings and delayed canopy closure may detract from the potential benefits of the fast early growth of selected tree species and intensive management for the production of biomass for energy use. The same holds for conventional forestry (long-rotation forestry), where energy biomass is produced in integration with timber production. Tree species and genotype choice, site properties (edaphic and climatic conditions), pests, competing vegetation and site preparation (soil management), planting material, and regeneration procedures are factors that can have an effect on successful regeneration in forestry utilizing either short or long rotation.

2.5.2 Choice of Tree Species and Genotypes

Choice of tree species is affected by the expected growth and the resistance to climatic variability and to possible attacks of insects and pests. In this context, the available seedlings may represent varying provenances or clones of the same species. All the European countries have specific laws governing forest management, which must be considered before planting. For example, there may be restrictions on the maximum or minimum number of clones per plantation when using vegetative material. Similarly, the scope for using non-native species may be restricted by national regulations or by forest certification organizations such as the Forest Stewardship Council (FSC) or Programme for the Endorsement of Forest Certification (PEFC).

The laws governing planting on forest land are generally different to those covering agricultural land. For example, in Sweden, land that is planted with *Salix* or *Populus* species whose wood will be used for energy production within the next 10 or 20 years is classed as agricultural land. However, if other tree species are used, the land would be treated as forest land. Such laws will differ from country to country and it is therefore essential to be aware of the details of laws in the country of interest and to check them prior to planting. Finally, the use of specific varieties may be restricted by breeders' rights (www.cpvo.europa.eu, the date of access May 28, 2013).

2.5.3 Site Properties and Site Preparation

2.5.3.1 Soil Moisture and Nutrient Availability

The environmental conditions on open sites to be reforested may deviate substantially from those prevailing on sites occupied by trees forming a closed canopy. When the tree canopy is removed, the amount of solar radiation will increase drastically on the soil surface, with a large impact on the energy balance on the site. Furthermore, both the hydrology and the nutrient status are changed, with the increased supply of water and nutrients for regeneration and growth (Grossnickle 2000). On open sites, there is a large fluctuation in air temperature between night and day, more rainfall reaches the ground, and the evaporative demand and vapour pressure deficit (VPD) is higher than on forested sites (Fig. 2.7). These rapid changes and new climatic conditions can cause drought, excessive soil water and strong winds, which affect the establishment and growth of seedlings.

Soil texture, topography and the position of the site in the landscape are some of the factors that affect soil moisture and nutrient availability. After harvesting, the rainfall previously intercepted in the canopy reaches the soil surface, which may lead to excessive soil moisture with oxygen deficiency in the rooting zone. On the other hand, the availability of water in the rooting zone of seedlings may be low due to the uptake in competing vegetation. Therefore, it is important to identify microsites where water availability is relatively stable and sufficient for seedlings to become established. Scarification is among the methods used to improve soil

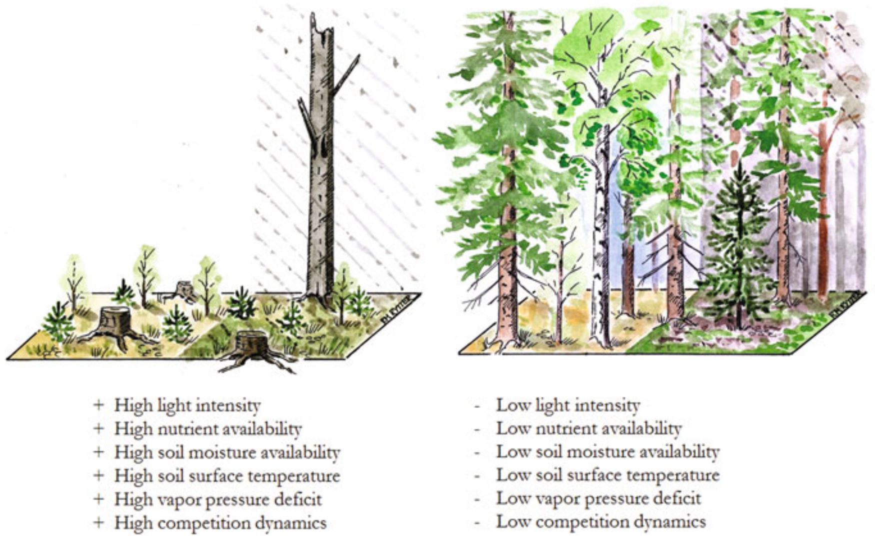

Fig. 2.7 Environmental conditions on an open site and under a closed forest canopy. (Illustration: Rose-Marie Rytter)

moisture conditions in the rooting zone. On sites where excessive water can be a problem, ditching may be necessary.

Nutrient availability generally increases in the site after clear cut. Although a lot of nutrients are removed with trees during harvest, the new environmental conditions increase the turnover of soil organic matter and the mineralization of nutrients on the forest floor. Fast-growing deciduous trees prefer lower acidity than most conifers. In Norway spruce forests, the pH is usually around 4–5 (Högbom et al. 2002), which is too low for poplars or hybrid aspen (Rytter et al. 2011b). Liming or ash deposition may be necessary to reduce the acidity if tree species with a preference for low acidity are to be planted.

2.5.3.2 Frost and Soil Temperature

Rapid changes in temperature that result in frosts can cause damage to the seedlings during both spring and autumn. Seedlings are most sensitive to frost damage during shoot elongation, which is the reason why species with late flushing should be used on frost-prone sites (Langvall et al. 2000). During autumn, frost may damage seedlings that are not fully hardened. The fluctuation in temperature can be reduced by shelter trees, thus reducing the risk of damage on frost-prone sites. To reduce the damage caused by climatic factors, it is important to identify the specific conditions that may appear on the site and to apply appropriate management measures to limit the harm caused by any restrictive factors.

Fig. 2.8 Rapid changes in the vegetation dynamics occur at a regeneration site. This site has been occupied by weed. (Photo: Karin Johansson)

2.5.3.3 Competing Vegetation and Site Preparation

Competition from both herbaceous and woody species is among the greatest problems in establishing tree plantations (Fig. 2.8). Such vegetation competes for light, water and nutrients, and can also damage the crop seedlings mechanically. Furthermore, competing vegetation provides habitats for rodents, insects and fungi, which can damage seedlings. Therefore, the suppression of competing vegetation, mechanically or chemically or by a combination of both is needed to prepare the planting.

2.5.4 Planting Material

The performance of seedlings depends on their genetics and morphological and physiological characteristics, and on the conditions at the site (Grossnickle 2000). In general, seedlings with a large amount of active root tips and a balanced ratio between root and shoot systems are needed for successful regeneration. Furthermore, height, diameter, nutrient status and bud size are other important traits for seedling establishment. Those traits reflect sturdiness and vitality of the seedlings and ameliorate high initial growth and ability to compete with suppressing vegetation and to avoid excess herbivory due to the vertebrates and invertebrates common on regeneration areas. Specific features of different types of seedlings are summarized in Box 2.2.

Box 2.2 Some main properties of seedlings currently used in forestry and tree plantations in the Nordic and Baltic countries

Containerized seedlings are grown in containers, usually in green houses but also outdoors. The goal is to grow a seedling that is well balanced regarding the relation between root and shoot and that has traits that are desirable for the planting site. The containers are often designed to promote a root system with many new and active root tips, ready for rapid establishment in the field. The method is applicable for most tree species.

Bare rooted seedlings are produced in seed beds outdoors. Seeds are usually sown at a high density and during the growth period the seedlings are transplanted a couple of times depending on species and growth rate. Transplanting is done to stimulate growth, but also to remove poor individuals. Prior to transplanting, the seedlings are root-pruned to facilitate the transplanting. Root pruning is also done during the growing season in order to get a well balanced root system.

Hybrid seedlings are used especially on sites where competing vegetation, pine weevil, frost or browsing may damage seedlings. Seedlings of this type are grown in containers for about the first ten weeks, and thereafter they are transplanted to nursery fields. This production method combines the advantages of containerized seedlings and bare rooted seedlings and creates a large seedling with a well balanced root system.

Cuttings are vegetatively propagated material (i.e. clones), where a piece of stem, twig or root is planted in a nursery or directly in the field. All plants originating from tissues of the same individual are genetically identical. Cuttings are usually produced as containerized plants, but the cuttings of some species (such as poplars and willows) can be stuck directly into soil. Poplars are almost always generated through cuttings.

Somatic embryogenesis is another method of cloning, in which new embryos can be mass produced from a single seed and the method has been used for instance in Norway spruce. Similarly, tissue culture methods can be used to produce plants from leaves, shoots or buds. This approach is common when working with hybrid aspen.

2.5.5 Regeneration Procedures

2.5.5.1 Planting

In general, planting is done by hand, but machine planting is slowly increasing and new equipment is being developed (Fig. 2.9). The spacing used in planting differs between tree species (Table 2.2). Seedlings can be planted throughout the whole

Fig. 2.9 Planting can be done by hand with a planting tube or by machine. (Photo: Erik Viklund and Karin Johansson)

growing season, but the planting in spring is the most successful due to sufficiently moist soils, low VPD and dormancy of seedlings during the planting operation. Later in the summer, dry periods and high VPD may reduce the success of plant establishment. The seedlings are also in active growth during this period, which makes them more sensitive to both physiological and mechanical stress. In the autumn, dormant seedlings can be planted, but frost-heaving and low soil temperatures may affect the success of plants becoming established.

2.5.5.2 Natural Regeneration

Some species are relatively easy to regenerate naturally by seeds, root suckers and/or stump sprouts. Hybrid aspen, for example, has the ability to produce root suckers and after clear cutting there could be up to 50,000–100,000 suckers per ha (Fig. 2.10). In producing energy biomass, the plantation could potentially be harvested continuously at an interval of 4–5 years, as shown by the experiments on some *Populus* and *Salix* species. For poplars, stump sprouts are common also for the second generation, allowing harvesting after 4–8 years (Rytter et al. 2011b). However, further research is needed to identify optimal management regimes for second rotation of fast-growing deciduous species.

Fig. 2.10 A two-year-old second-rotation stand of hybrid aspen cleaned with strip rows. (Photo: Lars Rytter)

Natural regeneration through seeding is common for many tree species. At the time of harvest, seed trees are left on the site enabling them both to produce seed and to shelter the next generation. Soil scarification will substantially increase the likelihood of successful regeneration (Karlsson and Örlander 2000). For example, birch species regenerate easily on sites with disturbed soil surface (Lehtosalo et al. 2010) and can be used as shelter trees to protect the planted seedlings, and thereafter harvested for energy biomass early in the rotation.

2.6 Conclusions

In the future, the selection of appropriate tree species, the ongoing progress of breeding programs, and the development of efficient methods for cultivation of tree plantations will increase the potential and sustainability of forests for energy biomass production in northern Europe. However, to achieve optimal results, it will be necessary to combine the knowledge from these three areas with respect to the establishment, growth and development of tree plantations. Regarding short-rotation forestry, our knowledge is still very much based on the research and experiences from conventional forestry with the tree species native to the Nordic and Baltic countries. There is a need for more research on the biomass potential of the different tree species in short-rotation forestry, including species both native and exotic in northern Europe.

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Chapter 3
Short-rotation Forestry for Supplying Biomass for Energy Production

Hardi Tullus, Arvo Tullus and Lars Rytter

Abstract In this chapter, we discuss the opportunities that short-rotation forestry may provide to meet the increasing demand for energy based on renewable resources. We present information on suitable species for northern Europe, their productivity, establishment and management. In this region, grey alder, hybrid aspen, poplars and willows are the most promising species for short-rotation forestry. The productivity of these species is around 5–12 Mg ha^{-1} a^{-1} in fertile sites. All of these species regenerate vegetatively, and they can be managed in several consecutive rotations without needing to repeat the establishment between rotations. No major negative environmental impacts have been found with the cultivation of these species. This is especially the case when plantations are established on abandoned agricultural land or otherwise degraded land.

Keywords Agro-forestry · Alder · Aspen · Bioenergy · Coppice system · Energy forestry · Management · Northern Europe · Plantation forestry · Planted forests · Poplar · Short-rotation forestry · Willow

3.1 Short-rotation Forestry in Supplying Energy Biomass—Why?

Two contradictory trends characterize modern forestry: (1) to increase the area of forests protected for biological conservation, and (2) to increase the land area under intensively managed plantations for producing biomass. In the current chapter, we

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focus mainly on the Nordic and Baltic countries, where forestry is traditionally based on long rotation periods (commonly 60 to 120 years) and where short-rotation forestry (SRF) with rotation periods of less than 30 years is still emerging (Weih 2004; Aosaar et al. 2012; Tullus et al. 2012a). The rapid increase of short-rotation plantation forestry is an important trend throughout the world. Globally, 6–7 % of the forests are planted (FAO 2010) but they provide around a quarter of the world's annual roundwood production and this share is predicted to increase considerably in the future (Siry et al. 2005).

There are two main reasons why SRF has received increasing attention during the last three decades in northern Europe. The first reason is that the agricultural use of land has decreased in several countries due to political, economic and social changes (Astover et al. 2006; Alcantara et al. 2012). In this case, SRF is an attractive land use justified both by economic and environmental reasons. The second reason is that that we need to substitute fossil fuels with renewable energy sources, a commitment of the member states of the European Union (EU). According to the EU energy policy, the share of energy from renewable sources is to increase to 20 % by the year 2020 (Directive 2009/28/EC). Increasing production of woody biomass in SRF plantations is a promising means to reach this goal. In this chapter, we discuss the general outlines of SRF and its potential application in the Nordic and Baltic countries. The focus is mainly on the most promising tree species and their management.

3.2 Management Concepts Used in Intensive Production of Woody Biomass

3.2.1 Coppice Systems and Agro-forestry

To start, we need to clarify management (silvicultural) concepts to characterize SRF at northern latitudes. In this regard, *coppice system (low forest)* refers to forests originating from root or stump sprouts (suckers), which restricts this concept to broadleaved trees. Historically, the main aim of the coppice system has been to provide forage and fuel wood. The opposite to low forest, *high forest* is used to refer to seed-originating tree stands managed for timber. High forest may represent naturally or artificially (sown, planted) regenerated coniferous or deciduous stands.

In a coppice forest, trees are cut at the stump level or branches are cut leaving 2–3 m standing stems to protect new sprouts against browsing by cattle. Nowadays, this method is no longer important in producing fuel wood or forage. However, it is still used by farmers in the Mediterranean region, where sparsely spaced trees are grown in grassland grazed by livestock. Usually, some branches are cut from living crowns to stimulate the flowering and fertility of these trees (e.g. oaks, olives, chestnuts, stone pines) as is also done to enhance apple crops in northern Europe. A land use that is simultaneously for forestry and agriculture is also known as *agro-forestry*. Traditional rotation in coppicing is about 20 years.

In the coppice system, clear cut can be used in final felling or some trees may be left to grow to larger dimensions. This system is called *coppice selection* in order to distinguish it from *coppices with standards*. The latter means that vegetative and seed-originating trees are grown together and selectively cut by leaving some trees to grow older and provide large specimens. In general, the short-rotation stands and plantations of willow, grey alder and hybrid aspen can be regarded as modern examples of low forests, although grey alder and *Populus* species will grow to large dimensions.

3.2.2 Plantation Forestry and Forest Plantations

Plantation forestry (silviculture) is a term that refers to the intensive production of woody biomass in forest plantations. A forest plantation is an artificially established set of trees (as an act of plantation forestry) on previous agricultural land, on otherwise artificially degraded land, or on forest land after clear felling. The main aim of plantation forestry is to maximize the production of wood or some other products of trees (e.g. fruits, leaves, bark, resin, natural rubber), while less attention is paid to the creation of a system resembling a real forest ecosystem. In international forestry terminology (as recognized by the FAO), the term plantation forestry incorporates all man-made (primarily planted) forests and management of such forests (FAO 2006a). In 2010 the FAO introduced a broader term, *planted forests*, instead of *forest plantations*. Planted forests comprise productive and protective forest plantations and planted components of semi-natural forests. Globally, planted forests cover 264 million ha, which constitutes 6.6 % of the forest area, whereas in the last decade the planted forest area has increased by an average of almost 5 million ha every year (FAO 2010).

3.2.3 Short-rotation Forestry and Energy Forestry

Short-rotation forestry (SRF) utilizes much shorter harvesting intervals and rotations than traditional forestry (high forestry) and therefore uses species with high productivity and early growth culmination. In general, the final harvesting in SRF is done after the culmination of mean annual increment (MAI), which coincides with the financial maturity. In boreal conditions, the rotation in SRF is about 30 years, whereas in the tropics the rotation could be as short as 5–7 years (FAO 2006b). SRF may incorporate high initial stand densities, is often monocultural, and requires optimal or close-to-optimal site conditions for the used species for an economically successful outcome.

Energy plantations (energy forestry) became well-known in the 1980s, when the first willow plantations for energy use were established in Sweden (e.g. Christersson et al. 1993). Energy plantations are usually managed on a rotation of 3–5 years. The main advantage of producing energy from woody biomass compared with herbaceous biomass (e.g. combustion of seeds or whole above-ground part of the crops) is that woody biomass is mainly composed of carbon, oxygen and hydrogen. Furthermore, the ash content of woody biomass is 5–10 times lower than that of herbaceous biomass. Concentrations of nutrients (N, P, K and others) in wood and bark are also lower, which reduces the amount of nutrients removed in harvesting, thus reducing the need to compensate nutrient losses through fertilization. However, bark has higher ash content than wood and thus woody biomass with a higher share of wood and a lower share of bark is generally preferred. Furthermore, energy produced from woody biomass is considered to be carbon neutral; i.e. the amount of carbon released at combustion equals the amount of carbon fixed in trees during their life-time. This is probably true in the long run, if indirect carbon emissions (e.g. emissions from fossil fuels used in management and logistics, and emissions due to the change of land use) are excluded. The wood of potential energy tree species has comparatively low density and the fuel value per volume is low, but calculated on a weight basis differences between different tree species are small (Nurmi 1993). A more important factor to consider is the moisture content of the actual biomass fraction, which depends on the species as well as harvest occasion and storage time.

3.3 Species Suitable in Short-Rotation Forestry

3.3.1 General Characterization

A suitable tree species for SRF in the boreal climate should combine rapid growth with high frost hardiness. Regarding these requirements, *alders* (grey and black alder), *aspens* (common and hybrid aspen), *poplars* and *willows* have a high potential to be used in SRF in northern Europe (Karačić et al. 2003; Rytter 2004; Weih 2004; Rytter and Verwijst 2009; Aosaar et al. 2012; Tullus et al. 2012a, Table 3.1, Fig. 3.1).

Furthermore, there is evidence that deciduous tree plantations on agricultural or forest lands have no negative effect on the site fertility.

3.3.2 Poplars, Aspens, Alders and Willows for Short-rotation Forestry

3.3.2.1 Poplars

The *Populus* genus (family Salicaceae) is spread over most of the northern hemisphere, mainly within the temperate zone. However, for most parts of northern Europe, only European aspen (*P. tremula* L.) is an endemic species, hence aspens and their hybrids are presented hereinafter separately from other poplars, which are exotic to the region's forestry. Most *Populus* species are fast-growing, and they are dioecious and hybrids between species are common. Poplars are divided into six sections; among them black poplars (e.g. *P. deltoides, P. nigra*), aspens and white poplars

Table 3.1 Main characteristics of selected deciduous tree species for short-rotation forestry (SRF) in the Nordic and Baltic countries representing hemiboreal region with northern temperate climate. The item "mean annual growth" refers to stem wood excluding tops and branches except willows, the growth of which includes also stem tops and branches. Growth in terms of mass represents dry mass.

Item	Poplars	Hybrid aspen	Grey alder	Willows
Establishment	Cuttings or rooted cutting in the first generation; new cuttings or vegetatively from root suckers in the following generations	Micro-propagated clonal plants in the first generation; vegetatively from root suckers in the following generations	Seedlings in the first generation; vegetatively from root suckers in the following generations	Cuttings in the first generation; vegetatively from root suckers in the following generations
Rotation	20–25 years for energy wood, pulpwood and logs, < 20 years for energy wood only	20–25 years for energy wood, pulpwood and logs, < 20 years for energy wood only	15–20 years for energy wood, pulpwood and logs, < 15 years for energy wood only	3–5 years for energy wood only
Site requirements	Fertile, fresh to moderately moist	Fertile, fresh to moderately moist	Fertile, fresh to moderately moist	Fertile, fresh to moist
Mean annual growth at fertile sites	$25\,\mathrm{m}^3\,\mathrm{ha}^{-1}\,\mathrm{a}^{-1}$ $7.5–10\,\mathrm{Mg\,ha}^{-1}\,\mathrm{a}^{-1}$	$20–25\,\mathrm{m}^3\,\mathrm{ha}^{-1}\,\mathrm{a}^{-1}$ $7–9\,\mathrm{Mg\,ha}^{-1}\,\mathrm{a}^{-1}$	$7–11\,\mathrm{m}^3\,\mathrm{ha}^{-1}\,\mathrm{a}^{-1}$ $3–5\,\mathrm{Mg\,ha}^{-1}\,\mathrm{a}^{-1}$	$8–12\,\mathrm{Mg\,ha}^{-1}\,\mathrm{a}^{-1}$
Main advantages	Highly productive (selected poplar clones are probably the most fast-growing trees in the region), easy to establish with cuttings	Highly productive with high quality wood fibres, cold-resistant	At young age, among the fastest growing domestic deciduous trees, not susceptible to damagers, improves soil N-content	Highly productive in very short rotations
Main concerns or constraints	Exotic for the region, cold-resistance in central and northern boreal areas is low	High establishment cost, half-exotic for the region, high risk of herbivore browsing	No market for other assortments than for energy wood	Requires fertilization for high productivity, pathogens reduce growth

(e.g. *P. alba*, *P. tremula*, *P. tremuloides*) and balsam poplars (e.g. *P. balsamifera*, *P. trichocarpa* and *P. maximowiczii*) are the most interesting for SRF.

Most experiences of poplars in the Nordic and Baltic countries come from southern Sweden and Denmark. In these conditions, balsam poplars (both pure clones and

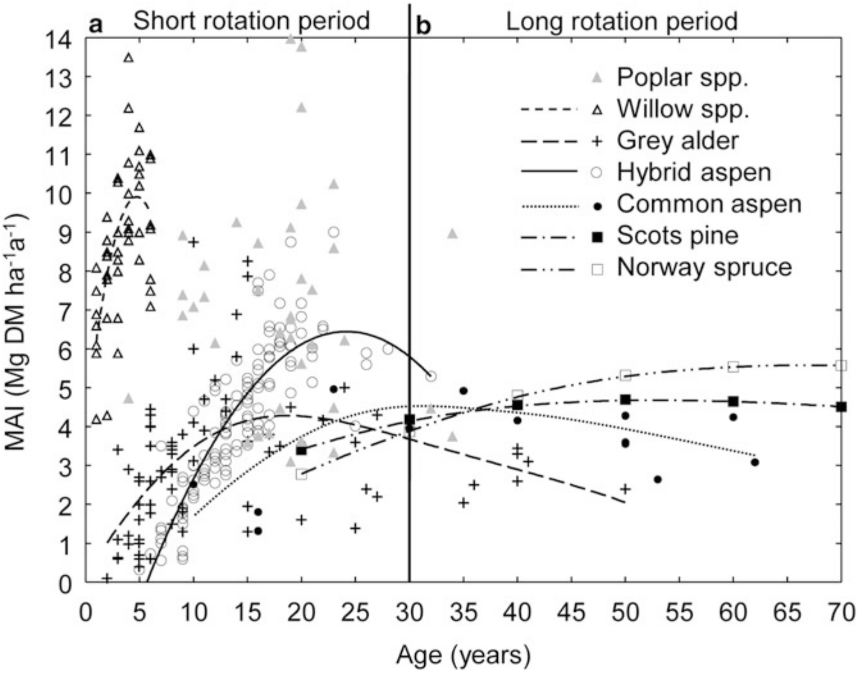

Fig. 3.1 Mean annual increment (MAI) of dry biomass of selected tree species potential for short- and long-rotation forestry in northern Europe. **a** total above-ground biomasses in planted and fertilized willow (Willebrand et al. 1993) and poplar stands (Johansson and Karačić 2011), stem biomasses in planted and natural grey alder (Aosaar et al. 2012) and planted hybrid aspen stands (Rytter and Stener 2005; Tullus et al. 2012a, unpublished data). **b** stem biomasses in planted and natural *P. tremula* stands (Krigul 1971; Tullus et al. 2012a), and natural Scots pine and Norway spruce stands in fertile sites (Krigul 1971)

hybrids) are very promising. Especially, the balsam hybrid clone OP42 (*P. maximowiczii* × *P. trichocarpa*) is a dominating clone, but today twelve other clones are available to be grown in mixtures. Furthermore, pure *P. trichocarpa* plantations are under evaluation (Christersson 2006). The balsam poplars (e.g. *Populus balsamifera, P. trichocarpa*) seem to have a higher potential to adapt to the Nordic conditions than the black poplars (e.g. *P. deltoides, P. nigra*) or the white poplar (*P. alba*).

In general, all poplars tested in northern Europe are nutrient-demanding species (Grosscurth 1972; Bergstedt 1981), which are most successful on well drained soils with continuous supply of water and low acidity (pH value of 5.5–7.5) (Bergstedt 1981; Boysen and Strobl 1991). This implies that poplars are sensitive to drought (Blake et al. 1996) due to their low efficiency of water use (Marron et al. 2008). However, breeding for increased drought tolerance is possible (Tyree et al. 1979) although the relationship between growth and drought tolerance seems to be low. In general, poplars are resistant to occasional flooding but such conditions seem to reduce their growth (Neuman et al. 1996). The high demand for light of black poplars makes them more sensitive to shading than is the case for other poplar groups including the balsam poplars (Bergstedt 1981).

3.3.2.2 Aspens, Hybrid Aspen

Aspens belong to the section *Populus* (formerly *Leuce*, aspens and white poplars) of the genus *Populus*. This section includes ten species (Eckenwalder 1996). Regarding SRF, the most promising species are *Populus tremula* L. (common, European or Eurasian aspen) and its hybrid with its North-American counterpart *P. tremuloides* Michx. (quaking or trembling aspen). *P. tremula* is among the most widely distributed tree species in the world (Worrell 1995), while *P. tremuloides* is among the most widely distributed tree species indigenous in North America (Dickmann and Kuzovkina 2008).

P. tremula is a common native deciduous tree in the Nordic and Baltic countries. It is a fast-growing species grown for energy biomass, pulp wood and logs. In the latter two cases, its economic value may be reduced by stem heart rot, which commonly damages mature trees. In this respect, the hybrid between *P. tremula* and *P. tremuloides*, known as hybrid aspen (*P.* × *wettsteinii* Hämet-Ahti = *P. tremula* L. × *P. tremuloides* Michx.) represents a high capacity to produce biomass with lower risks of damage by pathogens like stem heart rot (Tullus et al. 2012a). During recent decades about 4500 ha of hybrid aspen has been cultivated in the Nordic and Baltic countries both for experimental and practical purposes (Tullus et al. 2012a).

3.3.2.3 Alders, Grey Alder

Alders belong to the family Betulaceae, genus *Alnus*. Two species occur naturally in the Nordic and Baltic countries: grey alder (*A. incana* (L.) Moench.) and black alder (*A. glutinosa*). Grey alder grows throughout northern Europe, except in the very southern part of Sweden and Denmark (Hultén 1950). Black alder, on the other hand, has a more southern origin than grey alder. Black alder is absent from the very northern parts of the Nordic countries.

Grey alder is a fast-growing but relatively short-lived species. It has a rapid juvenile growth but is surpassed by black alder at the age of 25–30 years (Ljunger 1972). Alders are typically light-demanding species, which efficiently colonize bare land, e.g. abandoned agricultural land. Germ plants are small as are the nutrient reserves in seeds. Therefore, establishment from seed is hampered by thick humus and competition from other plants. Grey alder prefers moist and nutrient rich sites, and it can grow on somewhat drier and less fertile soils than black alder (Rytter 1996). Grey alder also tolerates acidity (low soil pH down to a pH value of 4) without negative effects on growth (Ericsson and Lindsjö 1981). A symbiotic nitrogen-fixing bacterium (*Frankia* spp.) in root-nodules makes alder species self-sufficient in nitrogen (N) supply. In grey alder, the fixation of atmospheric N_2 may exceed 100 kg N ha^{-1} a^{-1} (Rytter 1996), but the fixation rate is dependent on the age of the tree and soil and light conditions. Fertilization, leading to enhanced nutrient concentrations in the soil, may temporally reduce the N_2 fixation (Rytter et al. 1991).

3.3.2.4 Willows

Salix species represent a variable morphology spanning from trees (e.g. *S. caprea* L. and *S. fragilis* L.) through bushes (e.g. *S. viminalis* L. and *S. dasyclados* Wimmer) to small shrubs (e.g. *S. polaris* Wahlenb. and *S. repens* L.). In particular, the bush-formed species are used in biomass plantations and breeding. In this respect, an important attribute is the capacity for resprouting after harvest (cf. Sennerby-Forsse and Zsuffa 1995). *Salix* species are spread over the whole of northern Europe, but the species choice varies from region to region. In general, *Salix* species are well-adapted to the climate in northern Europe. *Salix* species consume large amounts of water per unit yield of biomass (Grip et al. 1989; Lindroth et al. 1994). Therefore, willow cultivation should be established only on sites where the availability of water is high enough in relation to needs for unlimited growth.

Most work in breeding *Salix* species is done in Sweden. Since the 1980s, more than twenty varieties have been produced. They all have the Community Plant Variety Right, and they are thus protected in the EU. The willow species mainly recommended for plantations include *S. dasyclados* Wimmer and crossings between *S. viminalis* L. and *S. schwerinii* E. L. Wolf. At the moment, research is being done to develop genetic markers for use in practical breeding (e.g. Berlin et al. 2011; Samils et al. 2011).

3.4 Productivity and Varieties of Poplar, Aspen, Alder and Willow

3.4.1 Poplars

Regarding poplar clones, the growth is up to 25 m^3 ha^{-1} a^{-1} of stem wood in the southern parts of the Nordic and Baltic countries (e.g. Rytter 2004), but future breeding work will most probably increase the growth (Telenius 1999; Karačić et al. 2003; Stener 2010). The basic wood density of the currently used poplars is 300–390 kg m^{-3} depending on the clone and the age of trees (Stener 2010). Consequently, the growth of dry matter of stem wood in poplars is 7.5–10 Mg ha^{-1} a^{-1}. Including branches without leaves, the biomass growth of poplars could increase by 7–20 % (cf. Rytter 2004) (Fig. 3.1). However, the growth of poplars is heavily dependent on the length of rotation and the degree of adaptation to the prevailing conditions. In the poplar experiments, rotations of only 8–12 years are used, but there are damages in the young trees (e.g. Stener 2010) indicating that the poplars used so far are not sufficiently adapted to the northern-European climate (Christersson 2006).

3.4.2 Aspen and Hybrid Aspen

Aspen plantations may supply raw material for pulp and paper mills, but their role in supplying biomass for energy use has increased rapidly (Christersson 1996; Beuker

2000; Rytter 2006). However, plantations of aspen are not as common in the Nordic and Baltic countries as is the case for hybrid aspen. Breeding programs have provided clones of hybrid aspen, whose annual mean growth is 20–25 m^3 ha^{-1} a^{-1} when applying a rotation of 20–30 years. In terms of biomass, this corresponds to 7–9 Mg ha^{-1} a^{-1} of dry mass (Fig. 3.1). Even if regenerated through root suckers, the productivity of hybrid aspen is high. In southern Sweden, the annual mean growth was 9.5 Mg ha^{-1} a^{-1} of dry mass already at the age of four years (Rytter 2006). Thus, young dense stands of hybrid aspen are as productive as willow stands. Due to high establishment costs, low planting densities are preferred in establishing hybrid aspen plantations and productivity figures can be quite low in young planted stands (Tullus et al. 2009).

The wood of hybrid aspen is high in carbohydrates and low in lignin, the average concentrations of cellulose and acid-insoluble lignin varying in the ranges of 50–60 % and 11–20 % (Zeps et al. 2008; Tullus et al. 2010). The cells of aspen wood are narrow in diameter, and the wood is composed of thin-walled fibres, which are ideal for producing a high-density paper with a smooth surface (Karl 1988; Dhak et al. 1997). Furthermore, the wood density of hybrid aspen is less than that of *P. tremula*; i.e. 350–370 kg m^{-3} for hybrid aspen and 370–380 kg m^{-3} for *P. tremula* (Kärki 2001; Rytter and Stener 2003; Rytter 2004; Heräjärvi and Junkkonen 2006; Stener 2010). However, the basic density is age-dependent, with lower values during the first ten years but increasing thereafter with age (Heräjärvi and Junkkonen 2006; Stener 2010). The energy value of hybrid aspen biomass is 19.3 kJ g^{-1} for stem wood and 20.3 kJ g^{-1} for current-year shoots (Tullus et al. 2009).

3.4.3 Grey Alder

The annual mean productivity of grey alder is usually 3–5 Mg ha^{-1} a^{-1} but may reach up to 8.8 Mg ha^{-1} a^{-1} of stem wood (Rytter 1996, 2004; Aosaar et al. 2012) (Fig. 3.1) depending on the stand age, site conditions, stand density, fertilization, and type of regeneration (natural, planted), the highest values representing dense stands. The current estimates exclude any contribution from breeding activities. However, Matthews (1987) regarded grey alder as a promising candidate for breeding, because it flowers at an early age and can be propagated vegetatively from cuttings or using tissue and cell cultures. The basic density of the wood of grey alder is 350–400 kg m^{-3} (Aosaar et al. 2012), which is close to that of *Populus* species and Norway spruce.

3.4.4 Willows

The productivity of *Salix* species varies substantially in commercial plantations (Fig. 3.1). In fertilized experiments, more than 10 Mg ha^{-1} a^{-1} of woody biomass is obtained (e.g. Willebrand et al. 1993; Alriksson 1997; Bullard et al. 2002; Heinsoo et al. 2002) (Fig. 3.1). In commercial willow plantations, the values are smaller depending on the management intensity and the choice of land for plantation (Dimitriou et al. 2011). The rotation for willow plantations is short ($\sim$ 3–5 years) with the consequence that the current annual increment is very different among years and cutting cycles, lower values representing the first or second year and higher values the years thereafter (e.g. Willebrand et al. 1993; Alriksson 1997).

3.5 Establishment, Management and Profitability of Short-Rotation Tree Plantations

3.5.1 Poplars

Poplars are planted either directly using stem cuttings or using nursery-rooted cuttings, whose survival is higher than cuttings with no roots used in direct planting. In direct planting, the optimal cutting is 25 cm long and 1–3 cm thick (Boysen and Strobl 1991). Planting density is between 3.6 × 3.6 m and 2.5 × 2.5 m; i.e. about 800 to 1,600 plants ha^{-1} (Boysen and Strobl 1991; Rytter et al. 2011a). Poplars are less expensive to establish than hybrid aspen, because a costly micro-propagation technique is used for hybrid aspen.

Management of poplars resembles that of hybrid aspen (Fig. 3.2). In general, pre-commercial thinning is not required due to the low initial stand density. The first thinning is recommended to be done after 10–15 years since planting depends on the initial stand density (Rytter et al. 2011a). No more than one thinning is needed before the final cut at 20–25 years from planting if the management aims only at producing energy biomass (Rytter et al. 2011b). After harvest, many clones regenerate from stump shoots, and these can be used for the next generation. Unfortunately, this is not always the case, sprouting of stumps may be small and/or sprouts will die (McCarthy and Rytter 2012). Regeneration of poplars from stump sprouts is still poorly known and further research about the sprouting ability among clones is needed for developing proper regeneration methods.

3.5.2 Aspen, Hybrid Aspen

In northern Europe, the aspen plantations represent mainly hybrid aspen, whose clones are propagated using micro-propagation or root cuttings (e.g. Stenvall et al. 2004). The planting density is generally 1,100–1,600 plants ha^{-1} (Tullus et al. 2012a). Usually site preparation is carried out, and chemical or mechanical weed control is needed during the first year(s), if the plantation is established on agricultural land. A higher planting density ($>$ 4000 plants ha^{-1}) could be used if the plantation is established for the production of energy biomass in very short rotations (5–10 years) as done in central Europe (Liesebach et al. 1999). Traditionally, high planting densities have not been used in northern Europe, where hybrid aspen is planted for producing merchantable timber. In such a case, the rotation is 20–25 years (Hynynen et al. 2004; Rytter and Stener 2005, Fig. 3.2), with the total yield of pulpwood and logs of 300–450 m^3 ha^{-1}. During the rotation, one to three thinnings

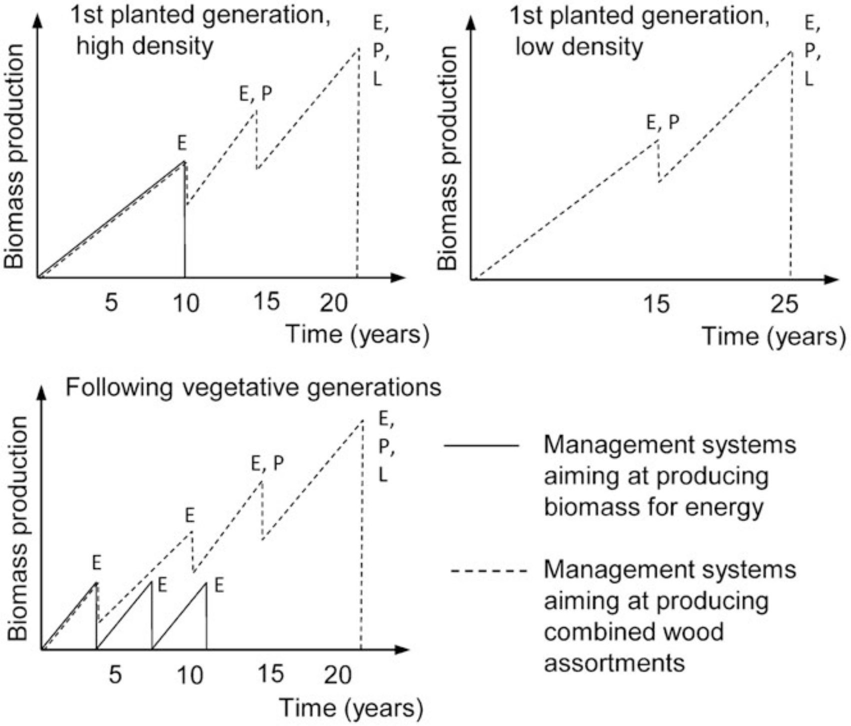

Fig. 3.2 Theoretical management (timing of thinnings and final fellings) schemes for *Populus* (poplar and hybrid aspen) plantations aimed at producing energy wood (E), pulpwood (P) and logs (L). The first thinning in the following vegetative generations corresponds to pre-commercial thinning in conventional forestry

are needed depending on the initial stand density, the growth rate and intensity of thinning.

When producing energy biomass in hybrid aspen plantations, management with shorter rotations and vegetative regeneration in establishing successive generations would probably be successful (Liesebach et al. 1999; Rytter 2006). High-quality energy biomass (i.e. higher wood content and lower bark content of the stems) may be obtained if the harvesting is done in the phase when the breast-height diameter of trees exceeds 4 cm (Tullus et al. 2009). Furthermore, management combining early harvest of root suckers in conventional forestry is proposed for hybrid aspen (Fig. 3.2). In this case, the biomass in root suckers could be exploited by using corridor cleaning, while the remaining trees could be managed by applying ordinary forestry practices to produce pulp wood and logs (Rytter 2006).

There are few economic assessments concerning the commercial plantation of hybrid aspen. Rytter et al. (2011b) and Tullus et al. (2012b) showed positive results with an internal rate of return (IRR) of around 8–10 % in Swedish and Estonian conditions, if the plantations were established on abandoned agricultural land. With current prices in Estonia, the IRR was 1 % higher than that for silver birch plantations

(Tullus et al. 2012b). Similarly, the economic calculations for the Swedish conditions show that hybrid aspen could be competitive with Norway spruce on forest land and with food production on agricultural land (Rytter et al. 2011b). However, the price relations between food, energy biomass, pulpwood and saw logs ultimately determine the economic ranking.

3.5.3 Grey Alder

In establishing grey alder plantations, seedlings should be inoculated with *Frankia* in order to enhance the early growth of seedlings (e.g. Hendrickson et al. 1993). Until now, there has been little experience on grey alder plantations, but the optimal planting density in biomass-oriented management is 2000–3000 plants ha^{-1} (cf. Matthews 1987; Almgren 1990). This implies that the canopy will close in a reasonable time after planting and no thinning will be needed during the rotation. Grey alder regenerates successfully also through seeds, root suckers and stump sprouts. In the latter two cases, regeneration through root suckers seems to dominate (Rytter et al. 2000). The initial density of natural grey alder stands is usually high, which may facilitate precommercial thinning combined with biomass removal. In such a case, the optimal density is probably $< 10{,}000$ plants ha^{-1} in the early phase, whereas 3,000 to 6,000 plants ha^{-1} may be left to the final harvest (Aosaar et al. 2012). In producing energy biomass in dense stands, a rotation length of 15–20 years is recommended, while when combining energy biomass production and the production of small-sized timber the rotation length of 25–30 years is more optimal (Daugavietes et al. 2009; Aosaar et al. 2012). During the rotation, up to two thinnings may be needed in order to avoid mortality due to self-shading in growing and developing stands (Rytter 1995).

The costs of establishing grey alder stands are probably similar to that of birch since the production costs of plants and the planting techniques are similar. On the other hand, there are no regeneration costs, if root and stump shoots are used in regeneration (Rytter et al. 2000). The economy of SRF based on grey alder is still poorly known. Opdahl and Veidahl (1993) found that the production of energy biomass combined with the production of pulp wood and saw logs was an economically attractive way to manage grey alder. Regarding more biomass-oriented forestry, Mizaras et al. (2011) concluded that SRF based on grey alder may be profitable, especially if whole-tree techniques were used in harvesting.

3.5.4 Willows

Current plant material allows the establishment of willow plantations over most of northern Europe, except the very northern areas with the harshest climates. Willow plantations are established only on fertile agricultural land with an ample supply of water and low acidity (pH over 5.5) (Gustafsson et al. 2007; Hollsten et al. 2012). In general, cuttings of 18–20 cm are used in mechanized planting, which is the most common way to establish *Salix* plantations (Gustafsson et al. 2007; Hollsten et al. 2012). In general, the planting density is 13,500–15,000 cuttings ha^{-1} (Gustafsson et al. 2007) in double rows with the spacing of 75 and 150 cm in order to facilitate mechanized harvesting operations. The distance between cuttings in the row is 60–65 cm. Biomass yield in *Salix* plantations may be further increased by increasing the planting density and shortening rotation (Bullard et al. 2002; Szczukowski et al. 2005).

Weed control, both before and after planting, can improve regeneration and enhance the early growth of *Salix* saplings. The rotation length of 3–5 years is commonly used, but it depends on site conditions and management. After the harvest, the sprouting stools establish a new generation. In general, the same stools may be used for 25 years (Gustafsson et al. 2007); i.e. five to six harvest cycles are possible, before new planting is necessary. Nitrogen fertilization will substantially enhance the growth of willows, e.g. an addition of 220 kg N ha^{-1} during a three-year cycle may increase the biomass yield up to 60 % compared with no N addition (Aronsson and Rosenqvist 2011). In general, the economics shows positive net results (e.g. Ledin 1996; Ericsson et al. 2009; Rosenqvist 2010), but the production is currently subsidized.

3.6 Environmental Issues

An increased area of poplar plantations on forest and/or agricultural lands has both positive and negative environmental impacts. Above all, the biomass produced in poplar cultivations can be used to substitute for fossil fuels and/or enhance carbon sequestration in soil, especially on agricultural land. Furthermore, the energy balance of producing biomass in poplar plantations is positive compared with other crops (Börjeson 2006; Rytter et al. 2011b). On the other hand, poplar plantations may reduce the biodiversity, depending on where the plantations are located in the landscape and how they are managed. However, the biodiversity values of poplar plantations seem to be higher than Norway spruce plantations but lower than deciduous plantations or a mixture of deciduous and coniferous trees (Weih et al. 2003; Britt et al. 2007; Rytter et al. 2011b). Regarding the effects on recreational and amenity values of the landscape, the effects are probably more positive than those of Norway spruce plantations.

Grey alder is a common species in the Nordic and Baltic countries, and it has thus a natural value for nature conservation in raising the share of deciduous species in forests. Additionally, grey alder is a domestic species in northern Europe, with no complications with forest laws or certifications. The N_2-fixing ability of grey alder will provide the possibilities to produce energy biomass in a sustainable way, wherever grey alder dominates the tree communities used for biomass supply. Furthermore, grey alder is capable of growing on soils polluted by heavy metals (Hawrys 1987), and thus it can be used to ameliorate soil and enhance the afforestation of polluted land areas.

SRF with willows in its current form represents an intensive management with continuous supply of fertilizers. However, Weih (2009) concluded that willow plantations can improve the biodiversity at the landscape level, especially if they replace cereal cultivation or Norway spruce plantations in a homogenous agricultural landscape and make the landscape structure more variable. Willow species may also improve the properties of soils used for a long time for conventional agriculture, e.g. more carbon will be stored in the soil (Kahle et al. 2005; Rytter 2012). Furthermore, willow plantations may be used for the phyto-remediation of polluted soils (Aronsson and Perttu 2001; Bertholdsson 2001; Mleczek et al. 2010), e.g. to take up heavy metals from the soil (Sennerby-Forsse et al. 1993; Labrecque et al. 1995; Bertholdsson 2001).

3.7 Conclusions

Short-rotation forestry (SRF) is a promising and environmentally sound way to produce biomass for energy production in northern Europe. For establishing SRF plantations, there exist several domestic and exotic deciduous tree species whose productivity is high even in the northern climate. These include several poplar species and their hybrids, hybrid aspen, grey alder and several willow species and their hybrids. The annual mean biomass growth of these species can be as high as 5–12 Mg ha^{-1} a^{-1} during rotations of less than 20–30 years. However, the highest productivity is achieved only on fertile soils or if the cultivation is fertilized, as is the case of willows. These species provide flexible management opportunities in producing energy biomass alone or combined with the production of other materials (pulpwood, logs). Most of the environmental concerns regarding SRF are similar to those of traditional forestry.

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Chapter 4
Integrated Production of Timber and Energy Biomass in Forestry

Johanna Routa, Piritta Pyörälä, Ashraful Alam, Antti Kilpeläinen, Heli Peltola and Seppo Kellomäki

Abstract Current management aimed purely at producing timber is not necessarily appropriate in managing forests to combine the production of timber and energy biomass and to maintain or even to increase carbon storage in forest ecosystems. Key questions are; how to integrate the management efforts to enhance the production potentials, how to sustain the production (e.g. carbon and nutrient balances), and what their overall economic implications are for forestry. Proper choice of tree species and improved clones in planting, spacing (planting density, thinning regimes), rotation length, and fertilization are important in enhancing the combined production of timber and biomass, in increasing carbon stocks in forest ecosystems, and in the mitigation of climate change.

Keywords Bioenergy and pre-commercial thinning · Carbon neutrality · Climate change · Clone · Combined production · Direct and indirect emissions · Energy biomass · Fertilization · Forestry · Initial stand density · Management · Mitigation · Provenance · Rotation length · Terminal cut · Timber · Species · Yield

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4.1 Links Between Timber and Energy Biomass in Forestry

4.1.1 Supply of Energy Biomass in Forestry

In the Nordic countries, the main part of forest biomass used in the energy industry is comprised of logging residues originating from the harvest of timber (pulp wood, saw logs). In thinnings, logging residues include small-dimensioned stems, with branches and foliage (whole tree), whereas logging residues from final felling represent mainly top parts of stem, branches, and foliage, and even stumps and part of the coarse roots. Furthermore, whole trees removed in tending and pre-commercial thinning of seedling stands can be used for energy biomass. The close links between timber and energy biomass imply that the long-term supply of energy biomass is much affected by management and cutting regimes, which e.g. optimize the timber production under the given objectives and constraints. The same management operations are available for the production of timber and energy biomass, but the optimal way to combine them in integrated production is not well known. The key questions are how different management options, e.g., initial stand density, timing and intensity of thinning, rotation length, fertilization and choice of provenance, may affect the yields of timber and energy biomass separately and in combination. These issues are discussed in this chapter, with the focus on the effects of varying management on the potentials to produce timber and energy biomass in forestry.

4.1.2 Energy Biomass Obtainable in Forestry Optimized for Timber Production

The potentials to produce energy biomass in timber-oriented management are highlighted by Kärkkäinen et al. (2008). They used two management and cutting regimes in studying how much energy biomass would be potentially available by optimizing the timber production throughout Finland (23 million ha^{-1} of forest land). In the first option, the management aimed at “maximizing the net present value for timber production using a 5 % interest rate (Max cutting scenario)”. In the second option, the management aimed at “maximizing net present value from timber production using a 4 % interest rate with non-decreasing flow of pulpwood, saw logs and net income over a given period and net present value after the 50-year period greater than or equal to the beginning (Sust cutting scenario)”. The analysis was based on the management-oriented large-scale forestry model (MELA) using the sample plots and tree data of the National Forest Inventory (NFI). Furthermore, the potential impacts of climate change on growth were included in the analysis. Five 10-year periods; 2003–2012, 2013–2022, 2023–2032, 2033–2042, 2043–2052 (Fig. 4.1), were studied.

When the *maximum cutting scenario* was applied, the potential timber supply was 103 million $m^3\ a^{-1}$ under the scenario of projected climate change and 105 million $m^3\ a^{-1}$ under the current climate (i.e. in both cases about $4.5\ m^3\ ha^{-1}\ a^{-1}$)

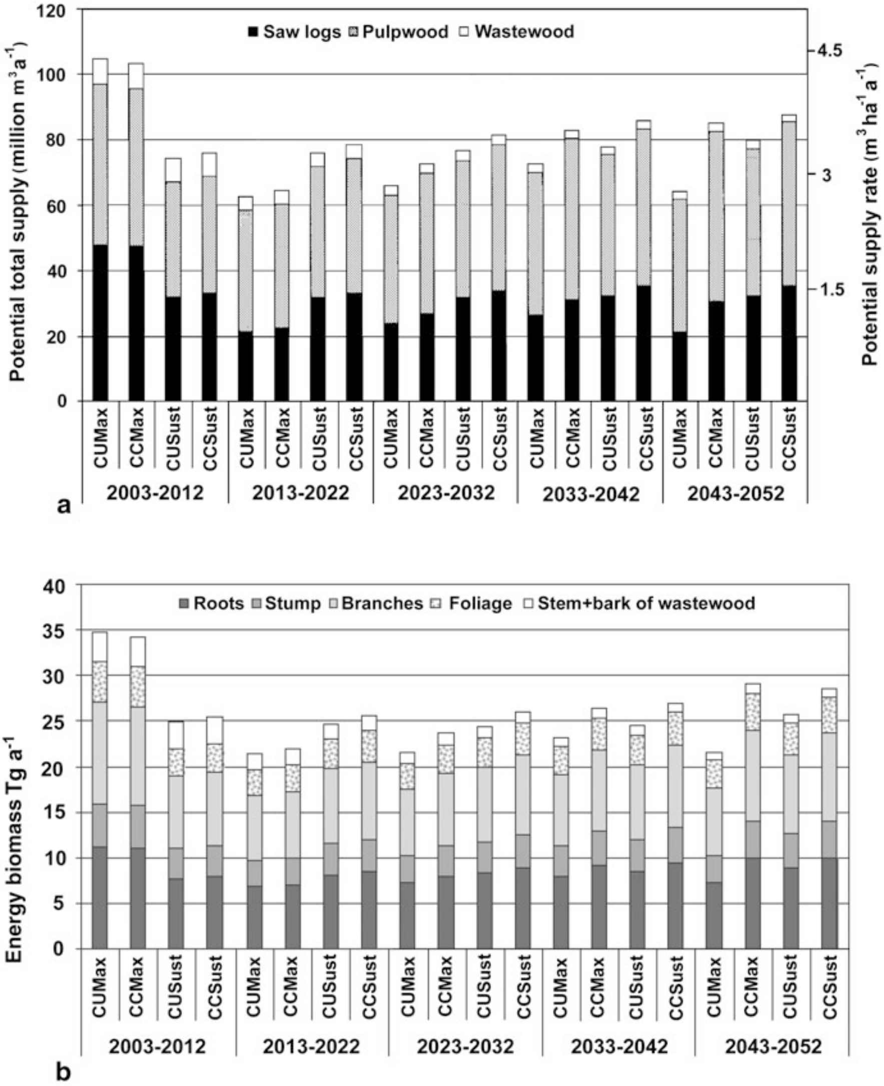


Fig. 4.1 Annual removal of timber (**a**) and energy biomass (**b**) under the current (*CU*) climate and changing climate (*CC*), when applying the maximum cutting (*Max*) and the sustainable cutting (*Sust*) scenarios (Kärkkäinen et al. 2008). Climate change (*CC*) implies a gradual increase of temperature by 6 °C from the current (*CU*) and increase of the atmospheric CO_2 from 350 ppm to 683 ppm by 2099

in the period 2003–2012 (Fig. 4.1). At the same time, the potential amount of energy biomass was about 35 Tg a^{-1} (79 million m^3 a^{-1} (3.4 m^3 ha^{-1} a^{-1})) regardless of the climate scenario. In the Max scenario, the amount of potential timber supply decreased by the end of the simulation period (2052). This was because final cuttings in the first period changed the structure of forests, reducing the amount of forests with trees mature for final cut, thus reducing the total cutting potentials. In the final period (2043–2052), the potential timber supply was 64 million m^3 a^{-1} (2.8 m^3 ha^{-1} a^{-1}) and energy biomass 22 Tg a^{-1} (49 million m^3 a^{-1} (2.1 m^3 ha^{-1} a^{-1})) under the current climate, whereas under the climate change scenario the timber supply was 85 million m^3 a^{-1} (3.7 m^3 ha^{-1} a^{-1}) and energy biomass 29 Tg a^{-1} (66 million m^3 a^{-1} (2.9 m^3 ha^{-1} a^{-1})), respectively.

In the *sustainable cutting scenario*, the timber supply in the period 2003–2012 was 74 and 76 million m^3 a^{-1} (about 3.2 m^3 ha^{-1} a^{-1}) depending on the climate scenario, whereas the potential supply of energy biomass was 25 Tg a^{-1} (57 million m^3 a^{-1} (2.5 m^3 ha^{-1} a^{-1})) for both climate scenarios. During the period 2043–2052 with the current climate, the potential supply of timber was 80 million m^3 a^{-1} (3.5 m^3 ha^{-1} a^{-1}) and energy biomass 26 Tg a^{-1} (59 million m^3 a^{-1} (2.6 m^3 ha^{-1} a^{-1})). Under the climate change scenario, the timber supply was 88 million m^3 a^{-1} (3.8 m^3 ha^{-1} a^{-1}) and 29 Tg a^{-1} (65 million m^3 a^{-1} (2.8 m^3 ha^{-1} a^{-1})). Regardless of management and climate scenarios, the share of energy biomass was 44 % of the total removal including timber and energy biomass. In the Sust scenario, the amount of cuttings was more stable due to the constraints set for the cuttings in optimization.

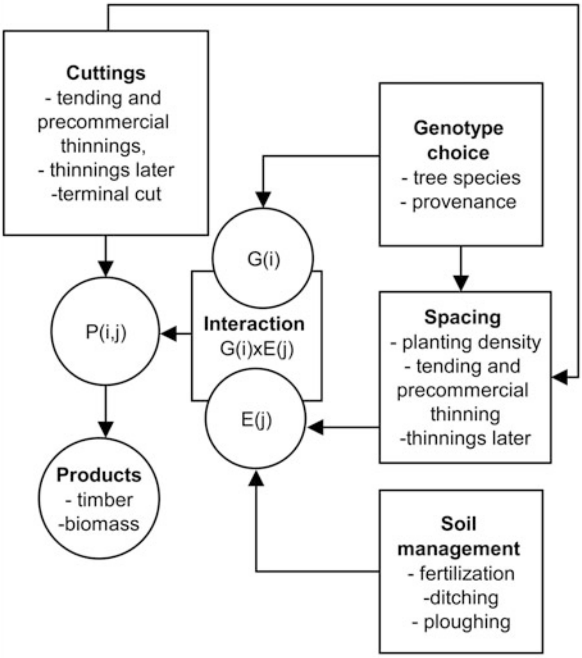
Regardless of the cutting and climate scenarios, Kärkkäinen et al. (2008) found further that the proportion of saw logs will decrease in cuttings during the coming 50 years, whereas the amount of pulpwood from cuttings will increase. This will have an effect on the yields of energy biomass provided by cutting residues. In future, the amount of forests in the thinning phase is likely to increase, with a consequence that the proportion of residues will be greater compared to that of stems harvestable for timber. Furthermore, the amount of spruce-dominated mature forest may decrease, which will reduce the total amount of residues from spruce-dominated forests in the future (Kärkkäinen et al. 2008).

4.2 Management for Producing Timber and Energy Biomass in Forestry

4.2.1 Interaction Between Environment and Genotype in Producing Tree Biomass

The production of timber and biomass is based on the management of the structure of forest ecosystems to produce timber and energy biomass as defined in management objectives. Management of the genetic properties of tree populations (*G (i)*) and the properties of sites (*E (j)*) or both can be used to maintain or increase the production (*P (i, j)*) as shown in Fig. 4.2. The choice of proper genotype (tree species or provenance of selected tree species) for the given site is fundamental. Management of the spacing of trees, from planting to the last thinning before terminal cut, controls the competition between different tree species and/or between individuals of the

Fig. 4.2 Schematic representations on how selected management measures effect on the interaction between the genotype and environment and how they are affecting the production of timber and biomass alone or combined



same species in order to enhance the supply of resources for growth. Similarly, soil management can be used to enhance the resources available. In the short term, fertilization is an effective way to enhance forest growth, whenever shortage of nutrients (e.g. nitrogen) is the main factor limiting forest growth. In successful management, the choice of genotype, tree spacing and soil management are combined in such a way that the site conditions are optimized (tailored) for a given genotype over the whole production cycle (rotation).

4.2.2 Recovery of Energy Biomass in Management

There are several ways to supply energy biomass in forest management (Fig. 4.3). Even in the seedling phase, energy biomass may be obtained in tending and pre-commercial thinning. *Pre-commercial thinning* is done some years before the first commercial thinning. Usually, pre-commercial thinning is a management option, wherever narrow spacing and competing tree species limit the growth due to neglected tending in the seedling phase. However, pre-commercial thinning may be delayed deliberately in order to allow bioenergy thinning before commercial thinning. *Bioenergy thinning* is similar to that of commercial thinning as shown in Fig. 4.3, where the grey part represents small-dimensioned trees not yet economic for thinning for pulp wood. *Commercial thinning* is used to control the stocking and harvest of timber

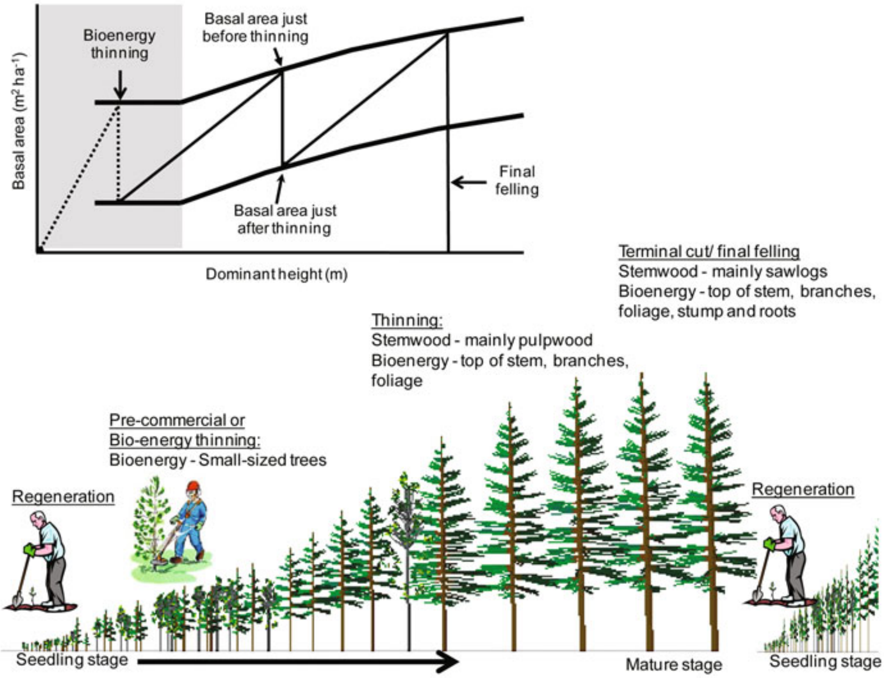


Fig. 4.3 Outlines of the growth and development of managed tree stand, with the supply of energy biomass in different phases of growth and development. The insert (*upper left*) represents the use of thinning, where the timing and intensity of removal of basal area is given as a function of dominant height acclimating the thinning procedure to the growth and development of tree stand. The grey area in the insert shows the phase for bioenergy thinning prior to commercial thinning

(mainly pulp wood) and energy biomass (whole trees excluding stumps and roots) to enhance the growth of remaining trees in order to increase the share of saw logs in terminal cut. In *terminal cuts*, the main part of the energy biomass represents stem tops, branches and foliage, but stumps and part of the coarse roots may be harvested too.

4.3 Biomass Recovery in Pre-commercial and Bioenergy Thinnings

In general, management that uses high initial stand density combined with late bioenergy thinning could produce the most energy biomass over the whole production cycle (e.g. Heikkilä et al. 2007; Routa 2011, 2012a; Alam et al. 2010, 2012). High initial stand density will shorten the time before the canopy closure, with full use of the available resources and fast accumulation of biomass in growth. Thus, a proper strategy for increased bioenergy production in forestry would be to use high initial stand density and exclude the pre-commercial thinning in order to increase the growth

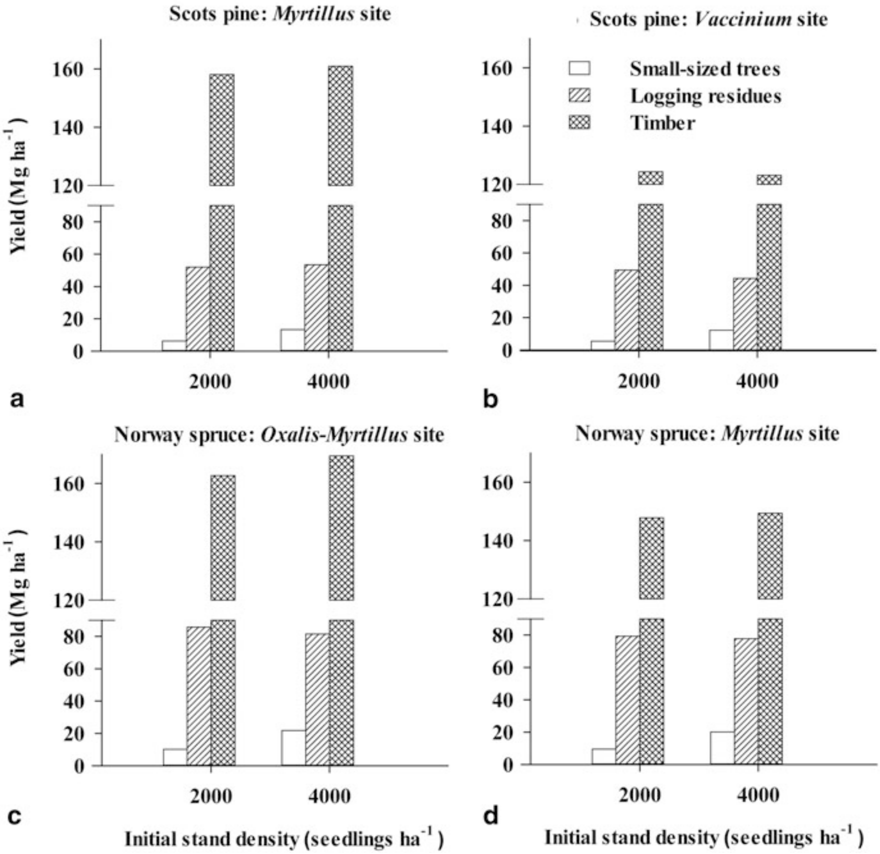


Fig. 4.4 Effects of initial stand density on the yields of timber (pulpwood and sawlogs) and energy biomass in Scots pine (*upper part*) at medium-fertile (*Myrtillus* type) (**a**) and poor (*Vaccinium* type) (**b**) sites and in Norway spruce (*lower part*) at fertile (*Oxalis-Myrtillus* type) (**c**) and medium-fertile (*Myrtillus* type) (**d**) sites. The simulations were done based on the model described in Box 4.1 using the current practices in bioenergy and commercial thinnings. Energy biomass (whole trees) was harvested only in bioenergy thinning and from terminal cut area in the form of logging residues. Timber was harvested in commercial thinnings and terminal cut

of energy biomass during the period from planting to the bioenergy thinning, which would be done as late as possible but not damaging the growth in coming years. However, the applicability of such a strategy is site- and species-specific, e.g., increasing initial stand density on fertile sites yields more energy biomass during bioenergy thinning than on poor sites. Furthermore, Norway spruce produces more energy biomass than Scots pine (Fig. 4.4), because Norway spruce is relatively insensitive to narrow spacing, producing long crowns with large amount of biomass even in trees in the thinning phase (Heikkilä et al. 2007; Routa 2011; Alam et al. 2012). This is not the case for Scots pine, whose crown development is much reduced by narrow spacing.

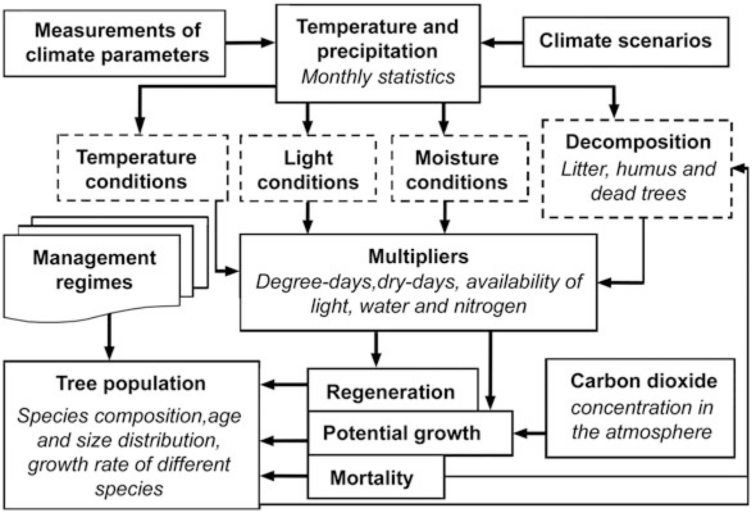


Fig. 4.5 Outlines of Sima model used in simulating the management for producing energy biomass

Box 4.1 Simulation of Managing Forest Ecosystems for Supply of Energy Biomass

In this chapter, the Sima model is used in several phases to investigate the management effects on the yields of timber and biomass. The model is a gap-type one, where the growth of trees is affected by the climatic and edaphic factors. Management and harvest operations include regeneration, tending of seedling stands, pre-commercial and commercial thinnings, choice of provenance of trees, nitrogen fertilization and terminal cut (Fig. 4.5).

The regeneration, growth and mortality of trees control the dynamics of tree populations/communities and the consequent uptake and emissions of carbon dioxide (CO_2). The growth of trees is based on diameter growth, i.e., $\Delta D = \Delta D_o \times M_1 \times ,\ldots, \times M_n$, where ΔD is diameter growth (cm a^{-1}); ΔD_o is diameter growth (cm a^{-1}) in optimal conditions; and $M_1,\ldots,M_n$ are multipliers representing the temperature sum (TS; $+5\,^\circ C$ threshold), prevailing light conditions in the stand, soil moisture and nitrogen supply. Optimal conditions refer to growth under no shading and no limitation of soil moisture and nitrogen supply. The values of ΔD_o are further related to maturity of trees (diameter of tree, D cm) and the atmospheric CO_2 (Kellomäki et al. 2008):

$$\Delta D_o = \exp\left(a + \frac{b}{0.01 \times CO_2}\right) \times D \times e^{DGRO \times D} \tag{4.1}$$

where a, b and DGRO are parameters. The parameter b indicates the level of potential growth determined by provenance of tree species (Routa et al. 2012b).

Diameter is further used to calculate the mass of stem, foliage, branches and roots based on the allometric relationship between the diameter and the mass of tree organs (Mass(i, j)):

$$Mass(i,j) = \exp\left[a(i,j) + b(i,j) \times \frac{D(i)}{c(i,j) + D(i)}\right] \tag{4.2}$$

where a(i, j), b(i, j) and c(i, j) are parameters specific for tree species (i) and mass component (j). Both litter from any tree organ and mortality of trees transfer carbon and nitrogen to the soil, where litter and humus decay, emitting CO_2 and releasing nitrogen for the use of growing trees .

4.4 Effects of Commercial Thinnings and Rotation Length on Biomass Recovery

4.4.1 Yield of Energy Biomass in Relation to Thinning Practices

There are a number of ways to integrate the timing and intensity of thinning in the production cycle, with varying effects on the yields of timber and energy biomass. They may be related to the age of trees or they may be adjusted to the growth of trees as shown in Fig. 4.6. By changing the basal area threshold that triggers thinning and the threshold for remaining trees, we simulated six different thinning regimes by utilizing the ecosystem model described in Box 4.1 in order to demonstrate the effects of thinning on the yields of timber and energy biomass. In the thinning regimes, the basal area thresholds currently used in Finland were increased or decreased. The simulations were done for three different initial stand densities from 2000 to 4000 seedlings ha^{-1} for Norway spruce growing at a fertile site (*Oxalis-Myrtillus* type) and for Scots pine growing at a poor site (*Vaccinium* type) using a rotation of 80 years.

In general, the increased thinning thresholds maintained higher tree stocking throughout the production cycle which delayed successive thinnings (Fig. 4.7). This was due to the reduced rate of diameter growth at the tree level and reduced rate of basal area growth at the stand level. Increased thresholds may, thus, reduce the timber yield, if the fixed rotation length is used (Mäkinen and Isomäki 2004a, b). However, a proper timing and choice of thinning regime may provide management options where the growth and the subsequent yield of energy biomass may be increased without losing too much timber (see also Thornley and Cannell 2000; Garcia-Gonzalo et al. 2007; Alam et al. 2008, 2012). This was the case for Norway spruce if the thinning threshold was increased by 30 % from the current levels (CU + 30 %BA) (Fig. 4.7). On the other hand, decreased thinning thresholds reduced the stocking throughout the production cycle. Consequently, the yield of energy biomass in

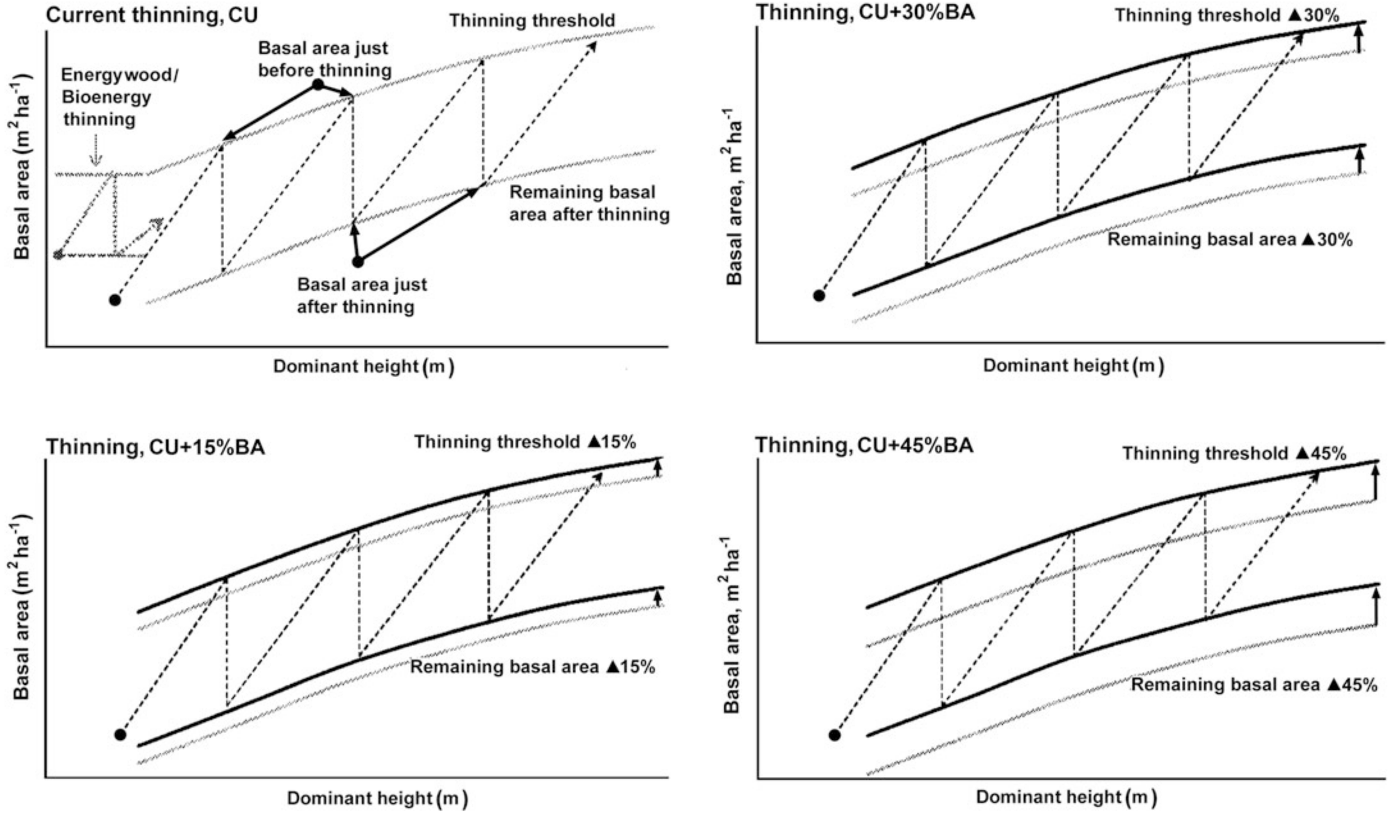


Fig. 4.6 Schematic representations of how the basal area is dependent on the dominant height and how this relationship can be used to vary timing and intensity of thinning. Legend: CU refers to the thinning regime currently recommended and CU+15%BA, CU+30%BA and CU+45%BA refers to the altered thinning regimes, with the percentage increase in basal area (BA) thinning threshold given the parenthesis. (Adapted from Alam et al. 2010)

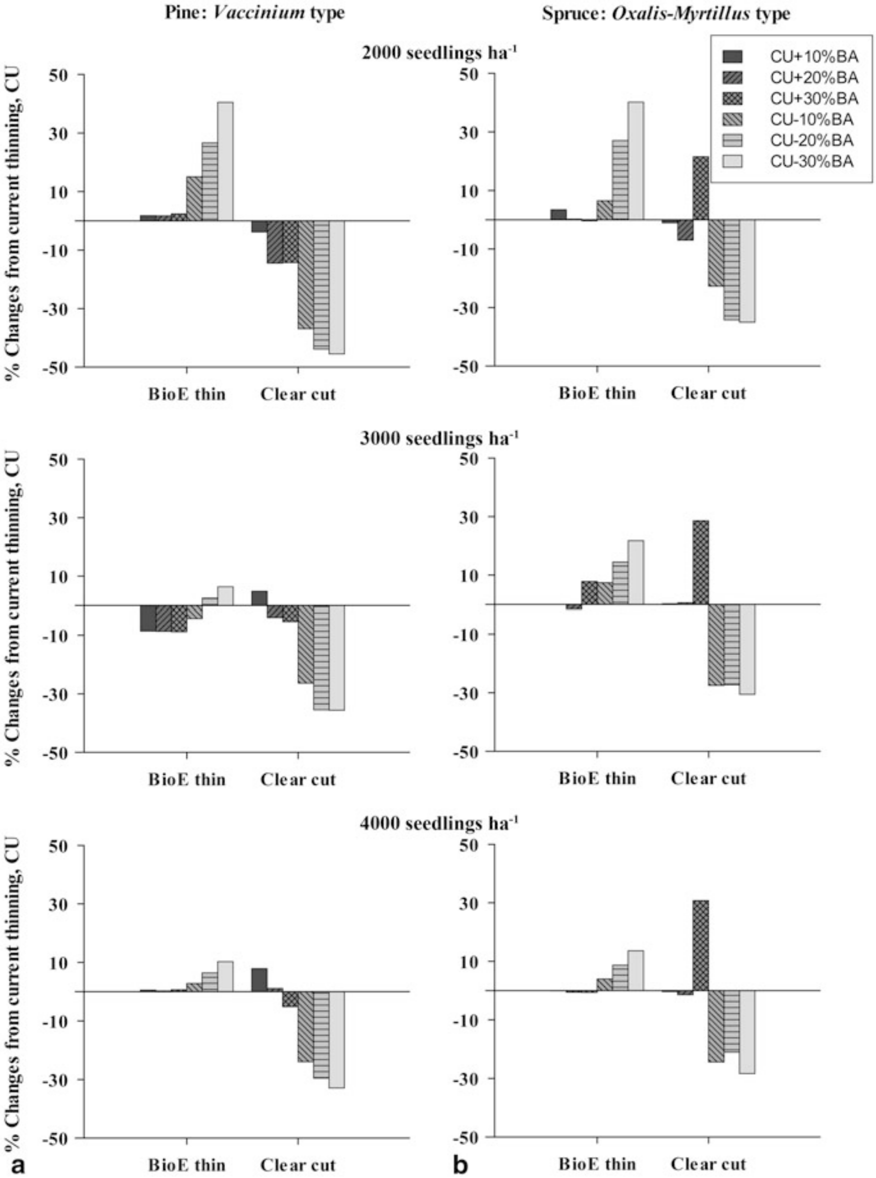


Fig. 4.7 Relative effects of initial stand density (seedlings ha^{-1}) on the yield of energy biomass in bioenergy thinning (BioE thin) and terminal cut as a function of tree species (**a**: Scots pine, **b**: Norway spruce), site fertility and thinning regime applying the 80-year rotation period. Thinning regimes represent regimes, where thinning thresholds are increased/decreased in relation to the recommended ones. For example, CU + 10 % BA indicates basal area (*BA*) thresholds increased by 10 % compared with the current one (*CU*), and CU-10 % BA denotes the basal area thresholds decreased by 10 %. The simulations are based on the model described in Box 4.1

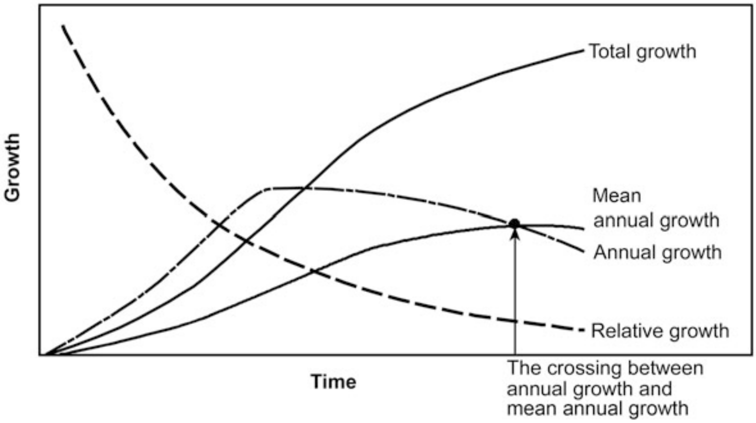


Fig. 4.8 Schematic presentations of the course of annual growth, mean annual growth, total growth and relative growth. Current growth refers to the annual growth (e.g., m^3 ha^{-1} a^{-1}), and mean annual growth to the mean growth during a selected period (e.g., m^3 ha^{-1} a^{-1}). Total growth is the sum of annual values of growth during a selected time period (e.g., m^3 ha^{-1} per period), and relative growth indicates percentage value of current growth in relation to stocking of trees (%)

bioenergy thinning was increased. Furthermore, the reduced stocking enhanced the growth of single trees, thus increasing the number of thinnings over the production cycle but reducing the timber yield at successive thinnings (see also Mäkinen and Isomäki 2004a, b; Thornley and Cannell 2000). The reduced stocking may decrease the yield of energy biomass (logging residues) at terminal cut.

4.4.2 Yield of Energy Biomass in Relation to Rotation Length

The length of the production cycle or rotation has a large effect on the yields of timber and energy biomass and the stocks of carbon in forest ecosystems. In the Nordic countries, the rotation length varies from 60 to 120 years depending on the species and the growing conditions. In general, the longer the rotation, the more carbon will be stored in the forest ecosystem. On the other hand, the carbon uptake (growth) is reduced along with increasing length of rotation, thus reducing the carbon sequestration in the forest ecosystem. The proper choice of rotation length integrated with other management practices (e.g. planting density, thinning intensity, fertilization) have a large effect on the capacity of forests to sequester and store carbon in producing timber and energy biomass.

Box 4.2 Rotation Length and Biomass Yield

Rotation length (rotation) refers to the production cycle from regeneration to terminal cut, when using stand-based management. In *biological terms*, optimum rotation length refers to the point where the slope of mean annual growth is zero (Fig. 4.8). This is equivalent to the intersection of the mean annual growth

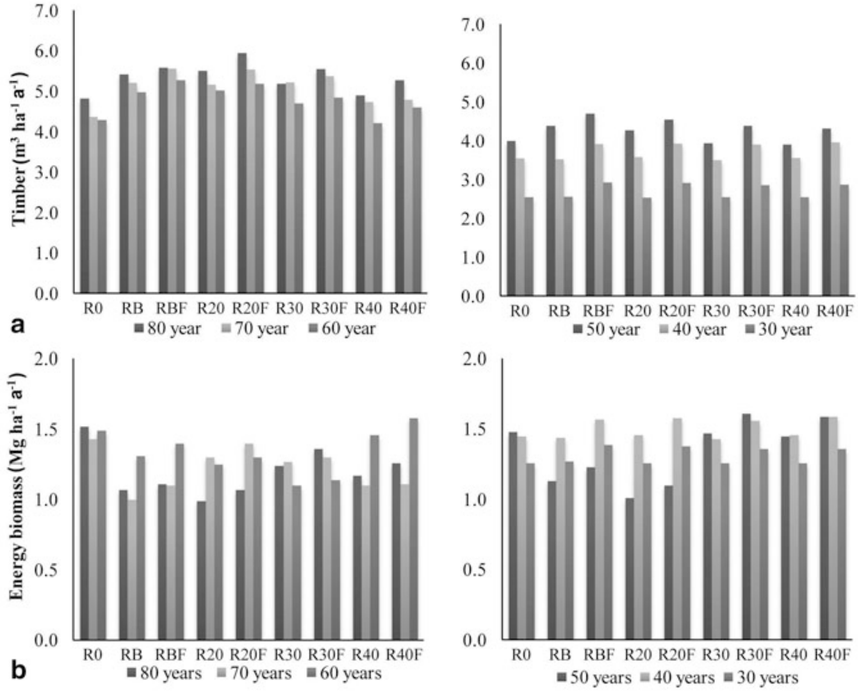


Fig. 4.9 Effects of rotation length, thinning regime and nitrogen fertilization on the mean annual yield of timber (pulpwood and sawlogs) (m^3 ha^{-1} a^{-1}) (**a**) and energy biomass (Mg ha^{-1} a^{-1}) (**b**) in Norway spruce on the fertile (*Oxalis-Myrtillus* type) site in central Finland (62° N) (Pyörälä et al. 2012). *F* fertilization (150 kg N ha^{-1}, one to two times over the rotation); *R0* only final felling used; *RB* thinnings based as currently recommended; *R20* increase of the basal area before and after thinning by 20 %; *R30* increase of the basal area before and after thinning by 30 % and *R40* increase of the basal area before and after thinning by 40 %, compared with RB. The results are based on simulation using the model outlined in Box 4.1

and the current annual growth, beyond which the mean annual growth reduces. In *economic terms*, the optimal rotation length is the one that maximizes the net present value (NPV) considering the costs and revenue of management and harvest. NPV = PVR − PVC, where PVR is the present value of revenue and PVC the present value of costs. Rotation length in economic terms covers the time from regeneration to the point where NPV is the maximum.

In general, the use of long rotations increases the annual mean yield of timber opposite to that of energy biomass (Pyörälä et al. 2012; Routa et al. 2012b) as highlighted in Fig. 4.9. The simulated values concern Norway spruce on a fertile site, with the initial spacing of 2500 seedlings ha^{-1}. The mean annual yield of timber was for long rotations (60–80 years) in the ranges of 5.2–6.2 m^3 ha^{-1} a^{-1} and 5.6–6.6 m^3 ha^{-1} a^{-1}

without and with nitrogen fertilization. The use of short rotations (30–50 years) gave lower values, in the ranges of 4.2–5.8 m^3 ha^{-1} a^{-1} and 4.4–5.9 m^3 ha^{-1} a^{-1}, which are 5–20 % less than those for the long rotations depending on fertilization. The mean annual yield of energy biomass for long rotations was 0.8–1.8 Mg ha^{-1} a^{-1} and 1.1–1.8 Mg ha^{-1} a^{-1} without and with the nitrogen fertilization, whereas the values for the short rotations were 1.3–1.8 Mg ha^{-1} a^{-1} and 1.4–1.9 Mg ha^{-1} a^{-1}, respectively. Thus, the values for short rotations were up to 60 % larger than those for the long rotations depending on fertilization. However, the timber yield remained smaller for short rotations with nitrogen fertilization than if using long rotations.

4.5 Effects of Choice of Tree Species and Provenance

The use of the most productive genetic entries in regeneration may increase the potential of biomass recovery in the integrated production of timber and energy biomass. Kilpeläinen et al. (2010) found large differences in biomass growth between the clones of Norway spruce. In southern and central Finland (60–62° N), the most productive clone produced 6 Mg ha^{-1} a^{-1} over a period of 28 years, if the planting density of 2,500 seedlings ha^{-1} was assumed (Kilpeläinen et al. 2010). This is 71 % above the average for all the clones included in the study. Over the whole period of 28 years, the total biomass production was 165 Mg ha^{-1}, which substantially exceeded the biomass yield of 43 Mg ha^{-1} typically provided by Norway spruce stands in terminal cut (Hakkila 2004). This highly productive clone showed the largest above-ground growth both in terms of stem wood and crown mass, with a high value of harvest index (stem mass/total above ground mass ratio) demonstrating a large allocation of growth into the stem. These properties make the clone a potential candidate for the combined production of timber and energy biomass.

Until now, large-scale experiences of using clonal material in biomass production are scarce in boreal conditions, and they mainly represent a time scale ($<$ 40 years) much shorter than the rotation length used in forestry. In this respect, a key question is whether young clonal trees have superior growth over longer periods compared with non-clonal trees. Regarding Norway spruce, this is probably true as demonstrated by Routa et al. (2013). Based on the consecutive measurements, they showed that the clonal differences in growth for seedlings are most probably repeated in more mature trees. Based on this finding, Routa et al. (2013) selected three clones with the highest height and diameter growth for further analysis, where the growth was compared with the simulated growth of non-clonal trees.

The height and diameter of these clones was 30–35 % above that found for non-clonal Norway spruce, if the same management was used in the simulations (Fig. 4.10). In general, the use of clonal seedlings increased both the yields of timber and energy biomass by 20–40 % and 1–22 %, depending on the management, compared with the use of non-clonal trees (Routa et al. 2013). Fertilization increased

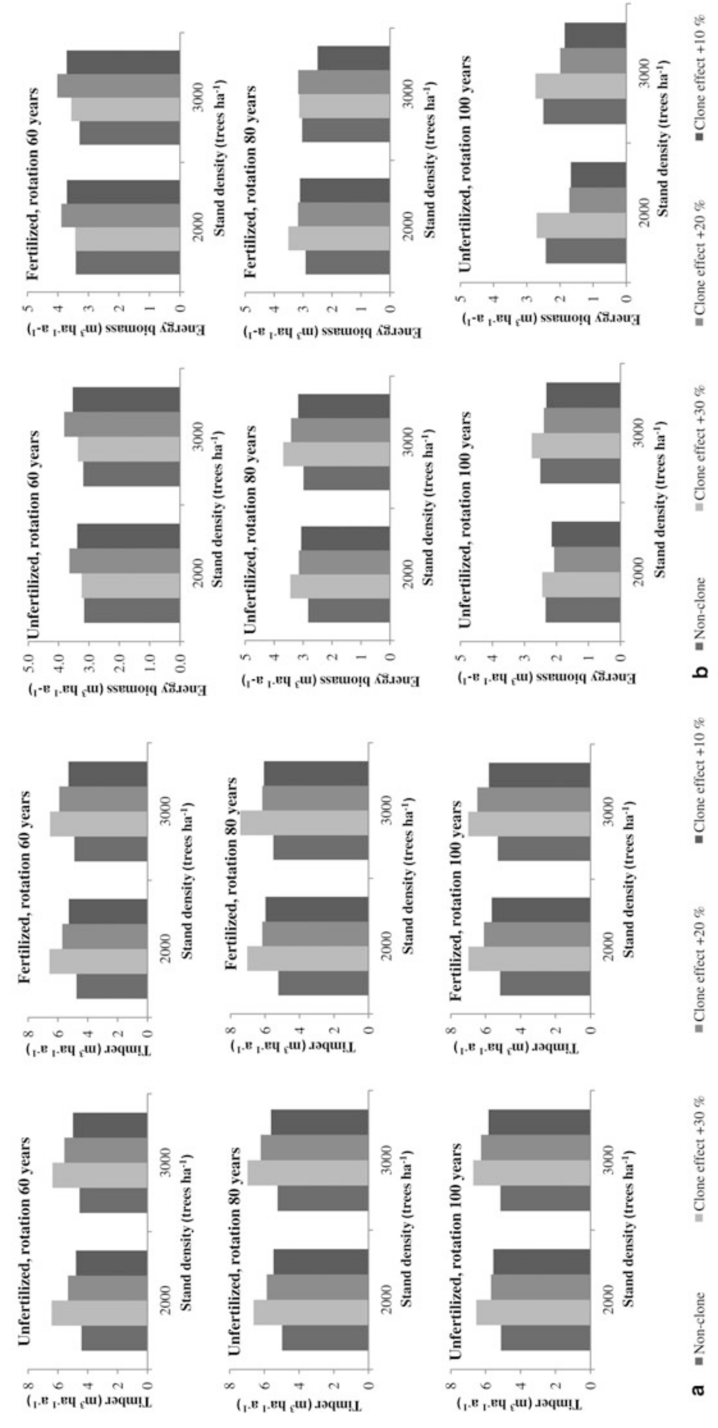


Fig. 4.10 Mean annual timber (pulpwood and sawlogs) (**a**) and energy biomass (**b**) yields ($m^3\ ha^{-1}\ a^{-1}$) as a function of initial spacing and rotation length with and without nitrogen fertilization for cloned and non-cloned Norway spruce (Routa et al. 2013). The simulations are based on the model described in Box 4.1

the production even more, but the difference in the production between cloned and non-cloned trees was reduced. Similarly, Rosvall et al. (2001) found that the use of seedlings representing seeds originating from third-generation seed orchards increased the growth of stem wood of Norway spruce by 15–25 % compared with unimproved material. However, there may be a trade-off between timber and energy biomass when using fast-growing clones in forest production. For example, Zubizarreta Gerendiain et al. (2009) found that the clones of Norway spruce with a higher growth rate had lower wood density. This problem may be solved if proper spacing is maintained throughout the rotation.

4.6 Management of Forest for Producing Carbon-neutral Energy—Case Study

4.6.1 Direct and Indirect Emissions and Carbon Neutrality

Energy based on forest biomass is widely assessed to be carbon neutral; i.e. carbon dioxide (CO_2) emissions due to burning of biomass are taken up in growth. In the long term, this is likely to occur if the forest cover remains the same as currently (Schlamadinger et al. 1995; IPCC 2000). In the short term, the combustion of biomass increases the atmospheric CO_2, which is further affected by CO_2 emissions due to the management, harvesting and logistical operations necessary to produce and supply the energy biomass. Furthermore, CO_2 is emitted from the forest ecosystem itself in the natural sink/source dynamics of carbon and depending also on management (indirect impacts). Including direct and indirect emissions in different phases of biomass supply and use, the energy based on forest biomass may not be carbon neutral, but it may reduce the CO_2 emissions per energy unit compared, for example, with energy based on coal (e.g. Searchinger et al. 2008; Melillo et al. 2009; Melin et al. 2010; Repo et al. 2011). However, there is evidence that the carbon neutrality of energy based on forest biomass may be enhanced if the management is properly adjusted to the carbon sink/source dynamics of the ecosystem.

4.6.2 Concepts and Simulations

In assessing the carbon neutrality of forest-based energy, Pyörälä et al. (2012) calculated the carbon balance for several management scenarios in producing timber and energy biomass, as discussed in Sect. 4.4. Furthermore, the carbon balance for each management scenario was calculated including the CO_2 uptake in growth, CO_2 emissions from decaying organic matter in soil, and CO_2 emissions from management, harvest and logistics and the CO_2 emissions from the combustion of the biomass. Carbon balance was further used to calculate the CO_2 emissions per unit of energy produced over a rotation period and the carbon-neutrality factor ($CN\ (t)$)

(Schlamadinger et al. 1995) for each management scenario. Carbon-neutrality factor refers to the ratio between the net reduction/increase of CO_2 emissions from using forest biomass and the CO_2 emissions from the use of coal (the reference energy system), which was to be substituted by forest biomass:

$$CN(t) = \frac{[E_r(t) - E_n(t)]}{E_r(t)} = 1 - \frac{E_n(t)}{E_r(t)} \tag{4.3}$$

where $E_r(t)$ is the CO_2 emissions from the reference energy system based on coal between the years 0 and t, and $E_n(t)$ is the CO_2 emissions from using forest biomass between the years 0 and t. The following cases can be separated: (1) $CN < 0$, if the emissions from the bioenergy system are higher than the emissions from the fossil fuel system; (2) $CN = 0$, if the emissions from both systems are equal; (3) $0 < \text{CN(t)} < 1$, if the emissions from the bioenergy system are less than from the fossil fuel system; (4) $CN = 1$, if the bioenergy system produces zero emissions in comparison with the fossil fuel system.

4.6.3 Carbon Balance

In general, the CO_2 uptake in the ecosystem exceeded the loss of CO_2 most if high stocking and long rotation were preferred in management (Pyörälä et al. 2012). The uptake exceeded the loss in a relative sense more on the fertile sites (*Oxalis-Myrtillus* type) than on the medium fertile sites (*Myrtillus* type). In the latter case, the carbon balance over the rotation varied from – 339 to – 545 g CO_2 m^{-2} for the long rotations without fertilization and from – 384 to – 509 g CO_2 m^{-2} with nitrogen fertilization (negative values indicate that CO_2 fixed in the ecosystem exceed that emitted in the supply chain and combustion). For the short rotation, the values ranged from −10 to −330 g CO_2 m^{-2} and from – 70 to – 401 g CO_2 m^{-2} without and with fertilization, respectively. On the fertile sites, the carbon balance for the long rotations without and with fertilization was between – 355 and – 662 g CO_2 m^{-2} and between – 473 and – 622 g CO_2 m^{-2}. The corresponding values for the short rotations were from – 165 to – 472 g CO_2 m^{-2} and from – 217 to – 556 g CO_2 m^{-2}, respectively.

4.6.4 Specific CO_2 Emissions per Energy unit

Pyörälä et al. (2012) found that the specific carbon emission increased along with a decrease of the carbon storage in the forest ecosystem regardless of the site fertility (Fig. 4.11). On the medium fertile sites (*Myrtillus* type), the CO_2 emissions per unit of energy ranged from 80–219 kg CO_2 MWh^{-1} for the long rotations without fertilization and from 135–191 kg CO_2 MWh^{-1} with nitrogen fertilization. The corresponding ranges for the short rotations were 231–564 and 188–485 kg CO_2

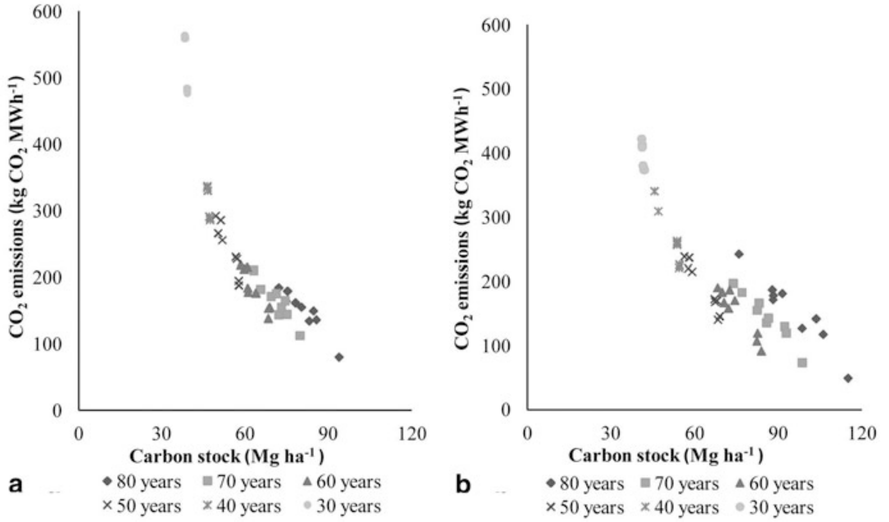


Fig. 4.11 CO_2 emissions per unit of energy (kg CO_2 MWh^{-1}) as a function of mean carbon stock in the forest ecosystem (Mg ha^{-1}) and rotation lengths on the medium fertile (*Myrtillus* type) site (**a**) and on the fertile (*Oxalis-Myrtillus* type) site (**b**). (Pyörälä et al. 2012)

MWh^{-1} without and with fertilization. On the fertile sites (*Oxalis-Myrtillus* type), the CO_2 emissions per unit of energy for the long rotations were in the range of 49–242 and 92–197 kg CO_2 MWh^{-1} without and with fertilization, respectively. In the case of short rotations, the values varied from 169 to 422 kg CO_2 MWh^{-1} and from 141 to 380 kg CO_2 MWh^{-1} without and with fertilization. In general, the use of long rotation lengths and maintenance of high stocking throughout the rotation (with higher carbon stock in forest ecosystem) decreased the CO_2 emissions per unit of energy.

4.6.5 Carbon Neutrality of Forest-based Energy in Substituting Fossil Fuels

Figure 4.12 shows that the values of carbon-neutrality factor were lower the shorter the rotation length, regardless of the site fertility. In general, the carbon neutrality of the bioenergy production chain was larger than that based on coal, except if the rotation length was short and the mean stocking was low throughout the rotation. The increase in the rotation length increased the carbon neutrality of the supply chain the most, if high mean stocking was maintained throughout the whole rotation. Nitrogen fertilization increased the carbon neutrality regardless of the site fertility and management regime applied in the simulations. This held even though the CO_2 emissions in manufacturing nitrogen fertilizers were included in the calculations. The large increase of CO_2 uptake in growth due to fertilizing greatly exceeded the CO_2 emissions due to the use of nitrogen fertilizers in management.

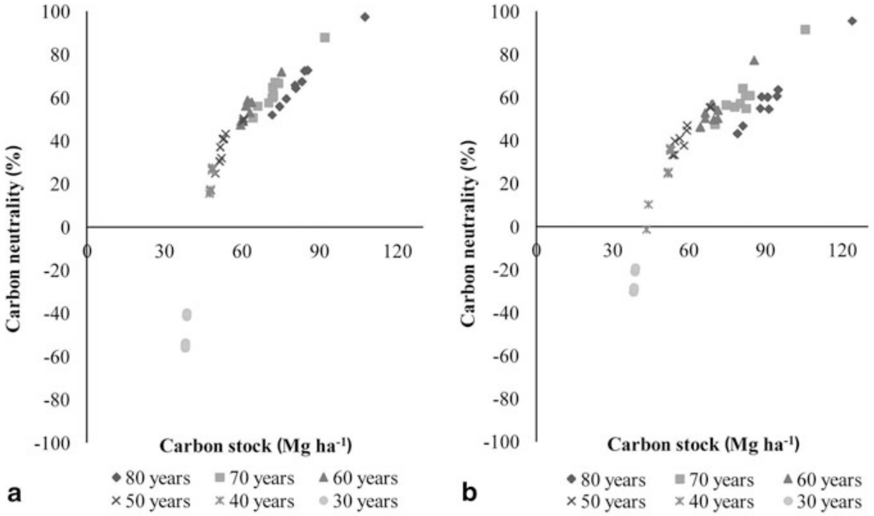


Fig. 4.12 Carbon neutrality of forest energy chain as a function of mean carbon stock in the forest ecosystem (Mg ha^{-1}) and the rotation lengths, on the medium fertile (*Myrtillus* type) site (**a**) and on the fertile (*Oxalis-Myrtillus* type) site (**b**). (Graphs modified from Pyörälä et al. 2012)

4.6.6 Carbon Neutrality and Profitability of Forest-based Energy

Currently, the forest-based energy system as such is profitable only if subsidized. This is due to the low price of energy biomass in relation to the costs of producing it. This is especially the case for pre-commercial thinning and bioenergy thinning. When combined with the production of timber, the situation may be different, especially if energy biomass is produced in final cuts, reducing the costs of harvesting and logistics. Thus, the supply chain of energy biomass is "subsidized" by the high profitability of timber production, and there is no need for additional subsidies.

The key question is still whether fossil fuels can be substituted in a profitable way by using forest biomass in energy production. In this context, Pyörälä et al. (2013, unpublished) calculated the carbon neutrality of the energy production chain based on forest biomass using the combined production of timber and energy biomass in forest management. The carbon neutrality of the management chains were related to the net present values (NPV, € ha^{-1} a^{-1}) of timber and biomass yields (Fig. 4.13). In general, high carbon neutrality could be obtained in management chains by using long rotations and by maintaining higher stocking (20–30 % increase in basal area thresholds used in producing only timber) over the rotation. This was especially the case when using nitrogen fertilization. On the other hand, the use of long rotations and fertilization provided the highest NPV. Similarly, high carbon neutrality and NPV could be obtained either by using long rotations with the baseline management or by increasing the thinning threshold values by 20 % and using fertilization. Compared with the current climate, the values of carbon neutrality and NPV were reduced to some extent under the scenario of a changing climate.

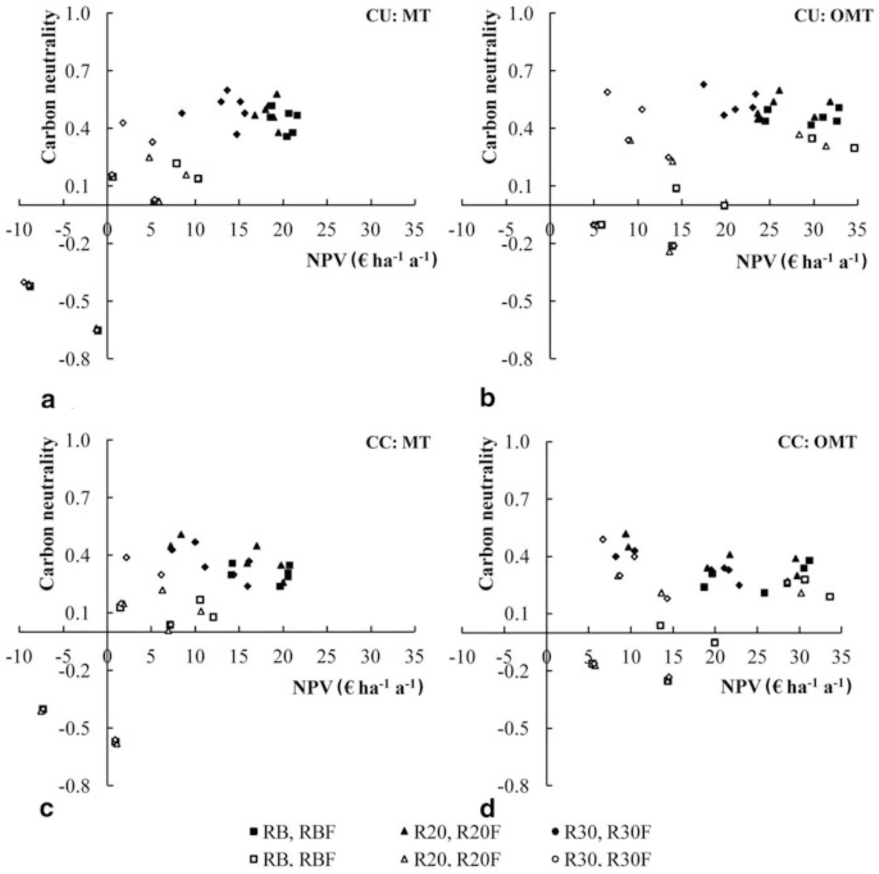


Fig. 4.13 Carbon neutrality of energy based on forest biomass as a function of the net present value at the interest rate of 3 % (*NPV*, € ha^{-1} a^{-1}) for varying rotation lengths and thinning regimes on the medium fertile (*Myrtillus* type) site (**a**, **c**) and on the fertile (*Oxalis-Myrtillus* type) site (**b**, **d**) under the current climate (*CU*) and changing climate (*CC*). *Black point* rotation lengths of 60, 70 and 80 years; *white point* rotation lengths of 30, 40 and 50 years; *square* thinning based on the Finnish Recommendations (*RBF*, *RB*) with and without nitrogen fertilization (150 kg N ha^{-1}); *triangle* 20 % increment of basal area before and after thinning (*R20F*, *R20*) with and without N fertilization (150 kg N ha^{-1}) and *circle* 30 % increment of basal area before and after thinning (*R30*) with and without N fertilization (150 kg N ha^{-1}) compared with basic thinning regime. (Pyörälä 2013, unpublished)

4.7 Discussion and Conclusions

Novel forest management systems may be needed in the future, because the current management that aims only at timber production is not necessarily appropriate in managing forests concurrently for timber and energy biomass and to maintain or even increase carbon storage in forest ecosystems. Key questions are how to integrate the management to enhance the production potentials of these products and services and what are the sustainability issues and the economic implications of such management. For example, the proper choice of spacing in regeneration and thinning, rotation length, use of improved clones and fertilization in management provides many possibilities to enhance the yields of timber and energy biomass and carbon stocks in forest ecosystems, thus enhancing the mitigation of climate change in forestry. The production and the profitability of energy biomass may be increased by using high initial spacing in the seedling phase, and by harvesting energy biomass at the first commercial thinning or prior to this. It is possible that nitrogen fertilization is needed to increase the growth and the biomass density to increase the values of NPV of energy biomass. Fertilization may be needed also to compensate for nutrient losses due to harvesting of logging residues in order to sustain the productivity of forest sites.

In general, the carbon neutrality of forest-based energy is affected by the carbon balance including the CO_2 uptake in growth and the CO_2 emissions related to the natural sink/source dynamics of the forest ecosystem. Furthermore, the CO_2 emissions due to management, harvest, logistics and other phases of the supply chain affect the carbon neutrality of forest-based energy. Pyörälä et al. (2013, unpublished) found that most of the management scenarios reduced the average CO_2 emissions per unit of energy compared with the values obtained by using coal in energy production. In this respect, the most positive effects were obtained if long rotations with high stocking and nitrogen fertilization were combined in management regardless of climate scenario. The mechanisms behind this were related to increased carbon sequestration and carbon stocks in the forest ecosystem, with the improved carbon balance in forestry. Pyörälä et al. (2013, unpublished) emphasize that "the positive effects of longer rotation length on carbon neutrality are due to the fact that in the early phase of rotation the CO_2 emissions from the soil will substantially exceed the CO_2 uptake in young trees, unlike in the later phases of rotation". However, maximizing NPV and carbon neutrality of the production chains simultaneously was, in general, not possible; i.e. high carbon sequestration and carbon stocks in the forest ecosystem provided high carbon neutrality, but not high NPV, and vice versa. However, some management regimes may provide simultaneously high carbon neutrality and NPV, which points to possibilities to develop management practices that balance the needs simultaneously to produce timber and energy biomass and to mitigate climate change in forestry.

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Chapter 5
Nutrient Management for Sustainable Production of Energy Biomass in Boreal Forests

Heljä-Sisko Helmisaari and Lilli Kaarakka

Abstract Long-term experiments have shown that biomass harvest may change the biogeochemical cycles of nutrients in forest ecosystems, especially when nutrient-rich logging residues are harvested. Compared with stem-only harvest, the harvest of logging residues increases the removal of nitrogen and phosphorus. These nutrients are the most limiting nutrients in the northern boreal forests, and their removal may have long-term effects on growth and biomass production. Therefore, the nutrient management of forest soil is among the key problems to be solved, when using forest biomass in energy production. In this chapter, the impacts of biomass harvest on the availability of nutrients and the growth of trees are discussed in order to outline how to avoid harmful effects of intensive use of forest biomass in energy production.

Keywords Biomass harvest · Effects on nutrients · Nutrient management · Boreal forests · Effects on growth

5.1 Harvest of Forest Biomass and Productivity of Forest Ecosystems

Forest biomass is increasingly being used as an energy source in Europe, following the targets for renewable energy in 2020 set by the European Union for reducing the emissions of greenhouse gases (European Commission 2008). Especially, in the forested regions of Europe the utilization of forest biomass improves also national energy security (Stupak et al. 2007). Many governments worldwide have similar targets and emphasize the role of forest biomass in energy production. Harvest of forest biomass for energy use has been studied widely, especially in Europe and North America. An important study area is the long-term sustainability of bioenergy based on logging residues including foliage, branches and even stumps. In this regard, there

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Fig. 5.1 Stumps waiting for transport (Photo:Heljä-Sisko Helmisaari)

are several long-term experiments providing data for studying the ecological effects of harvesting logging residues, e.g. on the productivity of forest sites, excluding the impacts of stump extraction (Fig. 5.1). The effects on stump extraction on the nutrient cycle in the forest ecosystem are still very poorly known.

Long-term experiments help to identify the conditions under which biomass harvest is likely to deteriorate the forest growth (Thiffault et al. 2011). Obviously, the harvest may disturb the biogeochemical nutrient cycles and reduce the nutrient supply for growth. This is especially the case when nutrient-rich residues are harvested. Compared with stem-only harvest, the harvest of logging residue increases the removal of nitrogen (N) and phosphorus (P). In particular, the availability of N is limiting the growth in the boreal forests (e.g. Mälkönen 1976; Kimmins 1977; Bonan and Shugart 1989; Burger 2002) making the productivity susceptible to the harvest of logging residues. The research to date shows that the harvest of logging residues with removal of N, especially, may have long-term effects on forest growth (Jacobson et al. 2000; Helmisaari et al. 2011), but the effects are not universal and consistent (Thiffault et al. 2011). Rather, the effects seem to be site-, soil- and practice-specific (Lattimore et al. 2009).

5.2 Harvest of Logging Residues and Effects on Physical Structure of Soil

The soil is the basis for renewable forest resources, but the soil itself is practically non-renewable. The formation of the physical structure of boreal soils with the distinct soil horizons has taken thousands of years, but the structure may be disturbed rapidly in management and harvest operations. On the other hand, organic matter originating from above- and below-ground litter accumulates slowly on mineral soil and within the soil profile. Potential impacts of harvesting forest biomass on soil include effects on physical and chemical properties of soil, with effects on the growth of trees (Lattimore et al. 2009).

Most management and harvesting operations disturb the physical and chemical properties of forest floor (Johnson et al. 1991a; Hope 2007; Brady and Weil 2008). Physical effects on soil include, for example, the exposure, compaction and erosion of soil surface, whereas chemical impacts include the loss of soil organic matter (SOM) and nutrients, and the acidification of soil (Stupak et al. 2013). The intensity of harvest operations determines, for example, the extent of disturbances (Jurgensen et al. 1997; Eliasson and Wästerlund 2007), which may be further enhanced by the traffic necessary in harvesting; i.e. the use of heavy machines compacts soil and makes ruts, thus destroying the original structure of the soil surface and profile.

In managed forest ecosystems, stumps are important stores of carbon and nutrients (Palviainen et al. 2010), which are lost when stumps are harvested. At the same time, stump harvesting may further enhance the soil compaction and mixing, with an increase in soil erosion and a decrease in the growth of trees planted in logging areas. Furthermore, stump removal might temporarily increase nutrient leaching out of the site in the groundwater and surface flows. Similar problems are evident when logging residues are no longer available to form protective mats on skidding trails as is the case in stem-only harvest. The protective role of logging residues is especially important on wet or moist soils. On the other hand, soil damages may be decreased through active measures, such as the reinforcement of the soil surface and improved planning of strip roads (Björheden 2013). Furthermore, logging machines with low surface pressure will reduce the risk of disturbing the soil surface and profile.

Regarding the harvesting of stumps, the British guidelines divide soils into three risk categories: low, medium or high. In the high-risk areas, only restricted stump harvesting is accepted, whereas on low- and medium-risk sites brash mats are recommended to be left to strengthen extraction routes. This is especially the case when there is a prolonged time gap between timber harvest and stump harvest, with a consequent reduction of bearing capacity of soil (Nisbet et al. 2009). The Finnish guidelines emphasize that logging operations should be carefully planned in order to avoid damage to the remaining trees and soil. Furthermore, it is important to make sufficiently broad (4–4.5 m) strip roads and leave enough logging residues to support soil (Äijälä et al. 2010).

In the Nordic countries, clear cut with planting is widely used in forest regeneration. This is especially the case when regenerating Norway spruce (Ylitalo 2011). In such cases, the timber logging is done with harvesters, which pile the logging residues on the site next to the skid trails. Thereafter, forwarders are used to transport residues to the road side for transporting to energy plants (Nurmi 2007). Harvesters and forwarders disturb the forest floor and upper layers of the mineral soil, thus mixing and redistributing organic and mineral materials in the soil profile (Johnson et al. 1991b). The extent of disturbance varies from light scarification to deep wheel ruts. Soil may be further disturbed by preparing soil (e.g. scarification, ploughing, mounding) for planting, wherever wet soil and/or competing grasses and herbs may reduce the survival and growth of seedlings (Hallsby and Örlander 2004). Physical disturbances are especially large in the areas, where harvest of logging residues and stump

Fig. 5.2 Stump uplifting site (Photo: Mikko Nurmi)

extraction are followed by soil preparation for planting operations. The effects of such a combination are much greater than the single effects of harvesting logging residues, extracting stumps or preparing soil alone. In the scarification, for example, 60 % of the soil surface remains undisturbed, but only 30 % if the site is scarified after extracting stumps (Fig. 5.2).

5.3 Harvest of Logging Residues and Effects on Nutrient Availability

5.3.1 Impacts on Soil Organic Matter and Carbon Content in Soil

In the boreal conditions, SOM binds a large amount of nutrients released in the decay of SOM and taken up to growth and recycled back to soil in litterfall. Thus, the harvest of timber and logging residues has direct effects on the amount of carbon and nutrients in the site and the availability of nutrients for growth. The harvest intensity may vary from the harvest of stem only to the harvest of whole biomass including foliage, branches and the top part of stem (WTH). Furthermore, stumps and coarse roots may be extracted, which further increases the removal of carbon and nutrients from site. However, the findings on the effects of WTH on soil carbon are conflicting. For example, Bélanger et al. (2003) found that the carbon content in forest floor was reduced in three years since WTH, whereas Johnson et al. (1991b) reported a higher SOM content in soil after WTH. In the latter case, the increase of SOM in the soil profile was due to the mixing of soil horizons during harvest, with no net loss of SOM. However, the amount of carbon in the humus layer in clear cut areas previously dominated by Norway spruce has been reported to decrease, but deeper in the soil profile the amount of carbon increased (Olsson et al. 1996b); i.e. SOM was partly decomposed and partly transported downwards in the soil profile after harvesting. In general, the harvest intensity seems to have little or no effect on the SOM content in the soil (Johnson et al. 1991b, 2002; Olsson et al. 1996b; Walmsley et al. 2009; Vanguelova et al. 2010; Wall and Hytönen 2011).

5.3.2 Nitrogen Cycle and Nitrogen Availability

5.3.2.1 Nitrogen Cycle in Stand Level

In the boreal forests, the main N cycling occurs through the biological cycle including uptake, litter fall and decay of SOM with the mineralization of N for further use in growth. Much of N in the within-ecosystem cycle is bound in foliage and branches (Likens et al. 1970; Helmisaari 1995; Piirainen 2002; Palviainen et al. 2004b). This makes the N cycle directly susceptible to the removal of logging residues. The quantity and timing of litterfall varies depending on the season and tree species (Ukonmaanaho et al. 2008), whereas the lignin (L) content or the L/N -ratio of litter has a strong effect on the decomposition of SOM and mineralization of N (Prescott et al. 2000). The C/N ratio in decaying litter or logging residues largely determines whether N is mineralized for direct use or immobilized in microbial biomass, where it may retained for several years before release (Brady and Weil 2008). In the boreal conditions, the high C/N ratio and cold climate slow the decomposition and promote the immobilization of N, thus reducing the risk of leaching of N outside the site.

Regarding Norway spruce and Scots pine, Ukonmaanaho et al. (2008) found that half of N in logging residues is in needles and half in branches. However, the decay rates of needles and branches are different affecting the release of N from logging residues. For example, Palviainen et al. (2004b) found that the mass of needles of Norway spruce reduced by 40 % in three years after clear cut, whereas the mass of branches was reduced only by 20 % in the same time (see also Smolander et al. 2008). The N retention in logging residues depends on the properties of residues, long-term effects on the N cycle as demonstrated by Hyvönen et al. (2000). They found that the N content in the needles of Scots pine and Norway spruce was 30 and 50 % of their the initial content after 6–8 years of decay. At the same time, no net release of N from branches occurred, thus implying that the released N was taken up efficiently by the microbial community. More than 20 years passed before half of the initial N was released from the logging residues (Hyvönen et al. 2000). However, N in logging residues seems to become available earlier on fertile sites than on poor sites (Johansson 1994).

5.3.2.2 Removal of Nitrogen in Harvest of Logging Residues

In logging residues, needles have their original nutrient content, as no senescence and redistribution of nutrients has occurred as is the case of needles falling after natural senescence. Thus, needles in logging residues left on site represent quantitatively and qualitatively larger N source than needle litter. Needles in logging residue are, a slow-release fertilizer, wherever N deficiency limits tree growth. The N concentrations in logging residue needles vary from 12–14 g N kg^{-1} of needle mass, whereas the N content in needle litter is 5–7 g N kg^{-1} of needle mass (Ukonmaanaho et al. 2008). This difference is due to the redistribution of N to the storage tissues of tree from senescing needles. Green needles in logging residues contain N compounds that can be taken up, either after the mineralization or directly, as is the case for amino acids of proteins. In the needle litter, N is tightly bound into the needle structures and released slowly in the decay of SOM.

The removal of N in needles integrated in residues in a single harvest event equals the amount of N bound in needle litter fallen over three to eight years, if WTH is used in thinning Scots pine and Norway spruce (Helmisaari et al. 2011). The removal of logging residue may especially affect the available N pool, whereas the effects may not be detected in the large N pool in the old, poorly decomposed SOM (Rosenberg and Jacobson 2004). Obviously, the removal of logging residue does not disturb the total N pool in soil in short term (Wall 2008), as one may conclude based on studies lasted only a few years. However, the slow mobilization of N in logging residue shows that the removal of logging residues may have long-term effects on the productivity of the site (Helmisaari et al. 2011).

Nitrogen removed in logging residue may be only partly compensated by the biological N fixation. In the boreal forests mosses and lichens fix 1.5–2 kg N ha^{-1} a^{-1} (DeLuca et al. 2002). Similarly, the anthropogenic N input in the atmospheric deposition is also low in the boreal conditions (2–6 kg N ha^{-1} a^{-1}), but it increases substantially towards the hemi-boreal and the temperate regions up to more than 10 kg N ha^{-1} a^{-1}. Also, the outputs of N in leaching or gaseous emissions are low, but the leaching may be high in the vicinity of industrialized areas leading to a more open N cycling.

5.3.3 Base Cations and Phosphorus

Weathering is the major source of phosphorus (P), calcium (Ca), potassium (K) and magnesium (Mg) in the boreal soils (Likens and Bormann 1995; Starr et al. 1998; Palviainen et al. 2012), particularly in the areas with low atmospheric deposition as in the Nordic countries (Ruoho-Airola et al. 2003). Ca and Mg are both relatively abundant in forest soils, and they are bound in the structures of soil minerals. They are in cation form in soil solutes (i.e. plant-available), and only a small fraction of them is bound in vegetation (Likens and Bormann 1995). In contrast, K remains in the ionic form in the soil complex and in plant material, thus making it mobile (Olsson et al. 1996a; Brady and Weil 2008). After clear cut, K is released rapidly from the decomposing residue (Fahey et al. 1991; Palviainen et al. 2004a). Base cations are important nutrients for plants, but they are also neutralizers in soils naturally susceptible to acidification (Starr et al. 1998). Ca could enhance decomposition by promoting the growth of lignin-decaying fungi (Johansson 1994).

In clear cut areas, the release of phosphorus from logging residues is rapid just after cut (Palviainen et al. 2004b), but P is effectively retained in the mineral soil as a component of metal-organic acid complexes (Piirainen et al. 2004). Almost all K is released from the logging residue during the first year, whereas Ca is released relatively slowly (Olsson et al. 1996a; Palviainen et al. 2004a). Ca is effectively retained in the woody litter for a longer time, implying that branches could provide a longer-term source of Ca (Fahey et al. 1991; Palviainen et al. 2004a). A large proportion of Ca released is bound in the floor and vegetation (Palviainen et al. 2005; Thiffault et al. 2006; Vanguelova et al. 2010).

Several studies have shown that WTH negatively affects the base cation pools (Nykvist and Rosen 1985; Johnson et al. 1991a; Olsson et al. 1996a; Rosenberg and Jacobson 2004; Thiffault et al. 2006; Vanguelova et al. 2010) and that there could be a risk of depletion of soil cations during only one rotation due to intensive biomass harvest (Sverdrup and Rosen 1998; Akselsson et al. 2007). However, these conclusions have been criticized because the base cations such as K and Mg and, particularly, Ca are taken up in excess by plants. On the other hand, the mass balances of these base cations are strongly affected by the weathering processes, which is difficult to be quantified as regards the differences between sites of varying fertility (Egnell 2013), increasing the uncertainty of mass balance calculations.

Harvesting of timber and biomass, in general, tends to acidify forest soil (Olsson et al. 1996a; Burger 2002). However, soils of fertile sites (i.e. more productive sites) are more resistant to the changes in acidity due to their high buffering capacity than soils representing low fertility. In general, WTH may reduce the cation exchange capacity (CEC), increase the exchangeable acidity (EA) and decrease the base saturation (BS) in the soil profile (Johnson et al. 1991a; Olsson et al. 1996a; Bélanger et al. 2003; Thiffault et al. 2006). On the other hand, no changes in soil pH after WTH have been found (Rosenberg and Jacobson 2004; Thiffault et al. 2006; Wall 2008; Saarsalmi et al. 2010). Changes in pH would be reflected in the proportional saturation of exchange capacity with acid and base cations. The exchange sites in the organic layer are predominantly pH-dependent, thus making this layer more sensitive to the changes in pH than mineral soil (Brady and Weil 2008).

5.4 Harvest of Logging Residues and Effects on Tree Growth

In the boreal conditions, WTH may reduce the growth of remaining trees due to nutrient losses in thinnings (Jacobson et al. 2000; Helmisaari et al. 2011) and the growth of trees planted after a clear cut (Proe and Dutch 1994; Walmsley et al. 2009; Wall and Hytönen 2011). However, the reduction is probably small extending the rotation by one to two years (Mattsson 1999), if the maturity for final cut is indicated by the given diameter of trees. For example, Helmisaari et al. (2011) reported a tendency towards lower basal area increment during the first and second 10-year period, when WTH was used once in thinning of Norway spruce and Scots pine stands. This also held for the stands thinned twice. However, the variation in the growth reaction to WTH was wide among and within experiments (Helmisaari et al. 2011). Regarding Norway spruce, there was a negative correlation between the mass of harvested residue and the relative growth after the second thinning; i.e. the more logging residue was harvested, the lower was the relative increment (Helmisaari et al. 2011). If logging residues were left at the site in stem-only harvesting, slowly decomposing logging residues from two thinnings provided a constant availability of nutrients, especially of N.

5.5 Nutrient Management

5.5.1 Residue Retention

In the boreal forests, the surface soil provides the main part of the nutrients available for growth. This makes the impacts of WTH on nutrient cycles particularly interesting from the management perspective (Worrell and Hampson 1997; Prescott et al. 2000; Raulund-Rasmussen et al. 2008; Palviainen and Finér 2012); i.e. how the inputs and outputs of nutrients are balanced under varying management regimes and how WTH is affecting the balances.

There is evidence that the use of WTH could reduce base cations substantially in relation to input even during one rotation (Sverdrup and Rosen 1998; Akselsson et al. 2007). The same has been found in the model simulations (Palviainen and Finér 2012); i.e. the removals in WTH exceeded the natural inputs during one harvest rotation, even though losses were partly compensated through weathering and atmospheric deposition (see also Sverdrup and Rosen 1998; Akselsson et al. 2007; Palviainen and Finér 2012). Thus, long rotation could buffer the nutrient balances against the nutrient losses (Kimmins 1977; Worrell and Hampson 1997). In this respect, the benefits are probably small in the Nordic countries, where the rotation times are already rather long (more than 60 years).

Several studies report that the impacts of WTH on tree growth are site- and species-specific (Olsson et al. 1996b; Thiffault et al. 2006; Raulund-Rasmussen et al. 2008; Wall and Hytönen 2011), thus indicating site-specific benefits from leaving all or part of residues at the site. Smolander et al. (2010), for example, found higher C-mineralization/N-mineralization ratios when the logging residues were left on the site than when they were removed. This indicated that the presence of residues facilitated the mineralization of N, which was especially important on poor sites. The benefits of retaining logging residue on the site might, however, be limited to the sites with small SOM pools (Thiffault et al. 2006; Tamminen et al. 2012); i.e. where a large proportion of nutrients are bound in trees, as on the sites of low fertility. The possibility of leaving green needles on the site (seasoning) for a while after clear cut is a potential method to reduce the removal of nutrients, but the results have been inconclusive (Olsson et al. 1996b; Wall 2008). On the other hand, the removal of logging residue in several phases would increase the traffic, with further compaction and damage to the forest floor. Growing coniferous and deciduous species in mixtures may reduce the detrimental effects of WTH, because the nutrient content in soil under such mixtures is high (Paré et al. 2002; Thelin et al. 2002). The use of this strategy is, however, limited only to the sites where the fertility allows both coniferous and deciduous species to grow successfully.

5.5.2 *Fertilization*

There is clear evidence that the growth reduction due to WTH can be ameliorated through proper fertilizing (NPK), which compensates for nutrient losses (Olsson 1999; Jacobson et al. 2000). For example, Helmisaari et al. (2011) reported that compensatory fertilization (WTH + CoF) slightly increased the growth covering the losses due to the harvest of logging residues. Their results also showed that the removal of nutrients, especially N, in harvested residue was the most probable reason for growth losses due to WTH. A normal fertilization after both conventional stem-only (CH + F) and whole-tree thinning (WTH + F) clearly increased the growth in the first 10-year period, relatively more in Scots pine than in Norway spruce. The effects of fertilization repeated later tended to be smaller than that after the first thinning event. Helmisaari et al. (2011) reported that the positive response to a single addition of fertilizer lasted only the first 10-year period, and thereafter the effect was negative during the second 10-year period. However, the total growth response over the whole 20-year monitoring period was, on average, positive. This temporal pattern may be connected to the changes in biomass allocation induced by fertilization.

5.6 Discussion and Conclusions

Based on the experimental data across the Nordic countries, Jacobson et al. (2000) concluded that the possible growth reduction after WTH in thinning could be the result of reduced N supply. The reduction of growth could also be the result of indirect effects, such as changes in the microclimate or in the competition due to ground vegetation (Jacobson et al. 2000). Furthermore, Fahey et al. (1991) suggested that the ground vegetation could act as a nutrient sink after WTH, thus reducing the direct availability of nutrients for tree growth. Until now, no effects of harvest intensity on the abundance and cover of ground vegetation have been found (Olsson and Staaf 1995; Rosenberg and Jacobson 2004).

Obviously, the effects of WTH are site-, species- and practice-specific, and largely dependent on the demand for and availability of nutrients. Therefore, reduced growth may occur if the availability of N is scarce, whereas WTH may not reduce growth where there is an ample supply of N (Nord-Larsen 2002). While fertilization may be used to sustain the nutrient availability, leaving residue needles on site is beneficial as regards the maintenance of organic matter in the soil and returning nutrients to the soil. Furthermore, the maturity of trees or tree stands affects the loss of nutrient in WTH. Typically, the proportion of nutrient-rich foliage and bark in tree biomass decreases along with the maturity of trees or stand age (Mälkönen 1976; Augusto et al. 2000; Helmisaari et al. 2009, 2011), thus making sites with trees in thinning phase more sensitive to the removal of logging residue than sites with mature trees. According to Mälkönen (1976), a Norway spruce crown contains 30 % of the total dry mass but more than 65 % of the total nutrients at the thinning phase, making the removal of logging residue disproportionate to the nutrient demand of trees. Thus, WTH in thinning phase is likely to have more detrimental effects on the availability of nutrient than the harvest of logging residues from clear cut areas dominated by mature trees.

Scientific research utilizes results from controlled experiments, which are not entirely applicable in practical forestry (Egnell 2013; Tamminen et al. 2012). In practice, approximately 20–30 % of the logging residues, for example, are left on the harvest sites due to operational limitations (Nurmi 2007; Peltola et al. 2011). This implies that a similar proportion of nutrients are left in the site bound in logging residues. However, it is still open how much harvesting residue should be retained in order to sustain the site fertility. In North America, for example, the legal obligation is to leave 15–35 % of harvesting residue on the site (Titus et al. 2013), whereas in Finland the recommendation is to leave 30 % of logging residues evenly distributed in any site. Furthermore, WTH is not recommended on dry and poor sites or on the sites where Norway spruce accounts for over 75 % of stand density before thinning. Correspondingly, stump harvesting and WTH in final felling are not recommended on dry and poor sites. Furthermore, stump harvesting is not recommended in areas that are important sources of ground water (Äijälä et al. 2010). It is evident that the nutrient management must be optimized by considering the specific features of trees and sites in order to develop effective guidelines for sustaining nutrient management in intensive biomass harvesting.

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Chapter 6
Fertilization in Boreal and Temperate Forests and the Potential for Biomass Production

Per-Ola Hedwall and Johan Bergh

Abstract Forest fertilization is one of the most efficient methods in forest management to increase the short-term production of biomass. In this context, this chapter provides a brief background of the physiological response of trees to fertilization and increased nutrient availability. Furthermore, we shall describe different fertilization regimes and demonstrate the potentials of fertilization in enhancing biomass production, which will be performed by presenting relevant literature and some unpublished results. This chapter will also elaborate on some ideas for developing fertilization in operational forestry.

Keywords Biodiversity · Carbon balance · Climate change · Forest fertilization · Forest management · Nitrogen · Nutrient deficiency · Nutrient leakage · Nutrient optimization · Renewable energy · Tree growth

6.1 Fertilization in Enhancing Biomass Growth in Forestry

One of the most imminent challenges facing the forestry sector is producing raw forest material for pulp, paper and construction and, in particular, to meet the increasing demand for raw materials for renewable energy (e.g., European Commission 2009). At the same time, forest resources and forest land should be allocated to ecotourism, recreation, wildlife, water quality, biological diversity to conservation of unique natural values. Therefore, more wood has to be produced from a continuously shrinking area. The improved silvicultural methods currently available for boreal and temperate ecosystems can increase growth considerably (Larsson et al. 2009; Nilsson et al. 2011). Fertilization is one option to increase the production and harvest of biomass, both in the short- and long-term perspectives.

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A large share of tree biomass is located in leaves, branches, roots and the stump. All of these parts can be used to replace fossil fuels or for other purposes and are thus of interest when discussing the potential of fertilization on forest biomass production. Traditionally, research in silviculture has been focused on the production of stem wood, and forest fertilization makes no exception, although it may induce changes in the relative allocation of biomass between different components. However, there is still a strong correlation between the parts within trees of one species, which implies that the fertilization described here will also have effects on other parts of the trees.

6.2 Why Does Fertilization Improve Tree Growth?

Productivity in boreal and temperate forest is almost without exception limited by nutrients, primarily nitrogen (Tamm 1991; Tamm et al. 1999). Consequently, this limitation can be reduced by the supply of nitrogen. Biomass production in a forest stand is determined by the amount of light absorbed during the growing season and the conversion efficiency of the absorbed light into woody tissues (Cannell 1989). If the leaf area level in a stand is low to moderate due to low fertility and/or thinning, the additions of limiting nutrients will typically result in substantial increases in leaf area and light absorption (Linder 1987; Sampson and Allen 1999).

Figure 6.1 shows an aerial photograph of the forest fertilization experiment in Flakaliden (northern Sweden). The areas with intensively green (indicating a shift in needle color and a higher canopy cover and leaf area index) are fertilized annually with a multi-nutrient fertilizer. For example, Brockley (2010) found a strong linear relationship between leaf area index and stem wood increment in an experiment using five different fertilization treatments. Wood production was a direct function of the leaf area across the treatments (Fig. 6.2). Field experiments have also demonstrated that increased stem wood production is a result of a shift in the allocation of fixed carbon, the proportionately larger part of which is partitioned to above ground parts of the tree (Linder and Axelsson 1982; Albaugh et al. 1998; Bergh et al. 1999). In addition, fertilization enhances light use efficiency, including greater photosynthesis rates (Roberntz and Stockfors 1998). The potential to increase leaf area is therefore larger in young open stands compared with older more closed stands, which affects nutrient requirements and, consequently, fertilization regimes.

6.3 Effect of Nitrogen and Other Nutrients

According to Liebig's law of the minimum only one element is limiting to plant growth at one time. This idea implies that until a sufficient supply of this main deficient element is achieved, no additional growth will be gained from adding other nutrients.

Nitrogen (N) is commonly the main limiting nutrient in the boreal and temperate forests. Additional growth effects have been achieved when N has been combined with phosphorus (P) and potassium (K) (Albrektson et al. 1977; Harrington and Wierman 1990; Dralle and Larsen 1995; Tamm et al. 1999) separately and in combination. However, the effects of additional nutrients may be restricted to, or at least be

Fig. 6.1 Aerial photo of the Flakaliden experimental forest. Parts of this Norway spruce forest have been fertilized (*dark green areas*) since the middle of the 1980s. (Reprinted by courtesy of Sune Linder)

more significant, in younger forests (Nilsen 2001), and numerous studies have been unable to demonstrate any additional effect for P and K supplementation (Jacobson and Petterson 2001; Nilsen and Abrahamsen 2003; Newton and Amponsah 2006; Harrington and Devine 2011).

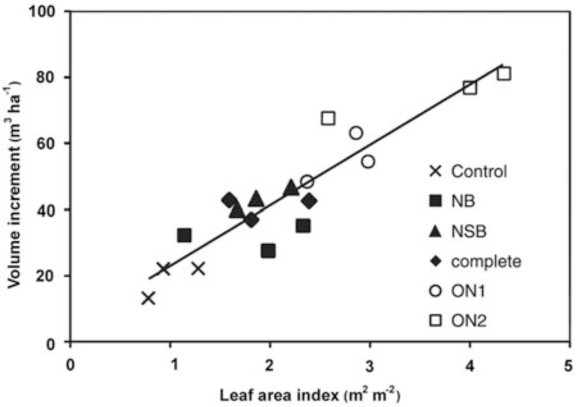


Fig. 6.2 The relationship between the leaf area index and the total stem wood increment for six different treatments, including periodic applications of nitrogen (*N*) and boron (*B*) (*NB*), nitrogen (*N*), sulfur (*S*) and boron (*B*) (*NSB*) or a multi-nutrient fertilizer (*complete*), or two different levels of nutrient optimization in a *Picea glauca* forest in BC, Canada. *Control* indicates no fertilization. (From Brockley 2010, © Canadian Science Publishing or its licensors)

Figure 6.3 shows the effects of different fertilization regimes on stem wood growth in a Norway spruce (*Picea abies)* stand in central Sweden. Two levels of N fertilization were combined with other nutrients. The lowest level of N fertilization more than doubled the increment, whereas more N did not result in any additional growth. The addition of P, K, Mg (magnesium) and Ca (calcium) increased growth at both N levels with the largest increment when the most intensive N + P fertilization was combined with the remaining nutrients. This treatment more than doubled the growth in comparison with only N and resulted in a five-fold increase in production compared with the control. The increase in tree growth by the addition of extra nutrients may be expected to decrease when the most limiting elements are supplied.

For example, Brockley (2007, 2010) observed very small or no effects for a complete fertilizer in comparison to a fertilizer supplemented with N, S (Sulfur) and B (Boron). Both treatments were repeated twice with an interval of six years and a total N supply of 400 kg ha^{-1}. There are at least four possible reasons for the inconsistency in the results from experiments dealing with the effects of additional nutrients besides N: (1) not enough N has been supplied to induce a deficiency of other nutrients; (2) the age of the stand; (3) the amounts of the additional nutrients were not large enough; and (4) the deficient nutrients were not correctly identified. Nutrient deficiencies are thus largely site dependent, and may be very hard to discover without thorough analyses. Thus a concept called nutrient optimization has been developed which is described later in this chapter.

The marginal benefit of adding more nutrients to the fertilizer can be expected to decline. This statement is true as long as a certain deficiency does not lead to decreasing tree vitality. In addition to a growth limitation in N-fertilized forests, deficiencies of other nutrients can also lead to physical damage to trees. Deficiency

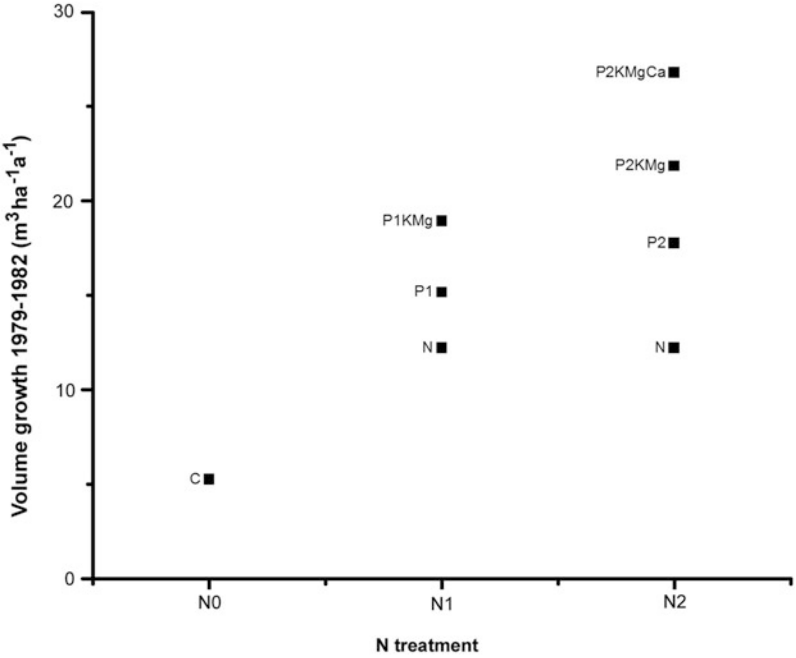


Fig. 6.3 Annual volume growths in m^3 ha^{-1} a^{-1} between 1979 and 1982 in a fertilized *Picea abies* forest in central Sweden. The figure shows the growth induced by two different *N* fertilization intensities, solely and in combination with two *P* (Phosphorus) intensities, *K* (Potassium) *Mg* (Magnesium) and *P K Mg Ca* (Calcium), together with an unfertilized control. (Redrawn from Bergh and Linder 2006. This experiment was originally described by Tamm 1991)

of B, which might occur with repeated N fertilizations on nutrient poor sites, may, for example, cause top dieback (Stone 1990), and this symptom can be reduced by B addition (Möller 1992; Sundberg 2010; Brockley 1990, 2003, 2007). Furthermore, fertilization with N alone has resulted in the increased mortality of ponderosa pine (*Pinus ponderosa*) due to the mountain pine beetle. However, the effect of mountain pine beetle was absent when N was fertilized together with K (Garrison-Johnston et al. 2005).

6.4 Nutrient Optimization

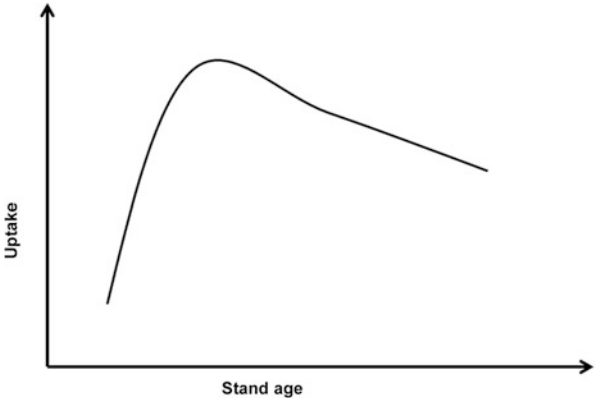
6.4.1 General

A key experimental contribution to the development of knowledge on forest tree nutrient requirements was conducted by Torsten Ingestad. His studies of forest plant growth under optimum and limited nutrient availability have laid the foundation of what is today known as nutrient optimization or balanced fertilization. He established the optimal nutritional requirements for Norway spruce and Scots pine (*Pinus sylvestris*) seedlings (e.g., Ingestad 1959, 1979a), birch (*Betula* spp) (Ingestad 1971, 1979b) and a number of other plants (Ingestad 1977). Based on these results, a complete nutrient solution was developed, which is used in many nurseries. In practice, the amount of N supplied at each occasion is determined from the needle N concentration of foliage samples, whereas the supply of other macro- and micronutrients is adjusted to initial target ratios of each element to N (Linder 1995). If the ratio of a nutrient element to N is below its target value, an extra amount is added at the next available occasion. Threshold leaf concentrations and proportions of essential nutrient elements for the attainment of unlimited growth have been determined from earlier laboratory (Ingestad 1977; Ericsson and Kähr 1993) and field experiments (Linder 1990, 1995). These proportions correspond with the recommendations provided by Braekke and Salih (2002).

6.4.2 Fertilization Rates

The efficiency of a fertilizer, measured as units of increased growth per unit of applied nutrients, is strongly dependent on the size of the application. The highest efficiency is achieved by annual applications with moderate amounts of N at each occasion of fertilization (Hyvönen et al. 2008). However, if the single dose is larger, the best efficiency can be achieved with a greater interval between the applications (Jacobson and Pettersson 2010). However, for practical and economic reasons, it is beneficial if fertilization can be performed through larger applications with an interval of a few or several years (however, also see the discussion below about environmental impacts). The efficiency of the fertilization in relation to the size of the application is, of course, strongly related to the nutrient demand and growth potential of the stand. Thus, the efficiency is strongly affected by the site conditions and by the tree species and the age of the stand (see also the discussion about fertilization in young forests below). Bergh et al. (2008) studied the effects of different fertilization intervals (one, two and three years) on growth in young stands of Norway spruce. Although the individual application and the total amount of nutrients were smaller when the forest was fertilized every second year, this treatment produced as much wood as one that was fertilized every year. However, the least intensive treatment (fertilization every third year) resulted in considerably less growth. There is a general belief that the fertilization effect on production decreases with repeated fertilizations (Jacobson and Nohrstedt 1993). However, more recent studies have demonstrated that repeated fertilizations maintain or even increase the fertilization effect if the stands are maintained unthinned (Bergh et al. submitted manuscript).

Fig. 6.4 Schematic graph indicating the relationship between stand age and nutrient uptake



6.4.3 *Fertilization in Young Forests*

As shown above, tree growth is directly connected to the size of foliage of trees and canopy of the stand. It can take several decades before the canopies of boreal and cold-temperate stands are fully closed, if ever (Bonan 1993), and considerable gains in growth can be achieved if this process is accelerated through fertilization. The uptake of and demand for N is highest in young stands, which is where the nutrient rich foliage mass is built up (Miller 1988; Chappel et al. 1991; Fig. 6.4). The demand for other nutrients in addition to N is also smaller for mature stands compared with younger stands (Johnson et al. 1982; Turner and Lambert 1986). Thus, the potential to increase leaf biomass by fertilization is greatest in young forests, where the canopies have not yet reached full closure. Accordingly, Dralle and Larsen (1995) observed a clearly negative trend in fertilization response along a gradient of tree height (5–25 m) for the Norway spruce, whereas age was indicated as an important factor for determining the fertilization response in the Douglas fir (*Pseudotsuga menziesii*) (northwest USA) by Gessel et al. (1990).

Growth and the development of needles in boreal and temperate coniferous forests can be considerably increased by the fertilization of young stands (Bennet et al. 2003; Brockley 1992, 2003, 2010). Because of the large nutrient demand in young forests, additional growth can be achieved if fertilized frequently, i.e., with one- to three-year intervals (Bergh et al. 1999, 2008; Brockley 2010). This fact does not necessarily mean that the effect of fertilization is more uncertain in middle- and old-aged stands; however, the amount and frequency of fertilization should probably be less. Likewise, forests that have previously been intensively fertilized when younger should probably also receive nutrient additions less frequently when maturing. However, there is very little knowledge about the long-term effects of fertilization on young forests and their subsequent management.

The higher N demand in young forests can also be accompanied by a deficiency of other nutrients. This deficiency can be further enhanced by N fertilization. This fact implies that although the concept of nutrient optimization is not inherently limited to

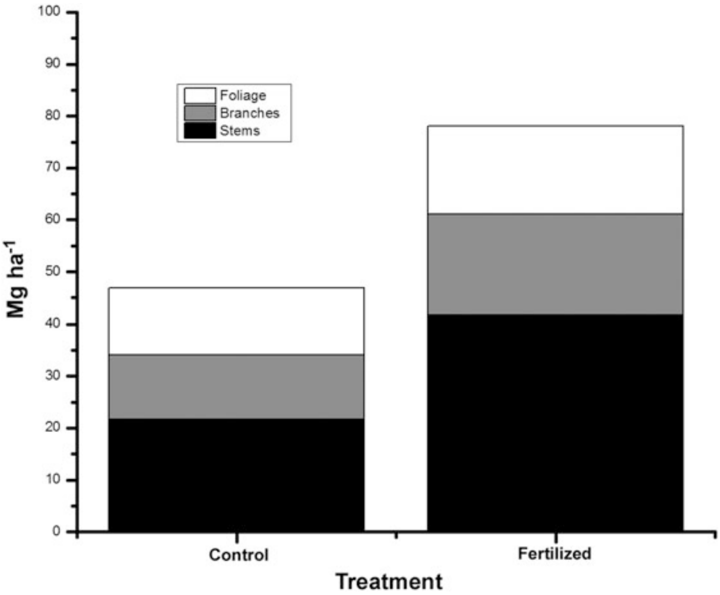


Fig. 6.5 Above-ground biomass in unfertilized young Norway spruce forests (*control*) and forests that have been fertilized every year between 2002 and 2011 at five locations in Sweden. The biomass is divided into three components: foliage, branches and stems. Tree diameters and heights were measured between the growth seasons in 2011 and 2012. The sizes of different biomass components were calculated using functions for intensively fertilized Norway spruce, developed by Albaugh et al. (2009, 2012). (For details about the experimental design, see Bergh et al. 2008)

young forests, it can be of the utmost importance at early stages of the rotation period. In British Columbia, in a lodgepole pine (*Pinus contorta*) forest (Kishchuk et al. 2002) where N alone was added, fertilization did not result in any significant growth response, whereas a mix of nutrients, determined by leaf nutrient analyses, resulted in a 48 % higher basal area. Similarly, Brockley (2007, 2010) has demonstrated that nutrient optimization can increase growth up to 277 % in white spruce (*Picea glauca*) and to a much smaller extent in lodgepole pine. Likewise, Swedish nutrient optimization experiments with another *Picea* species (*P. abies*) have demonstrated the potential to more than double growth (Bergh et al. 1999, 2008). New measurements from the experiments in the latter study after ten years of fertilization and combined with biomass functions developed for these types of forests (Albaugh et al. 2009, 2012) demonstrated that the total biomass in an unfertilized forest was 47 Mg ha^{-1} (Fig. 6.5), whereas the fertilized forest stored 67 % more biomass (78 Mg ha^{-1}). Stems and branches are the tree parts that are most feasible for extraction, whereas the foliage is typically left *in situ* because of its high nutrient content. The proportion of branches and stems was 78 % in the fertilized forest, whereas it was somewhat lower in the unfertilized forest (73 %). These results imply not only a larger biomass in the fertilized forest but also a larger extractable share.

6.4.4 Fertilization in Older Forests

Because the effect of fertilization on tree growth is dependent on the age of the forest, or rather on the space for crown expansion, we have chosen to discuss the fertilization of older forests separately. Although the response of trees in increasing leaf biomass is smaller in older forest, the physiology behind the effect is the same. In addition, more mature stands can substantially increase in production when fertilized. There are also other reasons why the fertilization of mature forests can be a practical option, among which the economic considerations are discussed later in this chapter. The majority of the fertilization studies in Scandinavia have been performed in middle-aged or older coniferous forests. Several extensive reviews of fertilization experiments in the Nordic countries (Denmark, Finland, Iceland, Norway and Sweden) were published in 2001 (Nilsen 2001; Nohrstedt 2001; Óskarsson and Sigurgeirsson 2001; Saarsalmi and Mälkönen 2001; Vejre et al. 2001) and have been summarized by Ingerslev et al. (2001).

These reviews conclude that a single application of 150 kg N ha^{-1} increases the growth of stem wood by 10–20 m^3 ha^{-1} or approximately 30 % in Norway spruce and Scots pine stands during a ten-year period in the areas with a low deposition of anthropogenic N. Similar results have been achieved in North America. Balster and Marshall (2000) measured a 25 % higher stem volume production over a seven- to eight-year period after a single application of 178 kg N ha^{-1} in 85-year-old Douglas fir (*Pseudotsuga menziesii*) stands in the northwestern USA, whereas, in 45-year-old jack pine (*Pinus banksiana*) in Ontario, fertilization (a single application of 336 kg N ha^{-1}) induced 2 m^3 ha^{-1} a^{-1} of extra growth (Groot et al. 1984). These numbers were later confirmed in a meta-analytical review by Newton and Amponsah (2006) for semi-mature (21- to 100-year old) black spruce (*Picea mariana*) and jack pine stands in Canada. Here, an application of 196–252 kg N ha^{-1} resulted in between 6 and 23 % higher growth during a ten-year period.

6.4.5 Carbon Balance and Climate Change Mitigation

Fertilization is also known to be positive for the overall carbon balance and to mitigate climate change (Sathre et al. 2010; Routa et al. 2013), partially through increased growth and the sequestration of carbon but also through the use of the additional growth for bioenergy purposes and increased possibilities to substitute coal, oil and natural gas. The production of conventional fertilizer and the transport and spreading of the fertilizer in a forest leads to the emission of greenhouse gases but only 5–10 % of the amount of carbon dioxide (CO_2 or CO_2 equivalents), which is taken up by the additional growth caused by the fertilization (Sathre et al. 2010). In addition to these direct effects, N fertilization increases the N content in forest soils because a major part of the added fertilizer is assimilated by microorganisms and organic materials, which also leads to increased carbon accumulation in the soil (Hyvönen et al. 2007, 2008).

6.5 Forest Fertilization in Practice

6.5.1 General

Forests supply humanity with an immense numbers of amenities (e.g., Gamfeldt et al. 2013), of which tree biomass is only one. Some of these amenities can be in conflict with the fertilization of forests (e.g., Hedwall et al. 2010) and need to be balanced against the biomass supply. This balancing act can, depending on the amenity, be performed on stand or landscape levels. For example, forest biodiversity can be maintained by forest zoning in which intensive management approaches are applied to a part of a forest landscape while other parts are managed partially or fully for conservation purposes (Ranius and Roberge 2011).

An important amenity of forests is clean fresh water from surface and ground waters. Fertilization with large amounts of N can have considerable negative impacts on the soil-water quality and commonly raises nitrate levels, especially immediately after the fertilization and after clear cutting previously fertilized stands (e.g., Ring 2004). These effects can be mobile in a forest landscape, and clean freshwater is thus a service that needs to, and can, be managed on the stand level. The leakage of N from fertilized forest soil has been shown to be more dependent on the size of single application than on the accumulated amount of N applied over several years (Bergh et al. 2008). There are economical and practical reasons for increasing the size of the application together with the interval, which implies a potential conflict with reducing the risk of nutrient leakage. A conflict with amenities other than biomass production implies limitations on the possibility of improving growth by fertilization and restricts nutrient addition in terms of spatial extent and intensity.

In addition to environmental restrictions, economic constraints limit the implementation of fertilization in forestry. The basic principles of economics for forestry imply that the earlier an investment is performed during a rotation period, the larger the revenue has to be to provide a positive return. Forests are exposed to a number of more or less stochastic threats. Wildfires, gales and pathogens commonly affect forests and forestry. The shorter the time a silvicultural investment forest is exposed to these risks, the better from an economic point of view.

6.5.2 Nordic Example

Fertilization is a common measure in Swedish and Finnish forestry and has been so since the mid-sixties. The spatial extent in Sweden reached its peak in the mid-seventies when approximately 200,000 ha ($\sim 1\,\%$ of the forested area) were fertilized annually. From a very low extent during the nineties and for the first few years of the new millennium, fertilization has again increased, and in 2010, 80,000 ha were treated (Lindkvist et al. 2011; Swedish Forest Agency 2011). For the economic reasons described above, fertilization is typically conducted in mature coniferous forests for operational forestry. Fertilization is normally performed using a nitrogen-rich fertilizer, commonly based on ammonium nitrate, with N supply of approximately 150 kg N ha^{-1} on one or more occasions. According to this scheme, fertilization is one of the most profitable measures in Swedish forestry, with internal rates of return that can exceed 15 %, depending on fertilizer prices and the market price on wood assortments (Jacobson and Pettersson 2010).

6.6 Conclusions and Future Possibilities

Nutrient optimization has the potential to substantially increase biomass production, especially when applied in young forests. There are fundamental differences between this type of management and traditional fertilization, as described above. Nutrient optimization implies frequent fertilization until the canopy closes, with a balanced supply of nutrients other than N. This type of fertilization regime provides a large reduction in rotation length compared with conventional silviculture. The rotation periods may be shortened by 10–30 years in the southern parts of northern Europe and by 30–60 years in the north (Bergh et al. 2005).

The first fertilization should be made when the trees are 2–4 m in height and then repeated frequently until the canopy closes. Experiments with balanced fertilization have demonstrated that the fertilization of Norway spruce can be performed every second year and still maintain the same level of production as with fertilization each year (Bergh et al. 2008), which is, of course, very important for the profitability of the operation. After canopy closure, fertilization is conducted every seven to every ten years in a manner similar to the traditional fertilization system of the Nordic countries (see the example above). In closed and mature stands, fertilization needs to be repeated one to three times after the first thinning. The last fertilization should be performed at least ten years before the final felling to exhaust the effects of the fertilization before the stand is cut. The total amount of N during a whole rotation will be 800–1500 kg N per ha (a lower amount in the southern parts of northern Europe and a higher amount in the north), from which approximately three-quarters will be supplied before canopy closure (Bergh and Linder 2006).

In a net present value calculation, fertilization later in the rotation period provides a superior economic return relative to fertilization starting from a young forest, where the fertilization is capital intensive and costs occur earlier during the rotation period. This cost is partially offset by the higher production rate and reduced rotation periods for balanced fertilization, where the interest rate is approximately 4–8 %, dependent on the initial site index, silvicultural program and climate conditions (Nilsson and Fahlvik 2006). However, highly productive sites and stands with less than 70 % Norway spruce should be avoided for economic reasons.

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Chapter 7
Integration of Management Measures for Bioenergy Production from Spatial and Temporal Perspectives in a Forest Region—the Case of Finland

Antti Kilpeläinen and Mitch Baker

Abstract The utilization and management of forest resources in the short term are dependent on the available resources in a region, which may not have been managed for bioenergy production. In the long term, the existing resources are developing under the control of the interaction between the initial structures, management interventions and changing climatic conditions, which affect the production potential of biomass and optimal forest management. Management changes the carbon stocks and carbon sequestration in the forest ecosystem. In general, undisturbed forests store the largest amount of carbon, while the utilization of biomass requires the removal of biomass from forests reducing the overall carbon stocks. On the other hand, forests act only as temporal sinks of carbon. Therefore, an analysis of the whole forest production system with various management objectives is needed to ensure sustainable production and optimal climate change mitigation in the long term.

Keywords Energy biomass · Carbon balance · Forest ecosystem · Management · Spatial and temporal scale · Bioenergy

7.1 Forest Management for Producing Energy Biomass

7.1.1 Factors Affecting the Utilization Potential of Forest Biomass in Energy Production

In this chapter, we discuss the integrated production of timber and energy biomass in forestry in varying spatial and temporal perspectives, with the focus on the production in the regional scale. In this scale, existing forests include a varying amount of single

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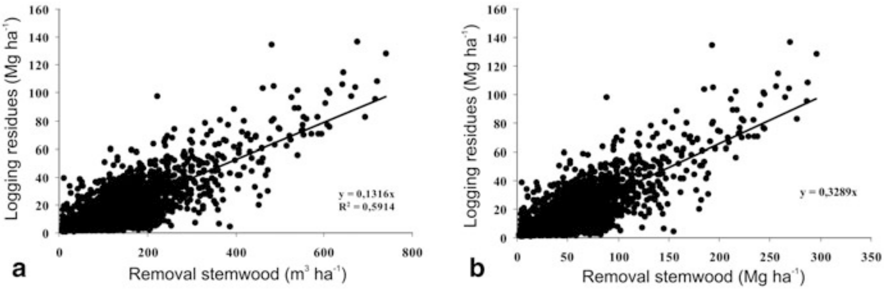


Fig. 7.1 Logging residues on clear cut areas as a function of timber harvest in $m^3\ ha^{-1}$ (*a*) and Mg ha^{-1} (*b*). Logging residues include potential for top parts of stems, branches and foliage, and stumps and coarse roots and they were harvested in Norway spruce, Scots pine and birch stands. The dots represent simulated average results for sample plots of national forest inventory (NFI) in Finland representing the current situation in forests

stands occupying sites with varying fertility and climatic conditions. In the long term, the existing forests are developing in the interaction between their current structure and management interventions, which affect the management and utilization potential of forests in producing energy biomass and timber. Furthermore, the changing climate may have effects on the growth and development of forests as is probably the case in Finland (e.g. Alam et al. 2010). All these factors affect the production potential of biomass and its optimal management in varying spatial and temporal scales. Production potential refers to the biologically possible production, excluding the technical, economic and societal limitations controlling the forest production if not otherwise indicated.

7.1.2 Potential of Energy Biomass in Forestry

The availability of energy biomass depends strongly on the age structure of the stands over a region. The share of final fellings and thinnings define the total cutting drain of the energy biomass (wood chips) and timber, and the quality of the harvested energy biomass (small-sized trees, logging residues, stumps and roots). Usually, forest bioenergy in the Nordic countries is harvested only in bioenergy thinnings as small-sized trees and in final fellings as logging residues. The gain of harvested biomass per area is higher in final fellings than in thinnings, which implies a strong correlation between timber and biomass harvestings (Fig. 7.1). This impact is further enhanced if stumps and roots are harvested for energy biomass.

Kärkkäinen et al. (2008) estimated that the current total recovery potential of logging residue in Finland is around 25Tg a^{-1} (57 million $m^3\ a^{-1}$) over the 50-year period from 2003 to 2052, but the proportion of different components changed during the simulation. For example, the decrease in the amount of final fellings was reflected as a decrease in the amount of waste wood from 10–13 % in the first

10-year period (2003–2013) to 4–6 % in the last period (2040–2053). Furthermore, Alam et al. (2010) found a greater relative effect of forest structure on the growth than that of climate change over a 90-year simulation period, the change being higher in southern than in northern Finland. This was a result of variability in the current forest structures in different parts of the country. Currently, the harvestable amount of energy biomass in Finland is 16 million m^3 a^{-1} (Helynen et al. 2007). Of this 45 % comes from harvesting of young forest (energy wood) and the first thinnings. The remaining 55 % represents harvesting of logging residue from final fellings; i.e. the top parts of the stem, branches, stumps and coarse roots.

The amount and properties of energy biomass available in the future will depend on the growth and development of the forests, their maturity (age class distribution) and tree species composition. In general, there will be more forests in the mature age classes in Finland, which is likely to increase the potential harvest of logging residues (Kärkkäinen et al. 2008). The amount of harvestable logging residues of Norway spruce is larger than that of Scots pine and birch, which are the other most common species grown and industrially utilized in the Nordic countries. Furthermore, the shallow root system of Norway spruce increases its harvest potential and its preference as a biomass producer. Therefore, the expected reduction of mature Norway spruce forests may reduce the harvest potential of energy biomass (Kärkkäinen et al. 2008), which is otherwise increasing due to the increasing growth in the future. Nevertheless, the realized harvest of timber and energy biomass will depend greatly on the capability to pay for timber for different end-use purposes and on the interests of forest owners in producing energy biomass.

Management has profound and prolonged effects on the forest structure and composition. In the Nordic countries, the primary objective of management has been to produce saw logs due to their value far above the value of pulpwood and wood chips. This is the reason why wide spacing is used in forestry, allowing trees to efficiently achieve the dimensions of saw logs. Regarding only the growth and yield of biomass, the current spacing may be too wide, but it is still open for discussion on how management should be optimized for the concurrent production of both timber and energy biomass (integrated forest management). Between the production of timber and energy biomass, there may be long-term trade-offs, which affect the carbon sequestration in ecosystems and the potential to mitigate climate change in forestry (e.g. Alam et al. 2010). Regarding energy production, optimal management and harvest chains reduce the CO_2 emissions per unit of energy compared with fossil fuels (e.g. coal) and increase the carbon density in the forest ecosystems for mitigating climate change. Regarding management on the regional scale, it is open for discussion whether to separate the production of energy biomass or to integrate it with the production of timber.

7.1.3 Enhancing Biomass Potential and Carbon Sequestration

Regarding the regional level, Routa et al. (2012) found that the late bioenergy thinning could produce more biomass out of the forest ecosystem. Because of the delayed thinning, the time interval between the last thinning and the final felling could be shortened and, therefore, the optimal growth potential remained unutilized during the later phases of the rotation. A similar trend was also found when both initial stand density and thinning thresholds were increased. This affected not only the production of energy biomass at final felling but also the timber yield and carbon storage (Alam et al. 2012). At the region scale, the management with higher initial stand densities sequestered and stored more carbon, while it slightly reduced the yield of energy biomass and timber compared with the management preferring lower initial densities (Baker 2012). The increase in carbon sequestration and storage was related to the large carbon gain in young stands of high density due to high carbon uptake and storing in the early rotation. The slight decrease in biomass and timber yields due to high initial densities was related to the over-crowding and higher competition for resources reducing the growth of single trees (see also Alam et al. 2012). This implied, for example, early dying of branches in lower crown and thus small amount of crown mass per tree.

7.2 Impacts of Varying Bioenergy Management on the Carbon Balance—the Case of Finland

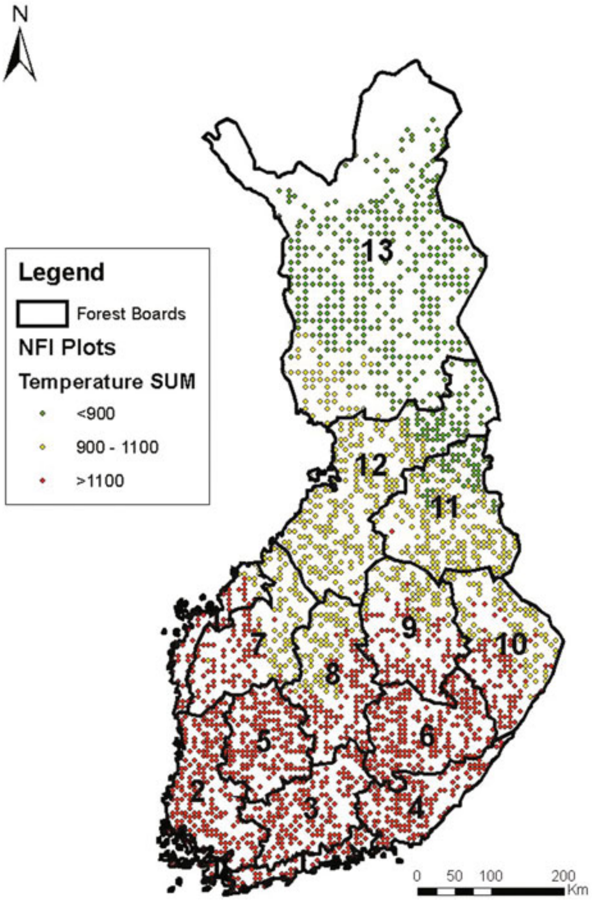
7.2.1 Approach

Regarding the mitigation against climate change in forestry, there is a need to integrate the substitution of fossil fuels with the uptake of carbon and control of the emissions in the forest ecosystem through management and harvesting of energy biomass. In the temporal and spatial scales, the existing biomass resources in the region define the bioenergy potential, but their management defines the future potential for carbon sequestration. In general, the carbon uptake peaks earlier than the maximum carbon stock is achieved, including both the carbon in trees and soil. Thus the consequent effects on the carbon balances and net CO_2 exchange will be highly important in determining how the production and use of forest biomass affect net climate impacts due to the substitution of fossil fuels in energy production (Kilpeläinen et al. 2012). This helps to identify optimized management systems to produce and utilize bioenergy for reducing the CO_2 emissions.

7.2.2 Simulations

Baker (2012) demonstrated how the regional differences in productivity, structure and management of forests affect the temporal and spatial distribution of carbon sequestration, and bioenergy and timber potentials over Finland. He split Finland into three regions (North, Central and South) based on differences in temperature sum; < 900 d.d. for the North, > 900 d.d. for Central and > 1100 d.d. for the South (Fig. 7.2). The sample plots of the Finnish National Forest Inventory (NFI) were

Fig. 7.2 Temperature sum region used in the simulations. The dots indicate the sample plots of the National Forest Inventory (NFI) and the lines are the boundaries of administrative regions (forest boards) used in forestry



utilized in the model simulations to study how several management and bioenergy production scenarios affected the carbon sequestration in the regional scale.

All the scenarios were defined based on the current regulations for forest management in Finland (Tapio 2006). The management included commercial thinning, final felling and stand establishment. Regardless of the management scenario, the harvest of timber and energy biomass took place in commercial thinnings and final fellings (Table 7.1). In addition to the control scenario (Business-as-usual; BAU), eight alternative management (AM) scenarios were used in identifying the effects of management. The increases in basal area thresholds (15 and 30 %) were used in defining alternative scenarios by assuming that the selected increase in tree stocking before and after thinning is a positive compromise between timber production and maintaining sufficient carbon stocks (Alam et al. 2008). The difference in the extent of biomass removed for bioenergy at the final felling (stems + branches + needles + roots *versus* only stems + branches) was used to determine the effect of reduced biomass removals on the carbon balance (Table 7.1).

Table 7.1 Description of the management scenarios

Scenario name	Initial seedling densities			Change in basal area threshold (%)	Biomass removals	
	Pine	Spruce	Birch		Thinning	Final felling
BAU	2000	2000	1800	0	St + Br	St + Br + Nd + Rt
AM15a	2000	2000	1800	15	St + Br	St + Br + Nd + Rt
AM15b	3000	2800	1800	15	St + Br	St + Br + Nd + Rt
AM15c	2000	2000	1800	15	St + Br	St + Br
AM15d	3000	2800	1800	15	St + Br	St + Br
AM30a	2000	2000	1800	30	St + Br	St + Br + Nd + Rt
AM30b	3000	2800	1800	30	St + Br	St + Br + Nd + Rt
AM30c	2000	2000	1800	30	St + Br	St + Br
AM30d	3000	2800	1800	30	St + Br	St + Br

St Stem, *Br* Branches, *Nd* Needles, *Rt* Roots

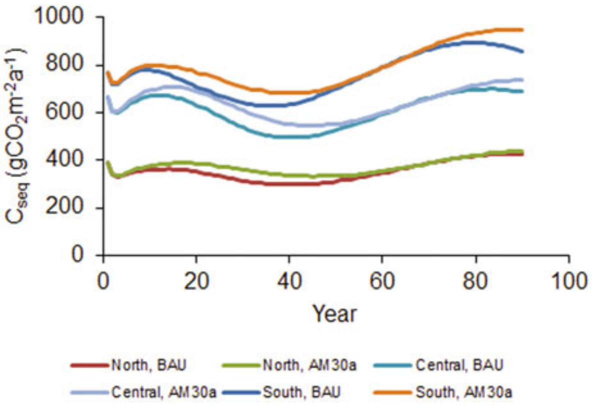


Fig. 7.3 Annual carbon sequestration (C_{seq}) in northern, central and southern Finland in a 90-year period (2010–2099) in each region for the selected management scenarios (BAU and AM30a). See Table 7.1 for the description of the management scenarios

7.2.3 Carbon Sequestration

Figure 7.3 shows that the annual carbon sequestration (above- and below-ground) was highest in the South and lowest in the North in the selected management scenarios. Carbon sequestration increased most in the South towards the end of the simulation period, whereas there was only a slight increase in the North and Central. Increasing the thinning threshold increased the carbon sequestration in the first half of the simulation period. The diversification of values between the management regimes occurred again during the last decades of the simulation period, especially in the South and Central.

7.2.4 Potentials for Energy Biomass and Timber

The scenarios producing the most energy biomass were the BAU, AM15 and AM30 (a, b) (Fig. 7.4). The scenarios AM15 and AM30 (c, d) produced less biomass than

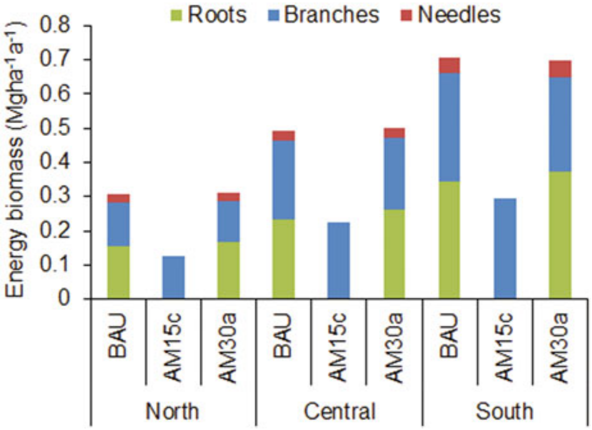


Fig. 7.4 Mean harvested energy biomass from each residue component for the selected management scenarios (BAU, AM15c and AM30a) over a 90-year period. See Table 7.1 for the description of the management scenarios

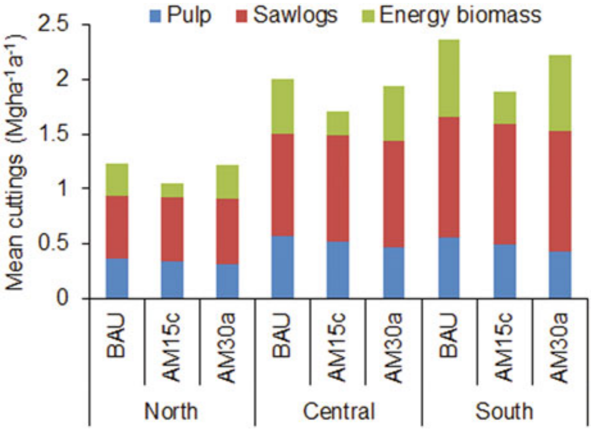


Fig. 7.5 Mean cuttings of energy biomass, sawlogs and pulpwood in each region for the selected management scenarios (BAU, AM15c and AM30a) over a 90-year period. See Table 7.1 for the description of the management scenarios

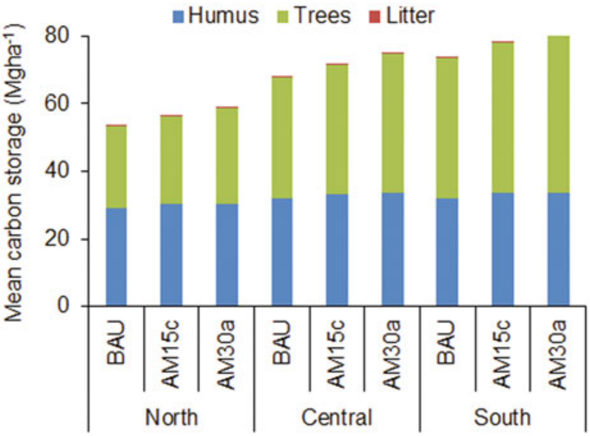
both the BAU and the other AM scenarios due to less biomass residues removed during the final felling (see Table 7.1). Regardless of the scenario, the amount of bioenergy produced in the South was significantly higher than that in the North (Fig. 7.4).

The AM scenarios produced less timber (both saw logs and pulp wood) than the BAU scenario (Fig. 7.5). The AM30 scenarios, which had the highest increase in the basal area threshold for thinning (30 %), produced the least amount of timber cuttings in all regions. Similar to the bioenergy production, the South produced significantly more cuttings than the North region in all scenarios (Fig. 7.5).

7.2.5 Above- and Below-ground Carbon Stocks

The AM scenarios had an effect on the amount of carbon stored in the forest in all the regions (Fig. 7.6). Compared to the BAU scenario, all of the AM scenarios stored more carbon in the forest with the highest storage in the AM scenarios with higher

Fig. 7.6 Mean carbon storage of litter, trees and humus in each region for the selected management scenarios (BAU, AM15c and AM30a) over a 90-year period (2010–2099). See Table 7.1 for the description of the management scenarios



initial stand densities. The AM scenarios storing the least carbon (while still higher than the BAU) were the scenarios with lower initial stand densities. The highest mean carbon storage was in the South and lowest in the North, which is expected with the differences in temperature sum.

7.2.6 Temporal and Regional Carbon Flows

Temporal and regional carbon flows were studied based on the life cycle assessment tool developed by Kilpeläinen et al. (2011, 2012, 2013). Carbon uptake, emissions and balance were used to calculate the CO_2 exchange (C_{net}) according to Eq. 7.1, which incorporates all the phases of bioenergy production within the system boundaries (cradle-to-gate) including both the ecosystem and the technosystem used to produce energy biomass and its use in producing energy and wood-based items. The accumulation or sequestration of carbon (both above and below ground) (C_{seq}) is expressed as a negative value, while the carbon emissions (i) through management (C_{man}), (ii) decomposition (C_{decomp}), (iii) biomass combustion and degradation of wood/pulp-based products (C_{harv}) are expressed as positive values.

$$C_{net} = C_{seq} + C_{decomp} + C_{harv} + C_{man} \tag{7.1}$$

The degradation (turn-over rate) of the wood/pulp-based products used in the LCA tool is based on Eq. 7.2 (Karjalainen et al. 1994):

$$PU = D - \frac{a}{1 + be^{-ct}} \tag{7.2}$$

where PU is the fraction (0–1) of the products in use, a, b and d are fixed parameters (120, 5, 120). T is time (a) and c (a^{-1}) is 0.5, 0.15, 0.065 and 0.03 specific for the life span of item short, medium-short, medium-long and long, respectively. The carbon

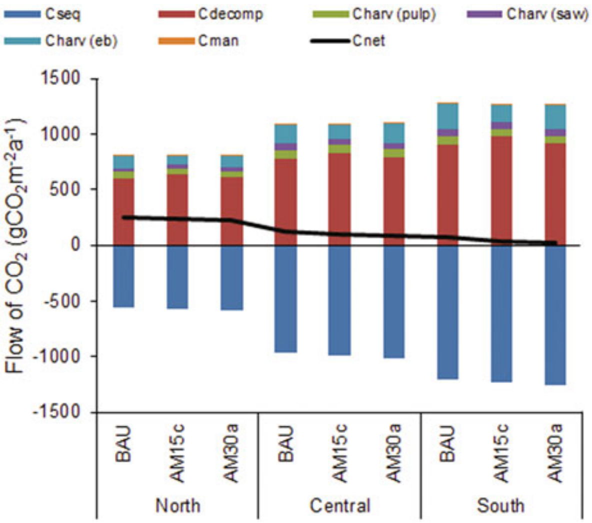


Fig. 7.7 Mean flows of CO_2 in each region for the selected management scenarios (BAU, AM15c and AM30a) over a 90-year period (2010–2099). See Table 7.1 for the description of the management scenarios

bound in energy biomass (C_{harv}) is added to the atmospheric CO_2 immediately after harvest, i.e. combusted in the power plant.

The AM scenarios had an impact on each of the components of the CO_2 exchange processes, and hence the overall carbon balances (Fig. 7.7). Under the scenarios with smaller biomass harvests and increased thinning thresholds (e.g. AM15c and AM30a), the values of net carbon exchanges (C_{net}) were the lowest. The scenarios with smaller biomass harvests represented lower CO_2 emissions from management (C_{man}) and energy biomass (C_{harv}) compared with BAU, while carbon sequestration (C_{seq}) was higher in the management scenarios with increased thinning thresholds. The carbon uptake and emissions were higher in the South than in other regions.

The structure and conditions of the forests in the three regions were reflected in the trends for cuttings, energy biomass and the carbon balance, with a greater effect in the southern region (Fig. 7.8). For example, the percentage of young forests was smaller in the North than in the South, where the uptake and emissions of carbon were more balanced than in the North. This implies that the timing of maximum values of the carbon balance is affected by the age structure of the forest in a given region. Furthermore, the increased thinning thresholds in management reduced the regional values of net CO_2 exchange compared with no changes in the threshold values. Interestingly, this switched in each region after the maximum values of C_{net} were reached.

7.2.7 Trade-off Between Bioenergy Production and the Carbon Balance

The AM scenarios with smaller biomass harvests (e.g. AM15c) produced the least amount of energy biomass and the lowest figures for the net CO_2 exchange (C_{net}) in

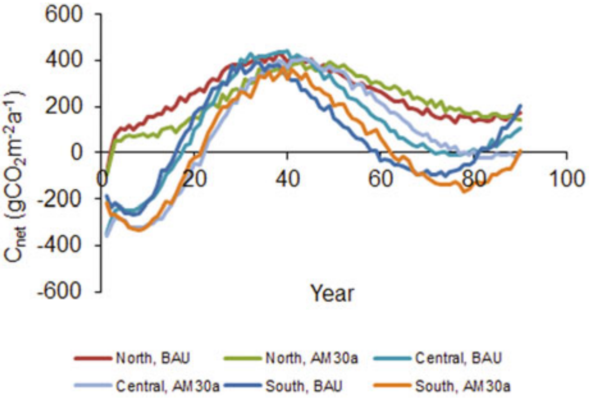


Fig. 7.8 Annual net exchange of CO_2 (C_{net}) in northern, central and southern Finland over a 90-year period (2010–2099) in each region for the selected management scenarios (BAU and AM30a). See Table 7.1 for the description of the management scenarios

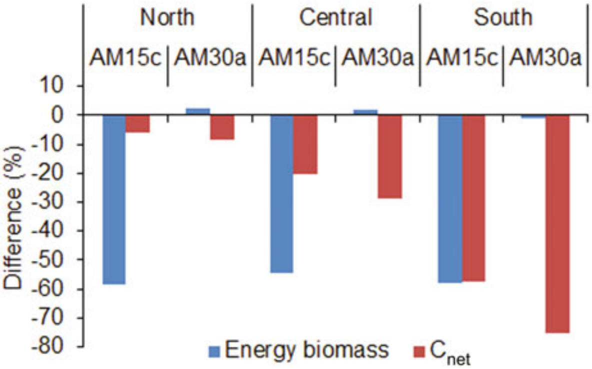


Fig. 7.9 The difference (%) from the BAU scenario in the mean amount of energy biomass (MWh ha^{-1}) and in the net exchange of CO_2 (C_{net}, gCO$_2$ m^{-2}a^{-1}) for the selected management scenarios (AM15c and AM30a) over a 90-year simulation period (2010–2099). See Table 7.1 for the description of the management scenarios

comparison with the BAU scenario (Fig. 7.9). In these scenarios, less biomass was removed in the harvests, which negatively affected the energy value and positively the CO_2 exchange, mainly due to the energy wood turnover (C_{harv}). The scenarios with less removal of energy biomass had a negative impact on the production of energy biomass, while having a positive impact on the carbon balance (C_{net}). This positive impact on the carbon balance is due to the fact that the biomass is not being directly converted into CO_2 emissions through the burning process and is remaining on the forest floor. In general, the trade-off between bioenergy production and the carbon balance was the smallest in the AM a and b scenarios, where the threshold values for thinning were increased by 15 and 30 % (Baker 2012). For example, the AM30a scenario provided more energy from biomass in the North and Central and a lower carbon balance than the BAU scenario in each region (Fig. 7.9).

7.3 Summary and Conclusions

This chapter highlighted the impacts of forest management on bioenergy and timber production, and the consequent changes in carbon stocks and carbon balance (C_{net}). In forest production, trade-offs occur between carbon sequestration and carbon storage due to the removal of biomass from forests. On the other hand, forests act only as temporal sinks of carbon controlled by the long-term development of the forest ecosystems and the effects of management and harvest on spatial biomass/carbon reservoirs. In general, forest composition skewed to the younger age classes is preferable for biomass production. In combining the production of biomass and timber and storing of carbon, more balanced age class distribution is needed in order to sustain the high carbon uptake capacity of young trees and high storage capacity of mature trees over time (Garcia-Gonzalo et al. 2007; Routa et al. 2012). Regarding the current age structure of forests in Finland, there seems to be no lack of energy biomass in the near future. However, the shift of the age structure to more mature stands may decrease the potential of energy biomass in the long run, especially in southern Finland. On the other hand, this is also affecting the net carbon sequestration potential of forests in general and reflects the prior management of the forests. In this respect, management allowing higher stocking than is optimal for timber production seems to satisfy both the needs to produce timber and energy biomass and to store carbon in the forest production.

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Part II
Mitigation and Carbon Sequestration

Chapter 8
Effects of Bioenergy Production on Carbon Sequestration in Forest Ecosystems

Seppo Kellomäki, Antti Kilpeläinen and Ashraful Alam

Abstract This chapter explores the dynamics of carbon sequestration in forest ecosystems. The focus is on the physiological and ecological mechanisms that control the carbon uptake and emissions in boreal forests. Findings from long-term monitoring and process-based modeling are used to demonstrate the sink/source dynamics of carbon sequestration in naturally growing and developing forest ecosystems and how this deviates from that under regular management. Carbon retention in the ecosystem is addressed based on model simulations, which represent both natural and managed forests. The effects of biomass production on carbon sequestration are analyzed based on model simulations in order to identify how management may affect carbon residence in the boreal forest ecosystem.

Keywords Carbon cycle · Carbon sequestration · Boreal forest ecosystem · Carbon residence · Eddy covariance measurement · Energy biomass · Ecosystem model · Litter · Humus · Emissions · Carbon sink · Carbon source · Decomposition · Physiological process · Ecological process · Net carbon exchange

8.1 Global Carbon Cycle

Managed forests provide biomass for energy production, but their potential to provide energy biomass is very much dependent on management. The harvest of forest biomass for energy alters the carbon sink/source dynamics of forest ecosystem. This links the carbon sequestration and the consequent carbon balance in the ecosystem

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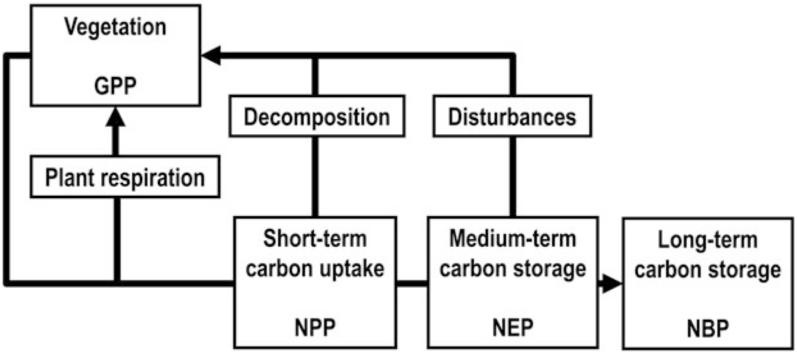

Fig. 8.1 Outlines of global carbon cycle as adapted from Grace (2005, p. 24). *Gross primary production* (*GPP*): carbon fixed in photosynthesis; *Net primary production* (*NPP*): gross primary production minus carbon lost in plant respiration (*RA*) (autotrophic respiration), $NPP = GPP - RA$; *Decomposition* (*RH*): carbon lost in heterotrophic respiration; *Net ecosystem production* (*NEP*): net primary production minus heterotrophic respiration (*RH*): $NEP = NPP - RH = GPP - (RA + RH)$; *Net biome production* (*NBP*): net gain or loss of carbon in a given area at the biome level equal to the net ecosystem production minus carbon loss in disturbances such as fire or logging; *Biome*: major region of distinctive plant and animal groups well adapted to the physical environment in its distribution area, e.g. boreal forests

with the global carbon (C) cycle. As outlined in Fig. 8.1, carbon is fixed in photosynthesis of plants (gross primary production) and lost in respiration of plants (autotrophic respiration) and in decay of litter and humus (heterotrophic respiration). Furthermore, disturbances like fire and harvesting of biomass cycle carbon back to the atmosphere. Between uptake and loss, carbon is retained for shorter and longer periods in the ecosystem (carbon storage), but finally carbon will end up in the atmosphere. Short-term carbon storage refers to net primary production (gross primary production minus autotrophic respiration), whereas medium-term carbon storage indicates the carbon bound in plant biomass and other organisms (gross primary production minus autotrophic and heterotrophic respiration), and organic matter in soil. Long-term carbon storage describes carbon storage over areas of varying spatial scale (stand, landscape, national, and continental), including effects due to changes in land use and practices in land management.

8.2 Bioenergy to Mitigate Climate Change

Carbon in the established forests is bound in trees and other plants and in soil in the form of decaying litter and humus. The residence time of carbon in forests depends on the management and the prevailing environmental conditions which control the decay of litter and humus. Thus, the mitigation of climate change through sequestration of carbon in forest ecosystems depends on their management.

Mitigation of climate change refers to the anthropogenic intervention to reduce the sources or enhance the sinks of greenhouse gases in order to reduce the intensity of radiative forcing, thus reducing the potential global warming due to the increasing concentration of greenhouse gases in the atmosphere (IPCC 2001, p. 990). In forestry, there are several ways to mitigate climate change by sequestrating carbon dioxide (CO_2) in forest ecosystems and reducing the emissions of CO_2 to the atmosphere (IPCC 2007):

(1) to reduce deforestation and degradation of forests;
(2) to increase forested land area through afforestation/reforestation;
(3) to increase the carbon density in existing forests;
(4) to increase the use of biomass to replace fossil fuels in energy production; and
(5) to increase the use of biomass-based products to increase carbon density outside existing forests.

Activities (1) and (2) aim to maintain and increase the sink capacity in terms of conserve forested land, whereas activity (3) refers mainly to increasing the carbon uptake rate through proper management. Activity (4) refers to the reduction of emissions of fossil carbon by substituting fossil fuels with biomass, thus cycling carbon in the atmosphere/biosphere system in order to slow the increase of atmospheric CO_2 or even to reduce it. Activity (5) indicates the carbon sequestrated in biomass-based products, thus cycling carbon through the biosphere/technosystem/atmosphere systems. The residence time of carbon in forest products varies from a few years in paper to several decades in solid wood products, like building materials. In biomass harvested for energy, the residence time is less than a year or even a matter of weeks.

8.3 Carbon Flow Through Forest Ecosystem in Physiological and Ecological Processes

Figure 8.2 shows the carbon flow through a forest ecosystem in physiological and ecological processes based on Waring and Running (2007). Gross primary production (*GPP*) is mainly controlled by the interception of radiation in tree canopies and the cycle of nutrients, in boreal conditions especially by the availability of nitrogen (Hyvönen et al. 2007). Uptake of CO_2 is further limited by temperature, humidity, concentration of atmospheric CO_2 and availability of soil water affecting stomatal conductance. Following Waring and Running (2007), the difference between the gross primary production and daytime respiration (R_m) in canopy yields the net canopy assimilation (A), which is partly lost in nighttime respiration. Gross primary production minus the carbon losses in canopy respiration gives the diurnal net canopy exchange (*NCE*). In this context, net primary production (*NPP*) refers to the gross primary production minus carbon lost in autotrophic respiration. Half of the gross amount of carbon assimilated in photosynthesis (*GPP*) is cycled back to the atmosphere in autotrophic respiration (*RA*) originating from the synthesis and maintenance of living cells in various organs of trees (Hyvönen et al. 2007).

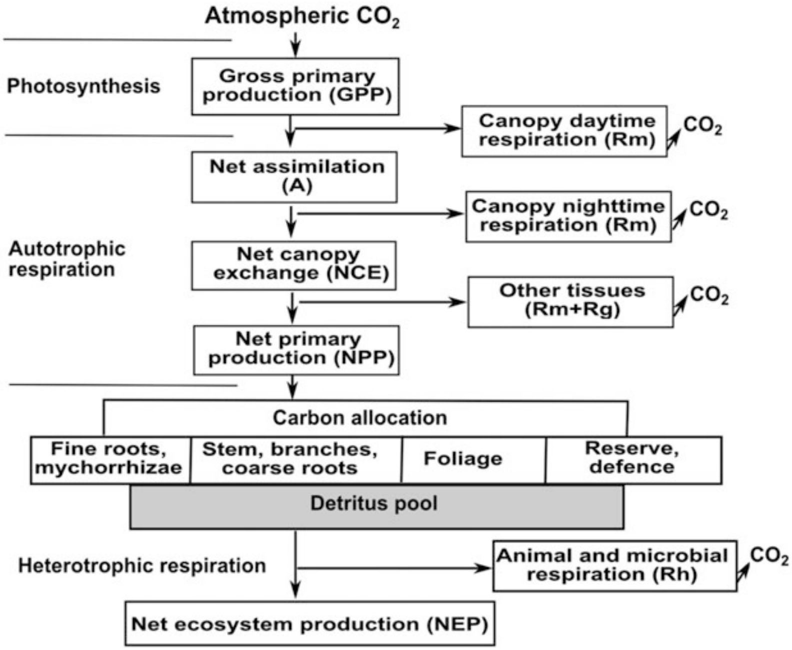

Fig. 8.2 Schematic representations of the flow of carbon through a forest ecosystem as adapted from Waring and Running (2007)

Carbon in net primary production is allocated into various organs; i.e. fine roots and mychorrhizae, stems, branches and coarse roots, foliage, and chemicals for defence (Fig. 8.2). Turnover of foliage and fine roots are the major contributors to litter on a seasonal basis, but the biomass in all organs finally enters the detritus pool following the turnover of mass in different organs. Decomposition of litter and release of CO_2 due to the activity of heterotrophic organisms are functions of substrate quality (e.g. the ratio between carbon (C) and nitrogen (N), C/N ratio) and the temperature and moisture conditions in soil profile (e.g. Chertov et al. 2001). In this context, the net ecosystem production (*NEP*) or net ecosystem exchange (*NEE*) indicates the difference between the carbon taken up in gross primary production and lost in autotrophic and heterotrophic respiration. Taking account of the loss of carbon in disturbances (e.g. forest fire, harvest of trees), one obtains the net biome production (*NBP*) linking the physiological uptake and loss of carbon with the losses in ecological processes. Their time steps are typically longer (> years and decades) than those for physiological processes (< few days). Table 8.1 summarizes the flow of carbon through the forest ecosystem in physiological terms.

Table 8.1 Concepts used in analyzing the carbon flow through forest ecosystems

Concepts	Definition
Gross primary production, *GPP* (g m^{-2} a^{-1})	Gross carbon fixation rate into the forest ecosystems through trees and other green organisms
Autotrophic respiration, *RA* (g m^{-2} a^{-1})	Loss rate of carbon through autotrophic respiration rate, $RA = R_{day} + R_m + R_g$
Day, R_{day}	
Dark, R_m	
Growth, R_g	
Net primary production, *NPP* (g m^{-2} a^{-1})	Gross production rate minus autotrophic respiration rate, $NPP = GPP - RA$
Heterotrophic respiration, *RH* (g m^{-2} a^{-1})	Loss rate of carbon through respiration of secondary producers
Total respiration, *RT* (g m^{-2} a^{-1})	Sum of autotrophic and heterotrophic respiration rates, $RT = RA + RH$ over a year
Net ecosystem production (*NEP*) or Net ecosystem exchange, *NEE* (g m^{-2} a^{-1})	Gross primary production minus total respiration or net primary production minus heterotrophic respiration, $NEE = GPP - RA - RH = NPP - RH$
Net biome production, *NBP* (g m^{-2} a^{-1})	Net biome production $NBP = NEE$ plus CO_2 lost in disturbances, e.g. harvest, fires etc.

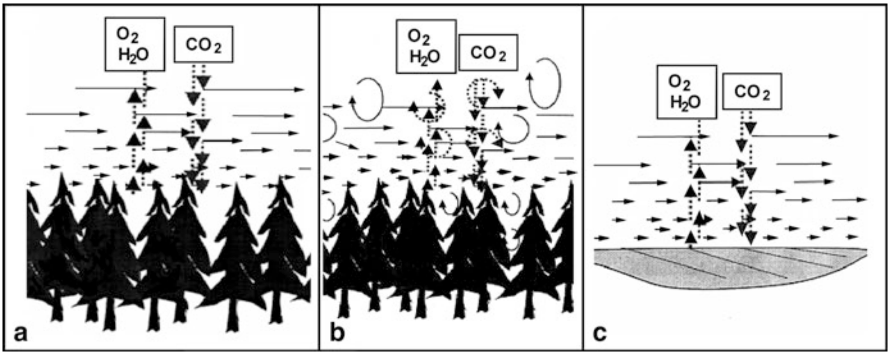

Fig. 8.3 Schematic representation of the exchange of CO_2 and H_2O in a forest ecosystem if air flow through canopy is laminar (**a**), and turbulent (**b**), and the exchange of CO_2 and H_2O in the boundary layer of leaves (**c**)

8.4 Carbon Sequestration in Boreal Forest Ecosystem

8.4.1 Net CO_2 Exchange in Forest Ecosystem (NEE) – Basic Mechanisms

The concept of eddy covariance (EC) with the proper technology is widely used to measure the net CO_2 exchange of ecosystem (*NEE*) in a forest ecosystem (e.g. Baldocchi 2003); i.e. the difference between carbon uptake through photosynthesis and carbon loss through autotrophic and heterotrophic respiration. Figure 8.3

shows air flow through the forest canopy of a high enough velocity to be classed as turbulent (middle of Figure) with eddies. The eddies carry CO_2 and H_2O back and forth between atmosphere and canopy, whereas a slower air flow is classed as laminar (left) with no/few eddies and with only a small exchange of CO_2 and H_2O between canopy and atmosphere. In both cases, the exchange of CO_2 and H_2O occurs in the boundary layer of leaves/canopy through uptake of CO_2 in photosynthesis and emission of H_2O in transpiration. In determining EC, the CO_2 content in eddies and the three-dimensional wind velocity are measured and used to calculate the exchange rate between the atmosphere and the ecosystem.

In EC measurements, photosynthesis is regarded as a negative flux into the ecosystem from the atmosphere, while respiration is a positive flux out from the ecosystem to the atmosphere. During the daytime, NEE_{day}, [g C m^{-2} ground day^{-1}] equals *GPP* in the canopy minus the carbon lost in autotrophic respiration in leaves (R_l, g C m^{-2} ground day^{-1}), branches, sapwood and roots ($R_{b,w}$, g C m^{-2} ground day^{-1}) and in heterotrophic respiration in soil (R_s, g C m^{-2} ground day^{-1}). Furthermore, autotrophic respiration in leaves may be divided into photorespiration ($R_{l.p}$) and respiration in darkness ($R_{l.d}$) (Ge et al. 2011):

$$\mathrm{NEE_{day}} = \mathrm{GPP} - (R_l + R_{b,w} + R_s) = \mathrm{GPP} - (R_{l.p} + R_{l.d} + R_{b,w} + R_s) \tag{8.1}$$

Net ecosystem exchange at night ($\mathrm{NEE_{night}}$, g C m^{-2} ground day^{-1}) includes only autotrophic losses in darkness and in heterotrophic respiration:

$$\mathrm{NEE_{night}} = R_{l.d} + R_{b,\,w} + R_s \tag{8.2}$$

On the daily basis, the net CO_2 exchange between the forest ecosystem and the atmosphere is the sum of eddy fluxes (F_e, g C m^{-2} ground day^{-1}) from the canopy, monitored by the EC instrument, and the flux representing the storage of CO_2 (F_s, g C m^{-2} ground day^{-1}) in the air layer below the canopy (Aubinet et al. 2000).

$$\mathrm{NEE} = F_e + F_s \tag{8.3}$$

The flux from the canopy (F_e) refers to the mean covariance between the vertical wind velocity (w') and the fluctuations in the CO_2 density (c') including the correction of measured CO_2 flux due to changes in air density (Aubinet 2000; Ge et al. 2011):

$$F_e = \left(\frac{\overline{p}\,\overline{T_i}}{\overline{p_i}\,\overline{T_a}}\right)\left[\overline{w'c'_i} + \frac{m_a}{m_v}(\overline{c'}/\overline{\rho_a})\overline{w'\rho'_{vi}}\right] \tag{8.4}$$

where p is pressure (Pa), T is temperature (°K), ρ_{vi} and ρ_a are the mean densities of water vapor and dry air respectively. Furthermore, m_a/m_v is the ratio between the molecular mass of dry air and that of water vapor. The bars over factors refer to the time averages and primes to the instantaneous fluctuations around the mean quantities (Aubinet et al. 2000). The carbon storage below the canopy (F_s) is obtained by integrating the rate of temporal change in the CO_2 concentration profile below the canopy:

$$F_s = \frac{v_a \Delta c_r}{m} \tag{8.5}$$

where v_a is the volume (m^3) of the air column below the height of the EC instrumentation, Δc_r is the change in CO_2 density per unit time at the canopy source height, and m is the molar volume of CO_2 (Aubinet et al. 2000).

Box 8.1 How to scale up from photosynthetic biochemistry per unit leaf area to canopy photosynthesis per unit ground area (Ge et al. 2011)

To scale up from photosynthetic per unit leaf area to canopy photosynthesis per unit ground area (A_c) is based on micro-meteorological models. For example, they may use an integrated sun/shade sub-model to consider the daily change in the fraction of sunlit (LAI_{sun}) and shaded leaf index (LAI_{sh}) in the canopy and the corresponding difference in photosynthetic rate and canopy stomatal conductance (g_{cs}). Net radiation absorbed in the canopy (R_{nc}) is divided into sunlit ($R_{nc.sun}$) and shaded ($R_{nc.sh}$) fractions of foliage; i.e. net canopy photosynthesis (A_c) includes photosynthesis both in sunlit ($A_{c.sun}$) and shaded ($A_{c.sh}$) foliage (Kellomäki and Wang 1999, 2000):

$$\begin{aligned} \mathrm{A_c} &= \mathrm{A_{c.sun}} + \mathrm{A_{c.sh}} \\ &= \int_0^{LAI} f(R_{\mathrm{nc.sun}}) f(T_{\mathrm{c}}) f(c_{\mathrm{a}}) f(N_{\mathrm{L}}) f(LAI_{\mathrm{sun}}) f(g_{\mathrm{s.sun}}) dLAI \\ &\quad + \int_0^{LAI} f(R_{\mathrm{nc.sh}}) f(T_{\mathrm{c}}) f(c_{\mathrm{a}}) f(N_{\mathrm{L}}) f(LAI_{\mathrm{sh}}) f(g_{\mathrm{s.sh}}) dLAI \end{aligned} \tag{8.6}$$

$$\mathrm{R_{nc}} = \mathrm{R_{nc.sun}} + \mathrm{R_{nc.sh}} \tag{8.7}$$

$$g_{\mathrm{cs}} = \frac{LAI_{\mathrm{sun}}}{LAI} \int_0^{LAI} g_{\mathrm{s.sun}} + \frac{LAI_{\mathrm{sh}}}{LAI} \int_0^{LAI} g_{\mathrm{s.sh}} \tag{8.8}$$

where the functions $f(T_c)$, $f(C_a)$, $f(NL)$, $f(g_{\mathrm{s.sun}})$ and $f(g_{\mathrm{s.sh}})$ refer to the effects of air temperature, atmospheric CO_2, foliage nitrogen content and stomatal conductance on the CO_2 uptake rate. Stomatal conductance (g_s) is controlled by radiation (R_{nc}), vapor pressure deficit (D_a), concentration of CO_2 (C_a), canopy temperature (T_c), soil moisture (W_{soil}) and soil temperature (T_s):

$$g_{\mathrm{s}} = g_{\mathrm{s.max}}\, f(R_{\mathrm{nc}}) f(D_{\mathrm{a}}) f(c_{\mathrm{a}}) f(T_{\mathrm{c}}) f(W_{\mathrm{soil}}) f(T_{\mathrm{s}}) \tag{8.9}$$

where $g_{\mathrm{s.max}}$ is the maximum values of stomatal conductance. Respiration in foliage in darkness ($R_{l.d}$) is controlled by canopy temperature and the nitrogen content in foliage. Heterotrophic respiration in the forest floor is the function of soil temperature (T_s) and water content (θ_s) of soil:

$$R_{\mathrm{l.d}} = R_{\mathrm{l.d(20)}} f_1(T_{\mathrm{c}}) \int_0^{LAI} f(N_{\mathrm{L}}) f(LAI) dLAI \tag{8.10}$$

$$R_{\mathrm{b.w}} = (M_{\mathrm{b}} R_{\mathrm{b(20)}} + M_{\mathrm{w}} R_{\mathrm{w(20)}}) f_2(T_{\mathrm{w}}) \tag{8.11}$$

$$R_s = R_{s(20)}(1 - e^{\theta_s c_\theta}) f_2(T_s) \tag{8.12}$$

where $R_{l.d(20)}$, $R_{b(20)}$, $\mathrm{R}_{w(20)}$ and $R_{s(20)}$ are the rates of autotrophic respiration in foliage, branches, sapwood and the rate of heterotrophic respiration in the forest floor at a temperature of 20 °C. Furthermore, T_w refers to the temperature in sapwood. The functions $f(N_L)$, $f_1(T_c)$ and $f(LAI)$ represent the nitrogen content in foliage, temperature in canopy, and the fraction of foliage area. The factors M_b and M_w (kg m^{-2} ground) are dry biomass of living branches and sapwood.

8.4.2 Carbon Sequestration in Forest Ecosystems – a Case Based on Eddy Covariance Measurements

Ge et al. (2011) reported a time series of EC measurements over the period 1999–2008 in a boreal Scots pine forest located in eastern Finland (62°52′N, 30°49′E, 145 m a.s.l) on soil of low fertility (*Vaccinium* site type). The mean density of the forest was 1175 trees ha^{-1}, with a mean height of 12 m (the mean of the dominant trees (upper 10 %) was 17.6 m) and a mean diameter of 11 cm. The soil at the site is a sandy loam with a water-holding capacity of 35 mm in the top 50 cm of soil. Climate at the site is characterized by cold winters with thick snow cover and short growing season (140–175 days). Mean annual precipitation in the area for the period 1961–2000 was around 700 mm, of which about 38 % was snow. The mean temperature is – 10.4 °C in January and 15.8 °C in July (Fig. 8.4) .

Figure 8.5 shows the daily data points of EC measurements and the model values, which are calculated on the basis of meteorological measurements using the process-based model outlined in Box 8.1 (Ge et al. 2011). For most of the winter time, the daily values of NEE were slightly positive (carbon source) until the end of April, representing the time of release of winter dormancy and enhancing photosynthesis; i.e. the site became a carbon sink (negative values of EC measurements). In May, the carbon uptake increased rapidly, and from June to August the site was a carbon sink, where the carbon uptake substantially exceeded the carbon emissions. Thereafter, the carbon uptake declined through September and October until trees fell into winter dormancy and the site became a slight carbon source once again. The model simulated in an adequate way the annual course of daily values of carbon flux as measured by the EC method (Ge et al. 2011).

During the growing season (May–September), the amount of modeled NEE values varied from −189 to −263 g C m^{-2}, and from −195 to −268 g C m^{-2} for EC measurements (Table 8.2). Outside the growing season, the simulated values varied from 34 to 69 g C m^{-2}, whereas the measured values fell in the range from 27 to 67 g C m^{-2}. On an annual basis, for the period 1999–2008, the modeled values of NEE ranged

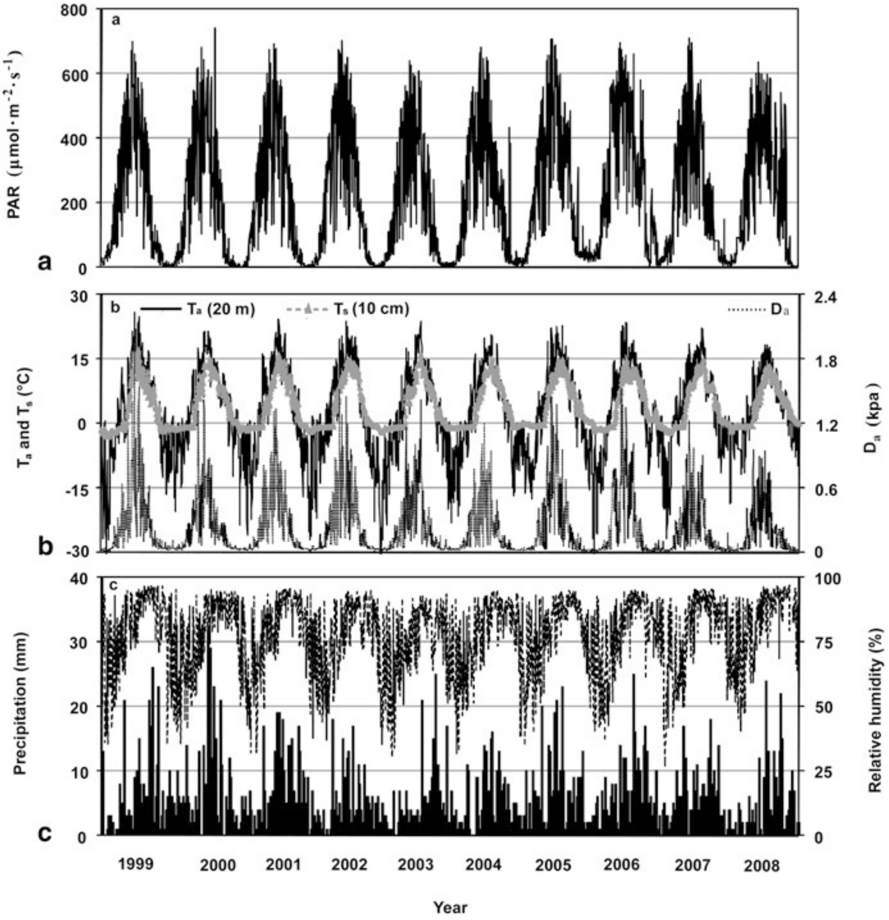

Fig. 8.4 Meteorological and soil conditions 1999–2008 in the Huhus monitoring site (Ge et al. 2011). (**a**) the integrated daily photosynthetically active radiation (*PAR*) above the canopy, (**b**) the mean daily canopy temperature (T_c) at a height of 20 m and soil temperature (T_s) at a depth of 10 cm, (**c**) the mean vapor pressure deficit (D_a) at a height 8 m and the daily precipitation and the mean relative humidity above the canopy. (Ge et al. 2011)

from -130 to $-203\,\mathrm{g\,C\,m^{-2}}$ and the measured values from -161 to $-232\,\mathrm{g\,C\,m^{-2}}$. During the growing season, the daily values of NEE were dependent on air temperature (T_c), photosynthetically active radiation (*PAR*) and vapor pressure deficit (D_a). The trend of NEE variation roughly met the response of net photosynthesis of Scots pine to corresponding environmental variables under laboratory conditions (Kellomäki and Wang 1997a, b, 1998; Wang 1996; Wang et al. 1996).

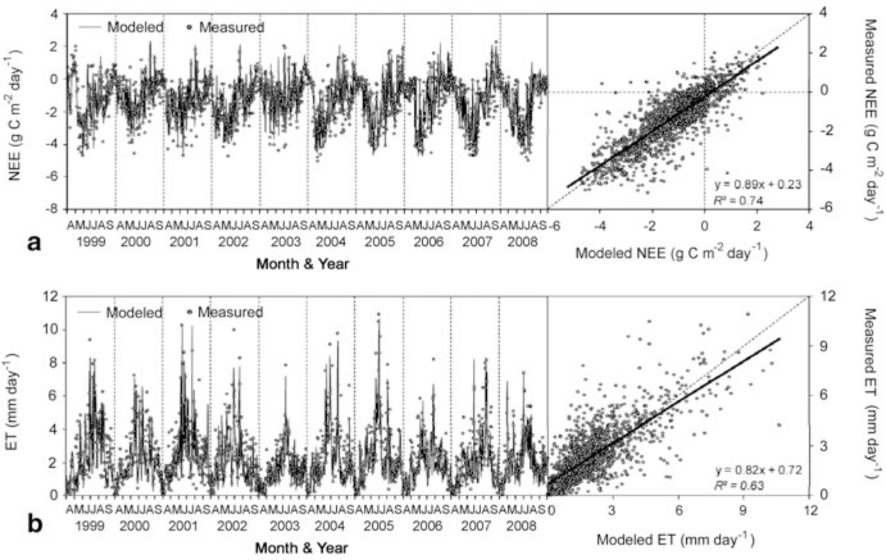

Fig. 8.5 Annual variation of daily ecosystem net carbon flux (*NEE*) in the boreal Scots pine stand in the Huhus monitoring site in 1999–2008. Points are measured fluxes and solid lines are modeled fluxes. (Ge et al. 2011)

Table 8.2 Seasonal (growing season: May–September, outside growing season: October–April) and annual amount of net ecosystem CO_2 exchange (NEE, g C m^{-2}) based on model computations and EC measurement. (Ge et al. 2011)

Year	Growing season		Outside growing season		Whole year	
	Modeled	Measured	Modeled	Measured	Modeled	Measured
1999	– 205.7	– 213.0	39.8	26.7	– 165.8	– 186.3
2000	– 189.0	– 195.0	59.2	34.0	– 129.8	– 161.0
2001	– 239.8	– 237.0	59.3	55.0	– 180.5	– 182.0
2002	– 246.7	– 250.4	43.9	36.7	– 202.8	– 213.7
2003	– 255.3	– 267.7	60.6	36.2	– 194.7	– 231.5
2004	– 189.8	– 206.8	34.3	48.6	– 155.4	– 158.1
2005	– 255.4	– 254.4	64.5	40.0	– 190.9	– 214.4
2006	– 252.4	– 240.9	65.9	50.2	– 186.5	– 190.7
2007	– 223.9	– 219.8	61.1	49.3	– 162.8	– 170.4
2008	– 263.1	– 257.0	68.8	67.2	– 194.3	– 189.8

8.5 Carbon Flow Through Forest Ecosystems in Ecological Processes

8.5.1 Dynamics of Carbon in Populations of Trees

In forestry, trees provide the basic unit for analyzing the sink/source dynamics of carbon. In physiological terms, trees represent the NPP production (i.e. gross production

Table 8.3 Selected concepts used in growth and yield studies in assessing the dynamic of populations and communities of trees

Concept	Explanation
Gross growth, m^3 ha^{-1} a^{-1}	Total annual growth of trees
Total gross growth, m^3 ha^{-1}	Total accumulated growth over time
Mortality, m^3 ha^{-1} a^{-1}	Annual rate of tree death
Total mortality m^3 ha^{-1}	Accumulated rate of death over time
Net growth, m^3 ha^{-1} a^{-1}	Rate of difference between the gross growth and mortality
Total net growth, m^3 ha^{-1}	Accumulated net rate of growth over time
Stocking, m^3 ha^{-1}	Amount of stem wood at a given point of time

rate minus autotrophic respiration rate, NPP = GPP − RA). In the forestry context, the dynamics of carbon in trees is related to the dynamics of population/community of trees defined by the birth (regeneration), growth and mortality of trees:

$$\begin{aligned} M(t) = {} & n(t-1)\cdot m(t) + n(t-1)\cdot g(t) \\ & + h(t)\cdot s(t) - k(t)\cdot (m(t)+g(t)) \end{aligned} \tag{8.13}$$

where $M(t)$ is the mass (volume) of trees at the moment t, $n(t)$ is the number of trees, $m(t)$ is the mean mass (volume) of trees, $g(t)$ is the mean growth (volume, mass) of trees, $h(t)$ the number of new trees (regeneration), $s(t)$ the mean mass (volume) of new trees, and $k(t)$ the dying trees at the moment t. Thus, the stocking at a given moment is the balance based on the mass in the previous moment plus the growth of trees and the mass of new (regenerated) trees minus the mass of dying trees and their growth at the current moment, as summarized in Table 8.3. In this case, the amount of trees is given in terms of m^3 ha^{-1} and the rate of change in terms of m^3 ha^{-1} a^{-1}, as widely used in forest inventories and forestry.

Based on Lönnroth (1929); Kilkki (1984) demonstrated the dynamics of a tree population by applying the graphic model in Fig. 8.6. The model includes further the litter and humus on the soil surface and humus deeper in the soil profile. In the model, growth is given in terms of net growth (*Zn*) and gross growth (*Zbr*), the latter indicating the increase of living mass in the tree population/community in a given period (in growth and yield studies, the living material also includes sound heartwood, even though it has no role as regards the metabolism of the tree). The gross growth includes the living mass and the mass dying during a given period, e.g. foliage, branches, bark and roots dying annually, and whole trees dying and falling down or continuing to stand for a while. The dying mass forms the removal, which may represent the natural removal and/or removal in cuttings. In both cases, the growth of dying trees during the given period contributes to the gross growth of population/communities of trees. The model includes also the ingrowth (K) of trees occurring during a given period. The ingrowth refers to the mass of trees that exceed the threshold (e.g. height, diameter of seedlings born previously and/or during a given period) used to include/exclude trees in inventory.

In Fig. 8.6, the left-hand boxes represent the naturally (a) growing and developing population/community of trees and the right-hand boxes the same, with thinning (b).

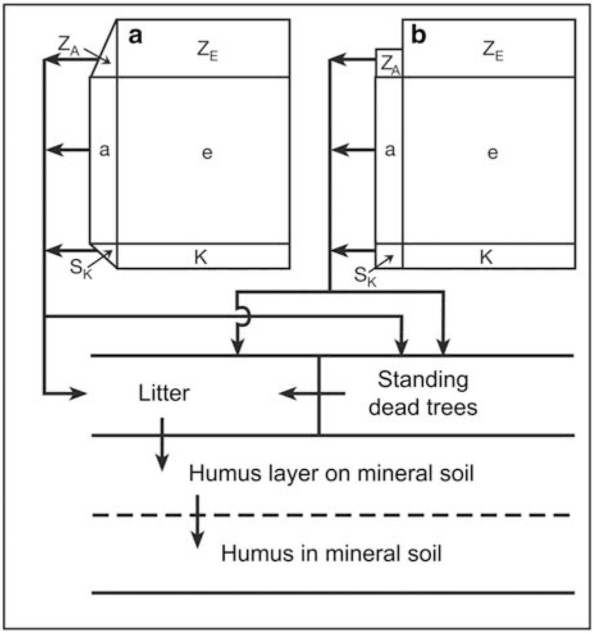

Fig. 8.6 Relations between growth, removal and stocking in natural developing (**a**) and managed (**b**) forests over a given period as presented by Kilkki (1984). $e =$ Initial stocking surviving to the end of the period; $a =$ Initial stocking removed naturally or in cuttings during the period; $K =$ Growth surviving to the end of the period for the trees exceeding the measuring accuracy at the beginning of the period; $S_K =$ Share of growth removed during the period for the trees exceeding the measuring accuracy at the beginning of the period; $Z_E =$ Share of growth for the initial stocking surviving to the end of the period; $Z_A =$ Growth of initial trees removed naturally during the period

In both cases, the initial stocking is divided into that surviving to the end of the period (e) and into that dying (a) naturally or in cuttings during a given period. The growth during the given period is divided into the growth of trees surviving throughout the period (Z_E), the growth of trees removed naturally or in cuttings during the period (Z_A) and the growth of ingrowth dying during the period (S_K). Consequently:

- Stocking at the beginning of the period: $B = e + a$
- Stocking at the end of the period: $E = e + Z_E + K$
- Gross growth during the period: $Z_{br} = Z_E + Z_A + K + S_K$
- Total removal during the period: $A = a + Z_A + S_K$
- Net growth during the period: $Z_N = E - B = Z_{br} - A = Z_E - a + K$

Kilkki (1984) demonstrated the performance of the model with a calculation where the initial stocking was 200 m^3 ha^{-1} and the stocking at the end of period was 225 m^3 ha^{-1}. At the middle of the 10-year period, the thinning removal was 50 m^3 ha^{-1}. During this period, the gross growth of stemwood was:

$$\begin{aligned} Z_{br} &= Z_E + Z_A + K + S_K = E - e - K + A - a - S_K + K + S_K \\ &= E - (a + e) + A = 225\ m^3\ ha^{-1} - 200\ m^3\ ha^{-1} + 50\ m^3\ ha^{-1} \\ &= 75\ m^3\ ha^{-1} \end{aligned}$$

Consequently, the mean annual gross growth was 7.5 $\mathrm{m^3\ ha^{-1}\ a^{-1}}$. The difference between the initial stocking and the stocking at the end of the period would give a growth of 25 $\mathrm{m^3\ ha^{-1}}$ over the whole period, which clearly underestimates the gross growth. Regarding naturally growing and developing forests, the growth of natural removal may contribute substantially to the gross growth over a given period as is the case for cutting removal in managed forest. The omission of growth of removed trees may seriously bias the estimates on the potentials of forests to remove atmospheric carbon into the forest ecosystem.

8.5.2 Dynamics of Litter Fall

Litter fall links the autotrophic production with the heterotrophic production, with driving the nutrient cycle for the reuse of nutrients in the physiology and growth of trees. Litter fall refers to the dying of tissues and organs of trees and of whole trees occurring annually, with the transfer of dead material into the detritus pool. In general, the litter is divided into the foliage, branch, and stem and root (coarse and fine roots) litter in regard to their origin. In coniferous forests, the litter fall represents mainly the mass of organs over several years except the fine root litter (e.g. Helmisaari et al. 2009). In deciduous forests, a large part of litter represents the current-year foliage and fine roots. In both cases, the woody mass (e.g. branches) of the ageing mass cohort will reduce gradually and fall away. Dead material in the form of dead branches may remain attached to the stem until braches are decayed enough to be broken by wind and snow loads. In stem, the dead heartwood is transferred to the heterotrophic system after the death of the whole tree. The litter provided by dead trees presents the carbon accumulated over tens of decades or even centuries as in stems, thus representing an important storage of carbon in the forest ecosystem.

In boreal and temperate conditions, litter fall is related to growth following the allometric relationship between the growth and mass between different tree organs, as found by Matala et al. (2008). They analyzed the data available in the literature by relating litter fall (total above-ground litter, kg $\mathrm{ha^{-1}\ a^{-1}}$) with the volume growth of trees in the stand level ($\mathrm{m^3\ ha^{-1}\ a^{-1}}$). Regardless of tree species, the litter fall was closely correlated with the growth of stem wood (Fig. 8.7). They also found a strong linear correlation between litter fall and basal area, stand volume and latitude. On the other hand, the litter fall was not correlated with the age and density of stand, contrary to what is often claimed in the literature. Litter fall seems to be closely related to the functioning and structure of trees in boreal and temperate conditions.

For further applications, Matala et al. (2008) calculated the relationship between the volume growth and litter fall as presented in Fig. 8.8. This was done in order to make it easier to integrate the litter fall with growth and yield applicable in supporting

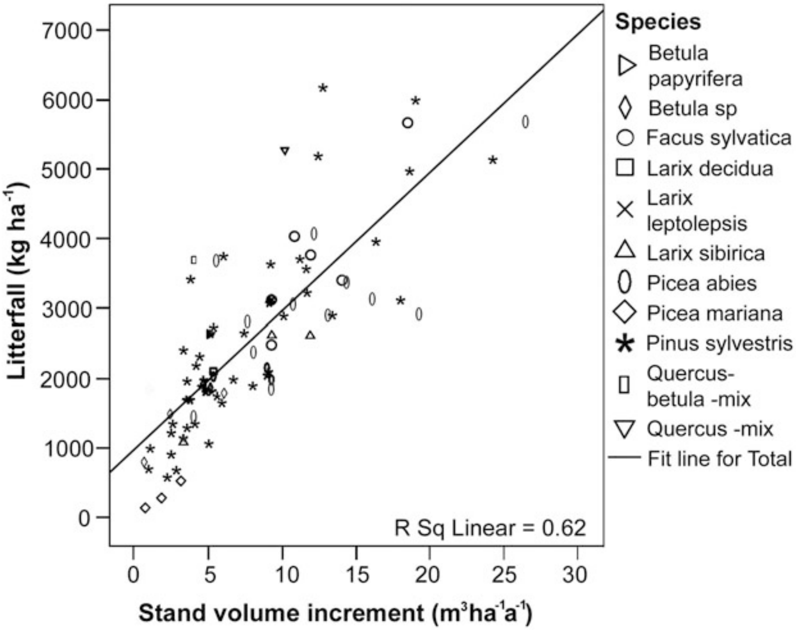

Fig. 8.7 Litter fall in relation to stand volume increment based on the literature representing boreal and temperate zones. (Matala et al. 2008)

the decision-making in forestry. Using linear regression, a non-intercept model was formulated:

$$LF(t) = a \times \Delta V(t) \tag{8.14}$$

where $LF(t)$ is litter fall from a tree (kg a^{-1}), $\Delta V(t)$ is volume growth of a tree (m^3 a^{-1}) and a (kg m^{-3}) is the species-specific parameter. The values of the regression coefficient varied from species to species; i.e. the values were 362 for Scots pine, 284 for Norway spruce and 317 for *Larix* species. Matala et al. (2008) applied the model across Finland using the data of the Finnish National Forest Inventory with the result showing that the mean litter fall on forest land varied from 340 kg ha^{-1} a^{-1} in northernmost Finland to 2300 kg ha^{-1} a^{-1} in southern Finland following an increasing pattern of forest growth from north to south.

8.6 Dynamics of Carbon in Soil

8.6.1 Outlines

Decay of organic matter in soil refers to the processes whereby litter (dead organic matter on soil with recognizable origin such as organs of the tree structure) and

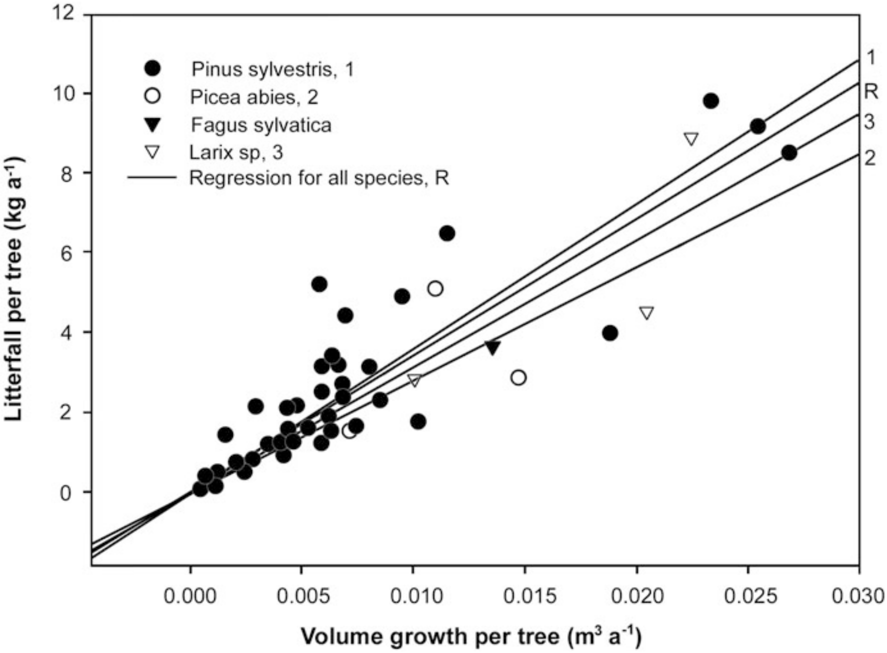

Fig. 8.8 Litter fall calculated per tree as related to volume growth of a tree. The species-specific regression lines in Eq. 8.7 are referred to by numbers and the line *R* represents the linear regression for all species combined in the analysis. (Matala et al. 2008)

humus (dead organic matter on soil without recognizable origin) decompose into CO_2, water, and nutrients (e.g. Pastor and Post 1986). Decay includes leaching, weathering, and biological decomposition of organic matter. Leaching and weathering are mainly physical and chemical processes, where several organic and inorganic substances are released and they increase the nutrient content in soil organic matter. Biological decay represents mainly fungal and bacterial activity (Fig. 8.9), but several invertebrates also graze on the soil organic matter. During the final phases of decay, the nutrients bound in organic matter will be released, and the litter converted into humus colloids. These decay slowly with a life-span of hundreds or thousands of years (Chertov et al. 2001).

8.6.2 Decay of Litter

In the following, the classic paper by Olson (1963) has been used to outline the carbon dynamics in soil. Let X denote the accumulated amount of the soil organic matter per area unit at the moment t. Following Olson (1963), the change of *X* for a discrete time interval (Δt) is:

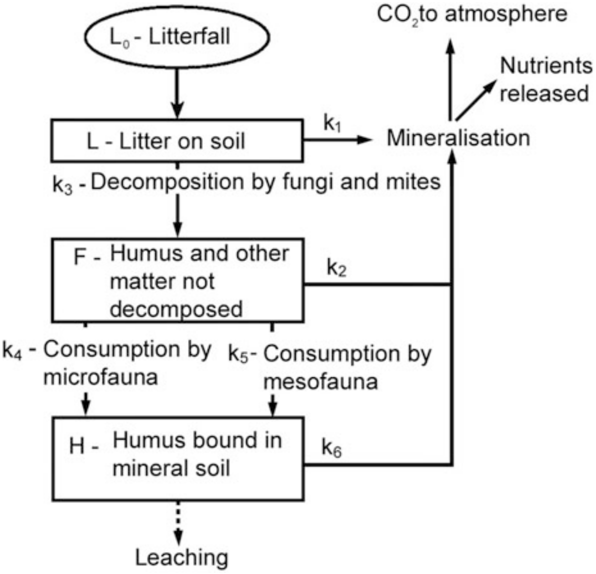

Fig. 8.9 Schematic representations of how litter is converted into humus with release of nutrient and emission of CO_2 to the atmosphere in the soil system under several decomposing processes driven by soil flora and fauna. (Chertov et al. 2001)

$$\frac{\Delta x}{\Delta t} = \text{Increase of matter} - \text{Decrease of matter} \tag{8.15}$$

If L indicates the constant (continuous) flow of litter on the soil, the change of soil organic is:

$$\frac{dx}{dt} = L - kX \tag{8.16}$$

where k is the instantaneous rate of decay or the fractional loss rate of weight. Let X_{ss} refer to the amount of accumulated matter at the steady state, where the flow and weight loss of litter balance each other. Consequently, $X_{ss} = L/k$, which gives the amount of matter at the steady state as a function of constant litter flow and decay rate.

Regarding the decay of a single litter cohort ($L = 0$, no constant flow of matter on soil) with the initial weight X, the constant rate of weight loss is (Olson 1963):

$$\frac{dX}{dt} = -k \times X \tag{8.17}$$

where k is the instantaneous rate of weight loss. Consequently, the constant fractional weight loss of the residual mass is:

$$\frac{dX}{dt} = -kdt \tag{8.18}$$

Let X_o refer to the original weight of the litter cohort. The remaining fraction (0–1) of original weight (X_o, $t = 0$) at the time t is (Olson 1963):

$$\frac{X}{X_o} = e^{-k \times t} \tag{8.19}$$

where X refers to the weight of litter cohort at the time t and k is the constant loss rate of weight (or decay rate). Consequently, the fraction of the original total mass (k') at the time t is:

$$k' = 1 - \frac{X}{X_o} = 1 - e^{-k \times t} \tag{8.20}$$

The relationship between the instantaneous weight loss (k) and the fraction of original total mass (k') is:

$$k = \ln\left(\frac{X}{X_o}\right) = \ln(1 - k') \tag{8.21}$$

8.6.3 Accumulation of Organic Matter on Soil Under Litter Fall

The fractional increase of litter mass on soil relative to the steady state mass under the constant flow of litter (L) is (Olson 1963):

$$\frac{dX}{(X_{ss} - X)} = \frac{dX}{L/k - X} = -kdt \tag{8.22}$$

and the accumulated mass (X) on soil at the time t is (Olson 1963):

$$X = (L/k) \times (1 - e^{-k \times t}) \tag{8.23}$$

Figure 8.10 shows the weight loss of a single litter cohort as a function of the decay rate and the steady state level of mass on soil for varying decay rate k assuming constant litter fall. For example, at the decay rate 0.25 the weight is halved in 2.8 years, whereas at the rate 0.0625 it takes 11.1 years for the weight to be halved. On the other hand, the same time is needed for the accumulation of matter to the value that is a half of that at the steady state; i.e. the half time is obtained from $X/X_o = 0.5 = e^{-k \times t} = -In\ (0.5)/k\ = 0693/k$ based on Eq. 8.10.

8.6.4 Factors Affecting Decay Rate

The decay rate of soil organic matter is a function of the site fertility (edaphic and climatic factors) and the quality of the litter (nitrogen and lignin contents). Furthermore, organisms using organic matter in soil as an energy source affect the decay rate. For example, the dominance of micro-flora implies low decay rate, as is typical for boreal conditions, whereas in temperate conditions bacteria have the dominant role in the faster decay rate (Chertov et al. 2001). On the other hand, the decay rate will be increased under higher temperatures and larger supply of water

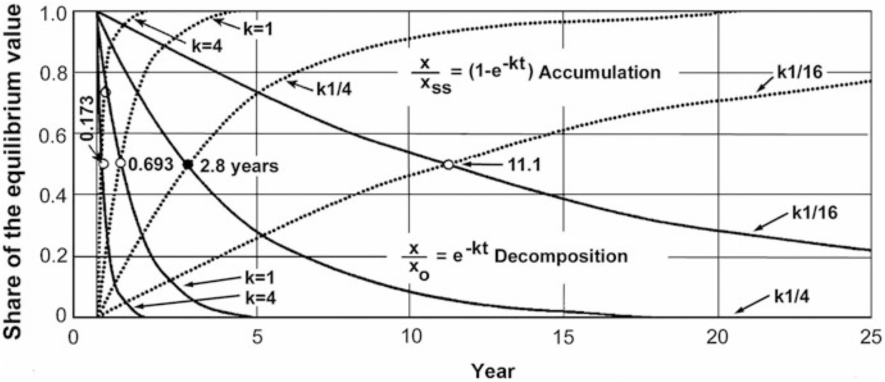

Fig. 8.10 Remaining amount of the original mass of a litter cohort (*solid line*) and the accumulation of the organic matter on soil assuming continuous litter fall (*dotted line*) as a function of decay rate (k) and time (t) in years (Olson 1963). The accumulation of organic matter will be enhanced rapidly if the decay rate decreases. For example, the mass of the litter cohort will decrease by half in 2.8 years if $k = 0.25$ but in 11.1 years if $k = 0.0625$. The same time is needed to achieve the steady state (balance between the litter fall and the decay of litter). The maximum amount of organic matter to be accumulated over time when $k = 0.25$ is smaller than that when $k = 0.0625$, if the same amount of litter fall is assumed

and nutrients. Especially, the decay of litter with a higher amount of nitrogen in relation to lignin will be faster compared with litter with a low nitrogen/lignin ratio, as demonstrated in Fig. 8.11. The simulated values based on the model of Chertov et al. (2001) show that under given temperature conditions, the decay of litter is much enhanced by greater nitrogen content.

8.7 Carbon Sink/Source Dynamics and Carbon Sequestration in Forest Ecosystems

8.7.1 Outlines of Carbon Sink/Source Dynamics

A forest ecosystem is a carbon sink if trees (and other plants) absorb more carbon than they release in autotrophic and heterotrophic respiration; and a carbon source if the carbon absorption is less than is released in autotrophic and heterotrophic respiration. In this context, the sink rate refers to the carbon uptake rate and the source rate refers to the carbon emission rate; the relation between these two is the carbon balance. The carbon balance may be indicated by the balance of carbon fixation and emission rates at a given moment or over a given period. Sink and source rates (carbon dynamics, carbon sink/source dynamics) vary over time depending on the factors driving the carbon fixation rate in photosynthesis and the carbon emission rate in autotrophic and heterotrophic respiration. In the following, a process-based model (Sima, e.g.

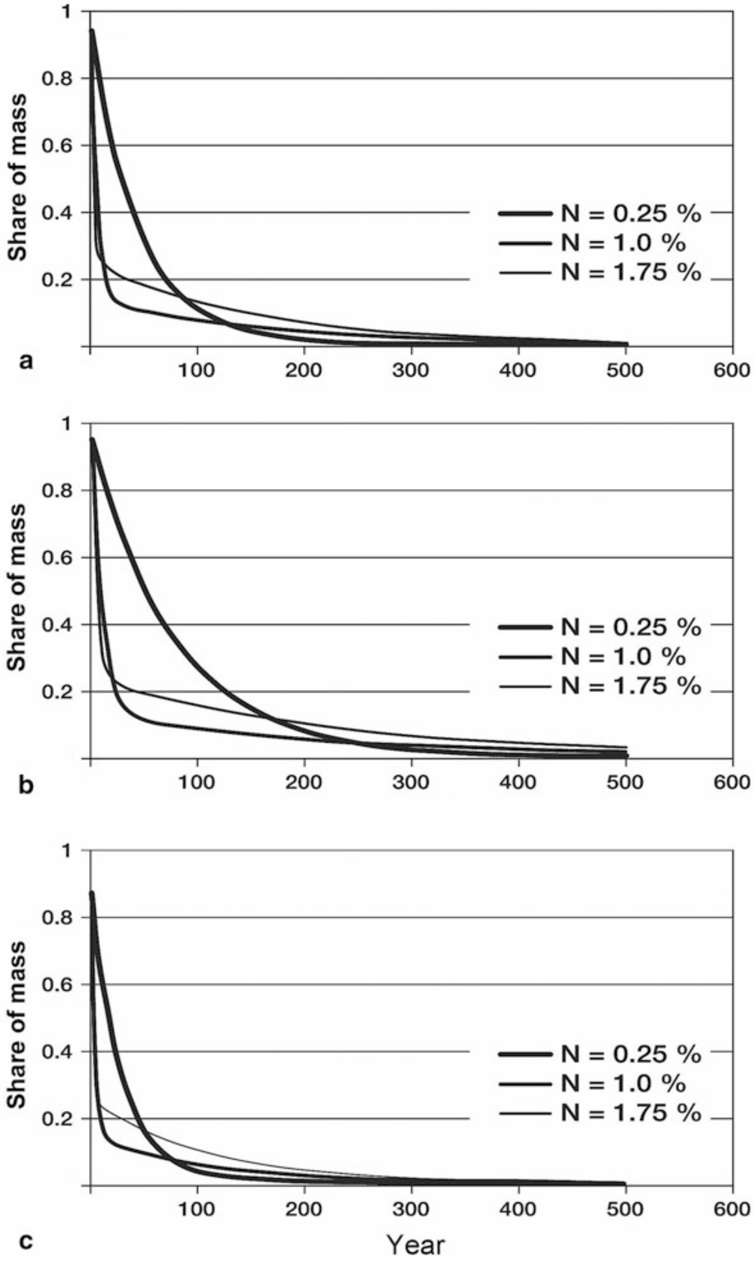

Fig. 8.11 Weight of litter cohort as a fraction of the original cohort as a function of time in years, nitrogen content of litter and climate. (**a**) Under the current climate (boreal conditions in Finland (62° N)); (**b**) Under a climate with temperature 4 °C less than the current climate, (**c**) Under a climate with temperature 4 °C higher than the current climate. (Simulated results based on the FinnFor model (Kellomäki and Väisänen 1997) using the decay model of Chertov et al. 2001)

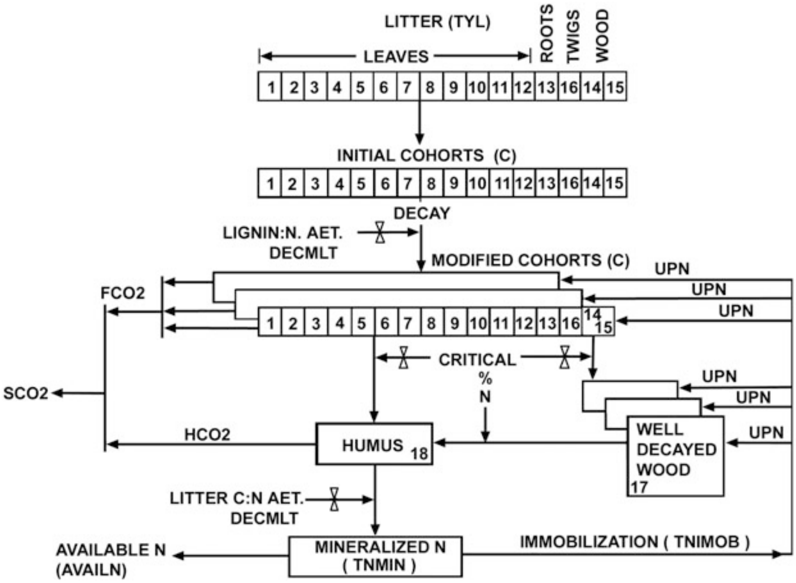

Fig. 8.12 Outlines of the model for the decay of soil organic matter in the Sima model (Pastor and Post 1986; Kellomäki et al. 1992) used to analyze the carbon dynamics in a forested ecosystem

Kellomäki et al. 1992, 2008; Alam et al. 2012) was used to analyze the carbon dynamics in a forested ecosystem over time and changes in carbon dynamics due to the production of timber and biomass. Some main details of the decomposition model are given in Box 8.2.

Box 8.2 Outlines of the model for the decay of soil organic matter in the Sima model (Kellomäki et al. 1992) used to analyze the carbon dynamics in forested ecosystem

The outlines of the decomposition model used in the Sima ecosystem model based on Pastor and Post (1986) (see also Meentemeyer 1978; Meentemeyer and Berg 1986) are presented in Fig. 8.12. Decay determines the weight loss and CO_2 emission from decaying litter and humus. Litter is divided into foliage, twig, root and woody litter. Decay is initiated by calculating the ash-free weight of the cohort. Weight loss (%) is a function of the current ratio between lignin (L) and nitrogen (N) contents (L/N) and evaporation (*AET*). Whenever the nitrogen content in decaying litter in a cohort exceeds the critical value, the organic matter and nitrogen in the cohort is transferred to the organic matter and nitrogen in humus, whose decay is dependent on *AET* and the C/N ratio in humus. The weight loss of litter and humus is converted to CO_2, which is emitted into the atmosphere.

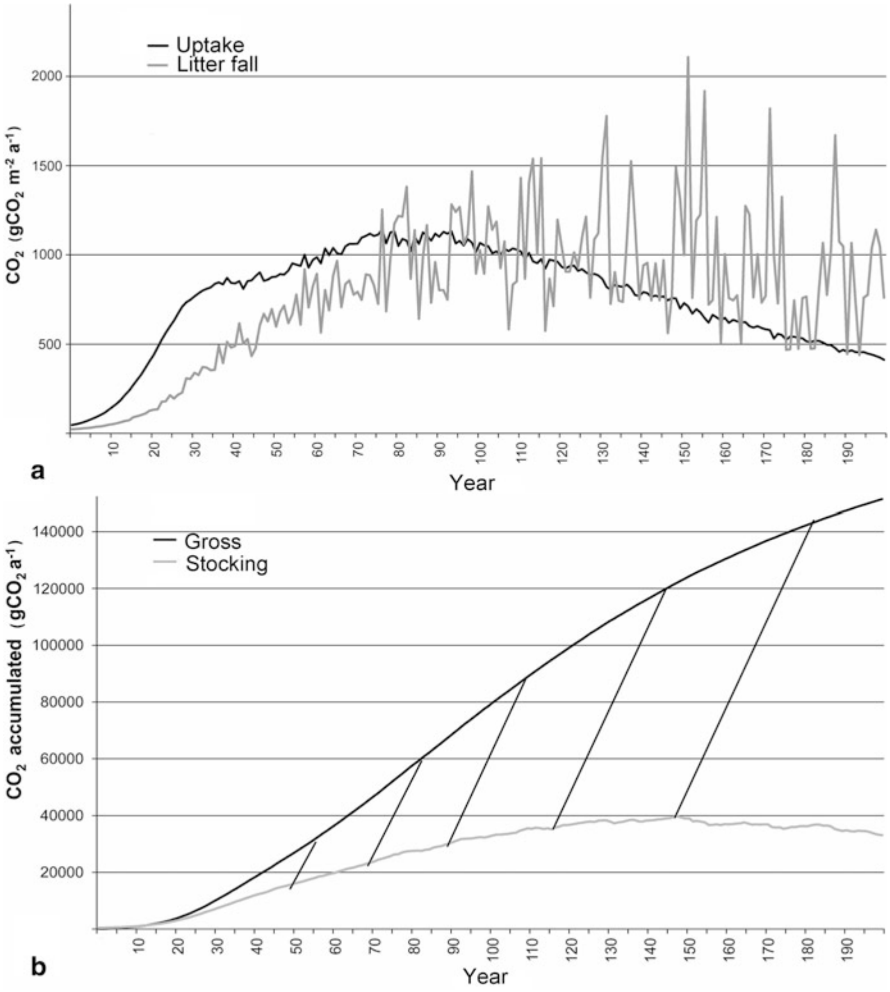

Fig. 8.13 *Panel a*: Uptake and loss of CO_2 in a forest ecosystem, and *Panel b*: accumulated total growth and storage of CO_2 in stem wood. The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) over 200-year period. The initial stand density was 1800 seedlings ha^{-1}. Simulations based on the Sima model. (Kellomäki et al. 1992)

8.7.2 CO_2 Emission From Litter on Soil

Figure 8.13 shows how the litter fall is related to the growth over time if both factors are given in terms of g CO_2 m^{-2} a^{-1}. Typically, the carbon uptake and litter fall are closely related to each other in young forests. In this case, the carbon uptake culminated 40–50 years after initiating the simulation. In this phase, the main part of litter represents foliage and branches and suppressed trees of small dimensions.

Thereafter, larger trees, too, were dying due to reducing space and the maturing of trees as indicated by the large variability in annual litter fall. This implies that the large part of gross growth is lost in litter and transferred into the soil for decomposition. Therefore, the stocking (the biomass in living trees) in naturally developed mature tree stands represents only a small part of total growth during the simulation period. For example, in the boreal conditions in Finland, the mortality of trees with stemwood litter represents 20–40 % of the total stemwood growth over a 100-year period, depending on tree species, site fertility and location. In general, the mortality takes place earlier in birch stands than in Scots pine and Norway spruce stands, earlier on fertile than on poor sites, and earlier in southern than in northern Finland, thus following the growth rate and life span of different tree species in given conditions.

8.7.3 Relation Between CO_2 Uptake and Emission from Ecosystem

Figure 8.14 shows the CO_2 uptake in growth and the CO_2 emission in decomposition of litter and humus. In the very early phase of the simulation, representing young seedlings, the emission exceeds the uptake because growth is still small but there is much litter and humus in the site, originating from the previous tree stand that was harvested before establishing the new tree stand. In fast growing young stand (age 20–60 years), the CO_2 uptake substantially exceeds the emission, which exceeds the uptake once again in more mature phases in the growth and development of the tree stand. In general, this seems to occur when the annual growth of trees falls below the mean annual growth (Kilpeläinen et al. 2011). However, CO_2 continues to accumulate in the trees and to stabilize at the level determined by the stocking of living trees, but fluctuating in relation to the mortality of trees and recovery of canopy. This phase also implies the culmination of mass of CO_2 in trees and soil and slow reduction thereafter due to declining growth and litter fall in a tree stand.

8.7.4 Total CO_2 Emission from Ecosystem

In managed forests, the current growth and development of trees is susceptible to the management history, which affects the current properties of the sites and thus the current carbon dynamics. In this respect, the carbon in soil may be divided into old carbon and new carbon, both of which contribute to the current CO_2 emission. Old CO_2 refers to the CO_2 bound in litter and humus in the time before establishing a new tree stand after a terminal cut, if standwise management is applied in forestry. New CO_2 refers to the CO_2 bound in litter and later in humus representing the time after establishment of a new tree stand. Figure 8.15 shows that the role of old CO_2 in the total emissions is very dominant in seedling stands, but its role will be reduced rapidly and later much exceeded by that of new CO_2 in the total emissions. Finally, the storage of new CO_2 will almost completely replace the storage of old CO_2 in soil.

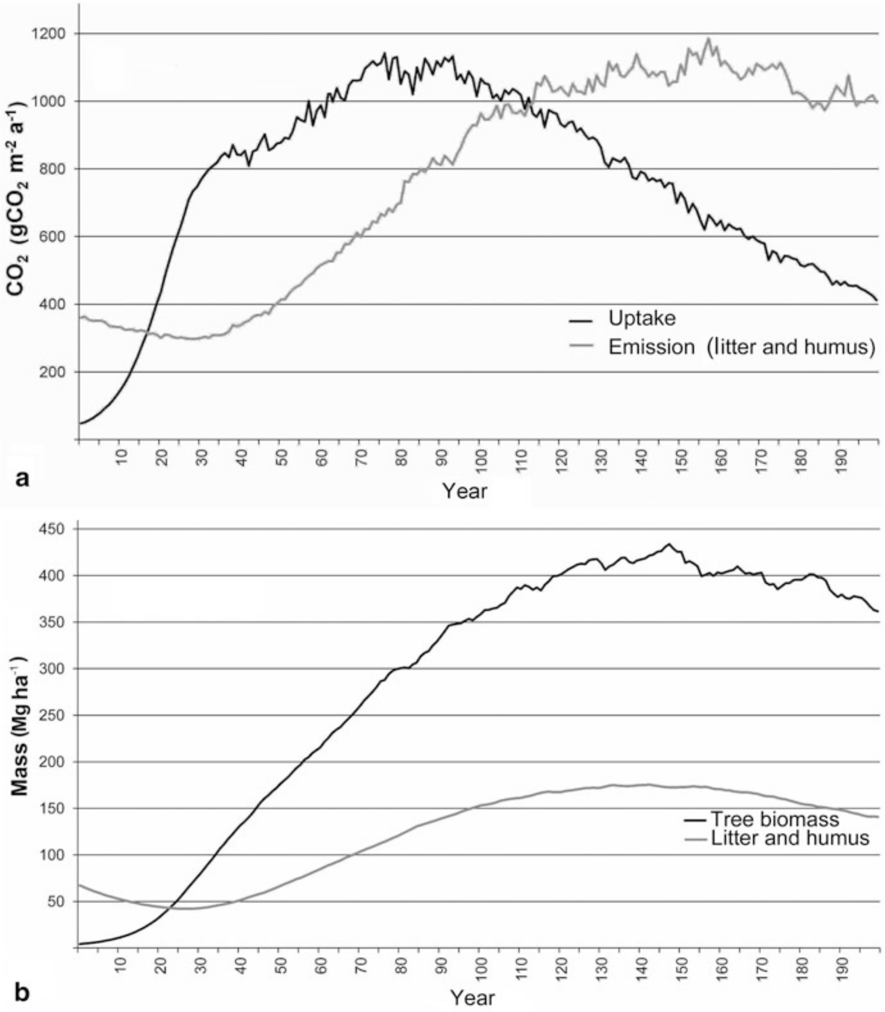

Fig. 8.14 *Panel a*: Annual CO_2 uptake in trees and CO_2 emission from soil (litter and humus), and *Panel b*: Stocking of trees and soil organic matter (litter and humus) over a 200-year period. The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) applying a 200-year rotation. The initial stand density was 1800 seedlings ha^{-1}. Simulations based on the Sima model. (Kellomäki et al. 1992)

8.8 Carbon Dynamics and Sequestration in Managed Forest Ecosystems

8.8.1 Carbon Flow Through Managed Forest Ecosystems

In general, carbon sequestration refers to storage of carbon that would otherwise be released into the atmosphere. Carbon sequestration may be based on natural or

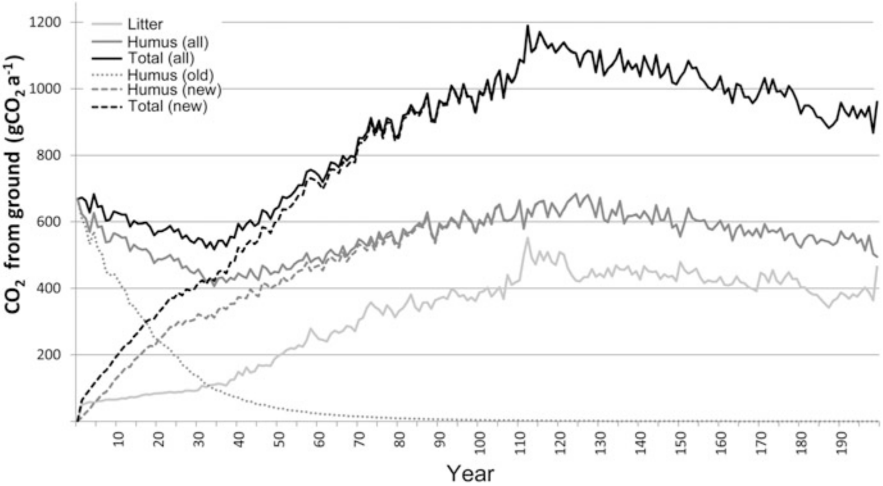

Fig. 8.15 Relationship between the total soil carbon and CO_2 emission, and storage of old and new carbon in soil profile and the emission from them over time. The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) applying a 200-year rotation. The initial stand density was 1800 seedlings ha^{-1}. Simulations based on the Sima model. (Kellomäki et al. 1992)

artificial processes by which CO_2 is absorbed and kept out of the atmosphere, where CO_2 contributes to the greenhouse effect and causes global warming by trapping heat in the lower atmosphere. In forestry, carbon sequestration implies management methods to enhance the CO_2 absorption and to prolong the residence time of CO_2 in the forest ecosystem.

The flow of carbon through the forest production cycle includes the carbon emitted in management/logistics needed in producing timber and biomass, and the uptake and emission of carbon in ecosystem processes controlled by management operations. Furthermore, the track of carbon is often extended to include also manufacturing and use of biomass in the energy industry and/or forest industry. The track of carbon can further include the carbon in abandoned wood-based products and the emission from decaying wood-based products, as demonstrated in Fig. 8.16. In the forest-based production system, the emissions of carbon from the ecosystem processes and from different phases of management, harvesting, logistics, manufacturing and use are balanced by the uptake of carbon in tree growth controlled by the management procedures. Such a life cycle assessment (LCA) in tracking carbon in the context of ecosystem/technosystem interaction with forestry applications is discussed in detail in Chap. 10 of this volume.

Figure 8.17 shows an example of how the carbon source/sink dynamics performs over time in a managed Norway spruce ecosystem on a fertile site in southern Finland, as adapted from Kilpeläinen et al. (2011). The carbon uptake follows the life span of trees; i.e. in young stand the uptake increases to the culmination of growth, and reduces thereafter following the maturation of trees. The uptake was maximized in the

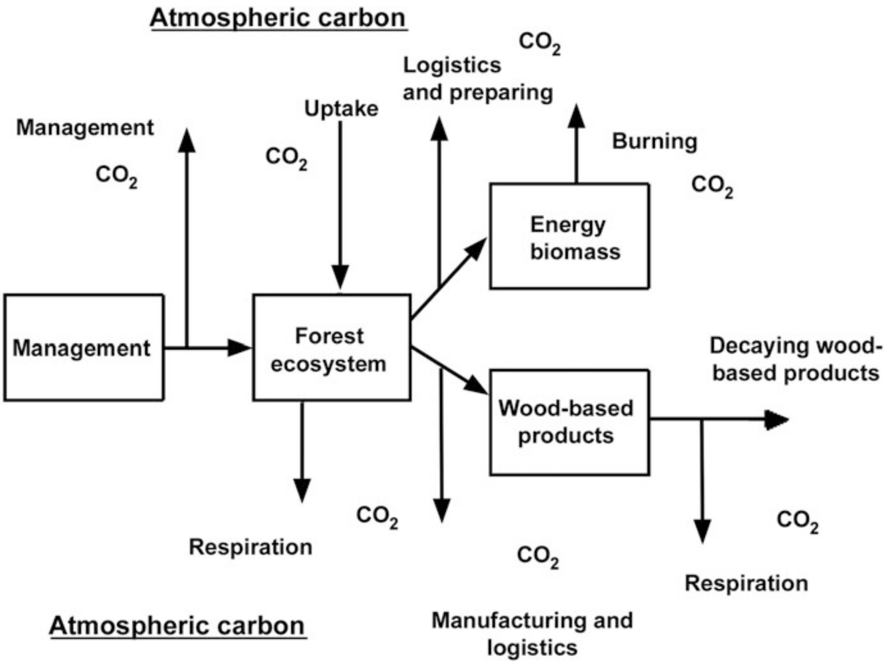

Fig. 8.16 Outline of the forest-based production system, which includes the emissions of carbon from the ecosystem processes and from different phases of management, harvesting, logistics, manufacturing and use. They are balanced by the uptake carbon in tree growth, which is controled by the management procedures

middle of the rotation, the value being about 1770 g CO_2 m^{-2} a^{-1}. Towards the end of the rotation, the CO_2 uptake decreased gradually due to the reducing growth and just before the final felling the uptake was about 1280 g CO_2 m^{-2} a^{-1}. On the other hand, the CO_2 uptake was reduced temporarily in thinning due to the sudden reduction of stocking, but it recovered in a few years along with enhancing growth of the remaining trees. At the same time, the CO_2 emission increased due to the sudden increase of litter in the form of harvesting residues (needles, branches, roots), which increased substantially the decaying matter. Over the rotation, the average CO_2 emission due to the decay of litter and humus was about 730 g CO_2 m^{-2} a^{-1}. The CO_2 emissions due to management were fixed for the planting practices (15 g CO_2 m^{-2}), whereas they varied for harvesting operations from 78 g CO_2 $m^{-2}a^{-1}$ for the first thinning to 239 g CO_2 $m^{-2}a^{-1}$ for the final felling. Regarding the uptake and emissions of CO_2 over the rotation, the average net CO_2 exchange was – 319 g CO_2 m^{-2} a^{-1}. Thus, the forest-based production system was a sink of CO_2, but the sink values varied substantially through the rotation; i.e. during the first 20 years the ecosystem was a source and thereafter a sink with the highest value of – 1100 g CO_2 m^{-2} a^{-1} just before the first thinning. Thereafter, the sink values reduced towards the end of the rotation, and just before the value was – 140 g CO_2 m^{-2} a^{-1}.

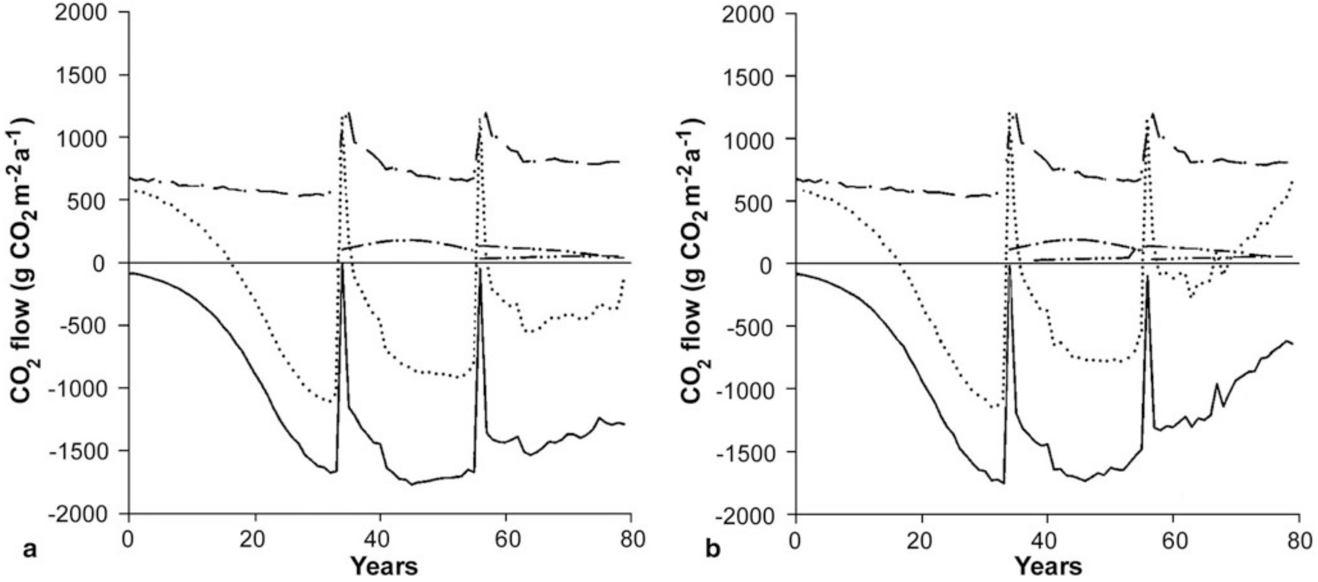

Fig. 8.17 Dynamics of carbon in a forest ecosystem in southern Finland (62° N, 29° E) managed for timber (**a**) under the current climate, (**b**) under the climate change when applying the 80-year rotation in both cases (Kilpeläinen et al. 2011). The site was of medium fertility (*Myrtillus* site type) with soil organic matter (litter and humus) of 67 Mg ha^{-1} in initializing the simulations. The initial density of Norway spruce (*Picea abies*) stand was 2500 seedlings ha^{-1}. In thinning, the rules used in the practical management were used. *Solid line*: carbon uptake in growth; *Dashed line*: carbon emission from decay of soil organic matter; *Dotted line*: net ecosystem exchange (*NEE*)

Kilpeläinen et al. (2011) found a similar pattern to hold in the simulation, where the climate was assumed to warm by 3.4 °C over the growing season (April to September), precipitation to increase by 10 % in the growing season and the atmospheric CO_2 to double from 390 ppm to 840 ppm during the rotation. The main difference in the carbon dynamics between these two climate scenarios was that the climate change enhanced the uptake in early phases of the rotation but reduced later in the rotation faster than under the current climate. This was due to enhanced maturation of trees with shorter life span under the climate change conditions. Again, the CO_2 uptake was the highest at the time of the first thinning, with a value of $-1710\,\mathrm{g\ CO_2\ m^{-2}\ a^{-1}}$. Thereafter, the CO_2 uptake decreased gradually towards the end of the rotation, and it was $-1230\,\mathrm{g\ CO_2\ m^{-2}\ a^{-1}}$ just before the final felling. Over the rotation, the average uptake was $-1040\,\mathrm{g\ CO_2\ m^{-2}\ a^{-1}}$, and the average CO_2 loss was $790\,\mathrm{g\ CO_2\ m^{-2}\ a^{-1}}$. Consequently, the average net CO_2 exchange was $-160\,\mathrm{g\ CO_2\ m^{-2}\ a^{-1}}$, which was slightly larger than that under the current climate.

8.8.2 Retention of Carbon in Managed Forest Ecosystems

Carbon sequestrated in the managed ecosystem may remain in the ecosystem over years or even decades depending on the growth rate of trees, management (enhancement of tree growth, soil management), harvesting (frequency, intensity) and the decomposition rate of litter and organic matter in the ecosystem. The duration of carbon retention in the ecosystem is referred to as the residence time (τ) in the ecosystem (Eq. 8.24), whereas the turnover rate is the inverse value of residence time. The residence time is obtained by dividing the storage of carbon (storage, e.g. $\mathrm{kg\ CO_2\ ha^{-1}}$) in the ecosystem by the emission rate (q, e.g. $\mathrm{kg\ CO_2\ ha^{-1}\ a^{-1}}$):

$$\tau = \frac{\mathit{Capacity\ of\ a\ system\ to\ hold\ carbon}}{\mathit{Rate\ of\ carbon\ flow\ through\ a\ system}} = \frac{\mathit{Storage}}{q} \tag{8.24}$$

In calculating the residence time, the system is assumed to represent steady state conditions. The residence time begins at the moment when carbon enters the system and ends at the moment when carbon leaves the system. Under this assumption, typical values of carbon residence in forest ecosystem are listed in Table 8.4 assuming the steady state of the ecosystem.

Under management, the forest is unbalanced representing varying amounts of carbon in the ecosystem depending on uptake rate and flow rate in relation to the phase of ecosystem development. The uptake and flow rates are related to the management and harvesting operations affecting the carbon storage, uptake and emission of carbon. Regarding new carbon, the residence time varies over time:

$$\tau(t) = \frac{\sum S(n,t)}{\frac{\sum q(n,t)}{n}} = n \times \frac{\sum S(n,t)}{\sum q(n,t)} \tag{8.25}$$

Table 8.4 Residence time and turnover of carbon in a forest ecosystem (Raich and Schlesinger 1992). Residence time and turnover rates are estimated by assuming that 30 % of soil respiration represents root respiration

Forest ecosystem	Soil carbon, C kg m^{-2}	Soil respiration, g C m^{-2} a^{-1}	Residence time, a	Turnover rate, a^{-1}
Boreal forests	20.6	322	91	0.011
Temperate forests	13.4	662	29	0.034
Tropical lowland forests	28.7	1092	38	0.026

where $S(n, t)$ refers to the mass in the cohort n (living, litter, humus) in the year t since the initiation of the cohort ($t = 0$) and $q\ (n, t)$ refers to the mean emission rate of CO_2 from the cohort $S(n, t)$.

Figure 8.18 shows, that the accumulation of new carbon in the ecosystem is a function of the initial stand density. On the other hand, thinning reduces the carbon accumulation; i.e. at the end of a 100-year rotation the amount of carbon at the initial stand density of 1800 seedlings ha^{-1} with basic or current thinning (130 Mg ha^{-1}) was clearly less than that under no thinning (150 Mg ha^{-1}). At the same time, the carbon emissions exceeded temporarily the emissions under thinning, but in the long term they stabilized lower than those under no thinning due to reducing carbon emission from decaying harvest residue. If the stocking in the stand under thinning is higher than under current thinning (basal area before and after thinning remain higher than that under current thinning), the stabilized carbon emission converged with that under no thinning.

By applying Eq. 8.24 one obtains the dynamics of residence time for the new carbon presented in Fig. 8.19 for Norway spruce. In young stands, not yet thinned or later left unthinned, carbon has a short residence time compared with that later due to the large share of foliage litter with a high rate of decay in relation to stem wood, whose amount was still small. On the other hand, the residence time in young stands increased along with the increasing initial stand density due to higher amount of stem wood. Over time, the share of foliage litter reduces and the share of woody litter increases, with a consequent increase of the carbon residence time. After the culmination of stem wood growth, the residence time stabilizes towards to the end of the rotation in the range 50–60 years under no thinning in such a way that the stabilization was earlier at the initial density of 1800 seedlings ha^{-1} with thinning than under other management regimes. Under thinning the residence time varied from 20 years just after thinning intervention up to 60 years just before the thinning intervention, the lowest values representing the first thinning in stands of high density.

Over the whole rotation, the mean residence time was about 50 years under no management regardless of the initial stand density (Table 8.5). Thinning reduced the mean residence time up to 15 % compared with that under no thinning, the most under current thinning. This implies that management based on the frequency and intensity of thinning and rotation time has a clear effect on the residence and turnover of carbon in the forest ecosystem.

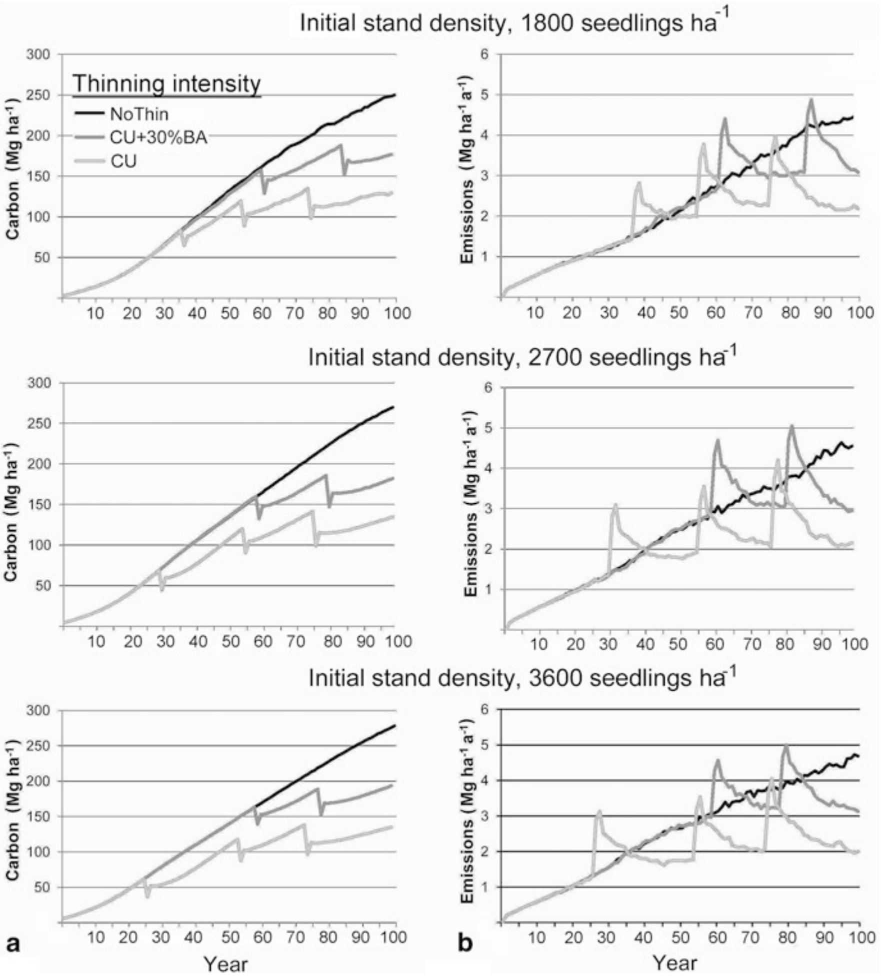

Fig. 8.18 Stocking (trees and soil) of new carbon (**a**) in the Norway spruce stand and the carbon emissions (**b**) from new carbon as a function of the initial stand density and varying thinning intensity. The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) applying a 100-year rotation. The current thinning (*CU*) follows that used in the practical management and no thinning (*NoThin*) refers that thinning has not been done during the rotation. Higher stocking in thinned stand (*CU* + *30%BA*) indicates that the basal area (*BA*) before and after thinning remains 30 % higher than that under current thinning. Simulations based on the Sima model. (Kellomäki et al. 1992)

8.8.3 Effect of Harvesting of Energy Biomass on Carbon Residence in Forest Ecosystems

In conventional forestry the focus is on timber production, where the residues (foliage, branches, small-dimensioned top part of stems, stumps) are left to decay in

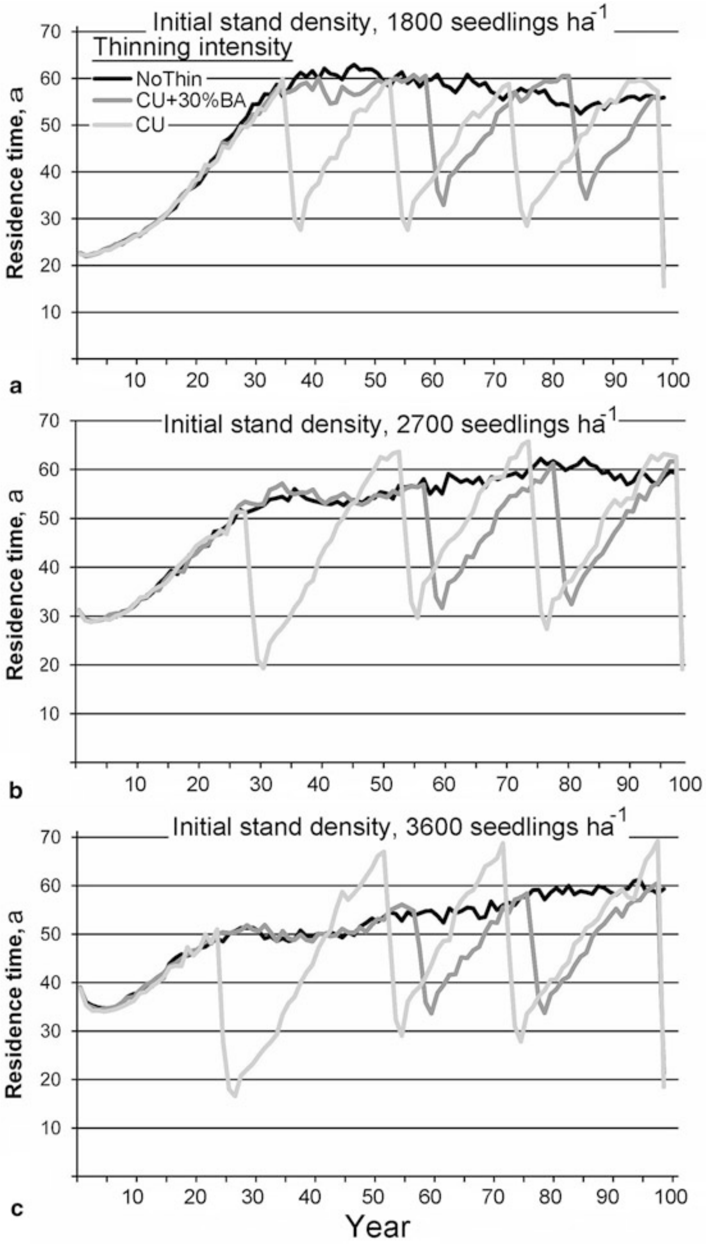

Fig. 8.19 Time series for the carbon residence time in the Norway spruce stand and the carbon emissions from new carbon as a function of the initial stand density and varying thinning intensity. The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) applying a 100-year rotation. The current thinning (*CU*) follows that used in the practical management and no thinning (*NoThin*) refers that thinning has not been done during the rotation. Higher stocking in thinned stand (*CU* + *30%BA*) indicates that the basal area (*BA*) before and after thinning remains 30 % higher than that under current thinning. Simulations based on the Sima model. (Kellomäki et al. 1992)

Table 8.5 Mean amount of carbon and mean carbon residence time in the ecosystem as a function of management regime. In parentheses, the percentage of mean carbon and mean residence time under a no-thinning regime

Management regime	Mean amount of carbon over rotation, Mg ha^{-1}	Mean residence time of carbon over rotation, a
Initial density 1800 seedlings per ha		
No thinning	127 –	51 –
Current thinning	83 (65)	45 (88)
Current thinning + 30 %	107 (84)	49 (96)
Initial density 2700 seedlings per ha		
No thinning	135 –	52 –
Current thinning	86 (64)	45 (87)
Current thinning + 30 %	113 (84)	47 (90)
Initial density 3600 seedlings per ha		
No thinning	139 –	51 –
Current thinning	85 (61)	45 (88)
Current thinning + 30 %	118 (85)	47 (92)

the forest. Thus, such residues have an important role in storing carbon in managed forests. This benefit is partly lost if the residues are used in energy production. The impact of stump extraction may be especially important, because the wood biomass in stumps is 20–40 % of the total tree biomass depending on the species and maturity of trees. This is demonstrated in Fig. 8.20, which shows the net ecosystem exchange in Norway spruce stand under varying management, including regimes with and without stump extraction. Stump extraction along with the delayed harvest reduced the CO_2 emission, and thus increased the CO_2 uptake in relation to emission compared with the current thinning practices, where stumps remain in the forest. The same pattern was repeated over both production cycles.

8.9 Conclusions

Managed forests provide biomass for energy production even when they are managed only for timber. The potential for energy biomass may be substantially increased by a management regime that combines the production of timber and energy biomass. However, the carbon sink/source dynamics in managed forests varies substantially over the whole production cycle following the amount of growth, litter fall and emission due to the decay of organic matter in the soil. The harvest of forest biomass for energy alters the carbon sink/source dynamics and the consequent carbon balance in the ecosystem, e.g. thinning and consequent harvest decreases the amount of growth and litter on the soil. The mitigation capacity of forests is thus affected by the ecosystem dynamics, management and harvest, including carbon emissions from management, harvesting and logistic operations. Therefore, it is important that the effects of management on carbon stocks are carefully considered when assessing of net atmospheric impacts of bioenergy production on carbon sequestration in managed

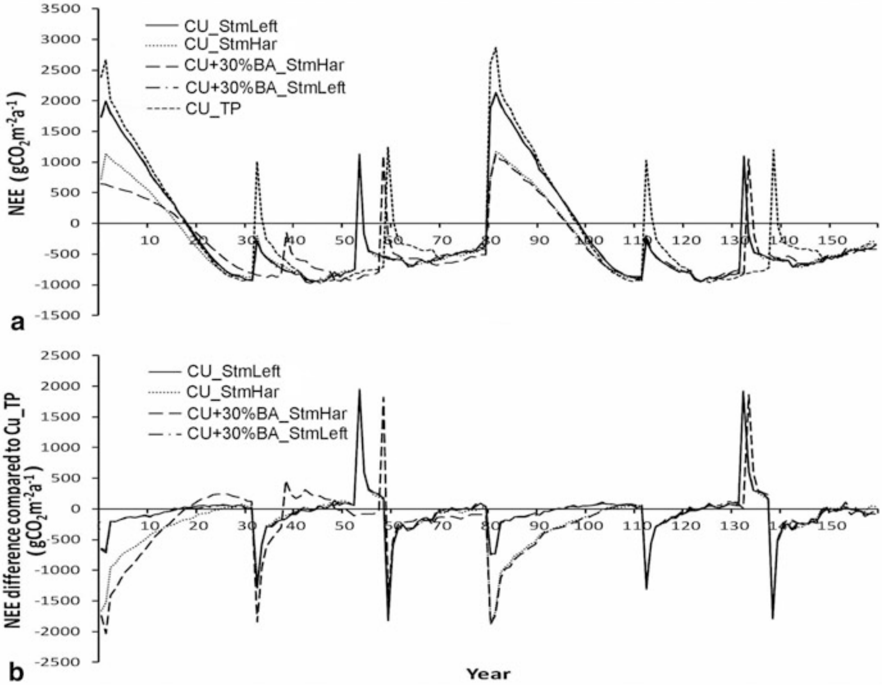

Fig. 8.20 Net ecosystem exchange of carbon (*above*) and in Norway spruce (*Picea abies*) ecosystem under varying management regimes, and the difference in net ecosystem exchange between the current management for timber only and the management for energy biomass including and excluding stump harvest (*below*). Calculations are done with the CO_2 Emission Calculation Tool for the life cycle assessment (LCA) developed by Kilpeläinen et al. (2011). The simulations concern a site of medium fertility (*Myrtillus* site type) in central Finland (62° N, 29° E) when applying two 80-year rotations one after another. The initial density of stand was 2500 seedlings ha^{-1}. In thinning, the rules used in practical management were applied. *CU_StmLeft:* bioenergy harvesting, current basal area (*BA*) thinning thresholds, stumps left on site; *CU_StmHar:* bioenergy harvesting, current basal area thinning thresholds, stumps harvested; *CU + 30%BA_StmHar* : bioenergy harvesting, 30 % increased thinning basal area thinning thresholds, stumps harvested; *CU + 30%BA_StmLeft*: bioenergy harvesting, 30 % increased thinning basal area thinning thresholds, stumps left on site, *CU_TP*: traditional timber harvesting, no bioenergy harvested

forest ecosystems. Full understanding of these underlying processes helps to identify the potential of forest ecosystems to produce biomass for energy, along with other products and services (e.g. pulp wood and timber), and simultaneously to mitigate climate change.

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Chapter 9
Stump Harvesting, Bioenergy Feedstock and Sequestration of Carbon in Soil

Ashraful Alam, Seppo Kellomäki and Antti Kilpeläinen

Abstract In the Nordic countries, the potential of stump biomass as a source of forest biomass for energy use is higher than that of other components of logging residues. In this chapter, we mainly discuss the potential of stumps as a feedstock for energy biomass, measures to improve the quality of energy biomass represented by stumps, and the impacts of stump harvesting on the potential of soils for carbon sequestration in the forest ecosystems. Furthermore, the ecological advantages and disadvantages of stump harvesting for energy biomass are discussed with regard to the use of stumps in substitution for fossil fuels in energy production.

Keywords Bioenergy · Biomass · Boreal forests · Carbon sequestration · Harvesting of stump · Logging residues · Nordic countries · Norway spruce · Quality of stump · Scots pine · Soil carbon · Substitute · Supply chain

9.1 Stumps and Roots—a Source of Biomass for Energy

Global environmental policy has a target to limit global warming to an increase not greater than 2 °C by the end of this century. The use of biomass as a substitute for fossil fuels is among many options to reduce the emissions of greenhouse gases that are behind the climate warming. Forest biomass as a renewable energy source provides great opportunities to substitute fossil fuels in the Nordic countries. In Finland and Sweden, the land area available for wood production is about 22 million ha with growing stock (stem wood) of 2300 and 2900 million m^3, respectively (Ylitalo 2011; Wigrup 2012). Currently, in both countries, forest biomass such as logging residues

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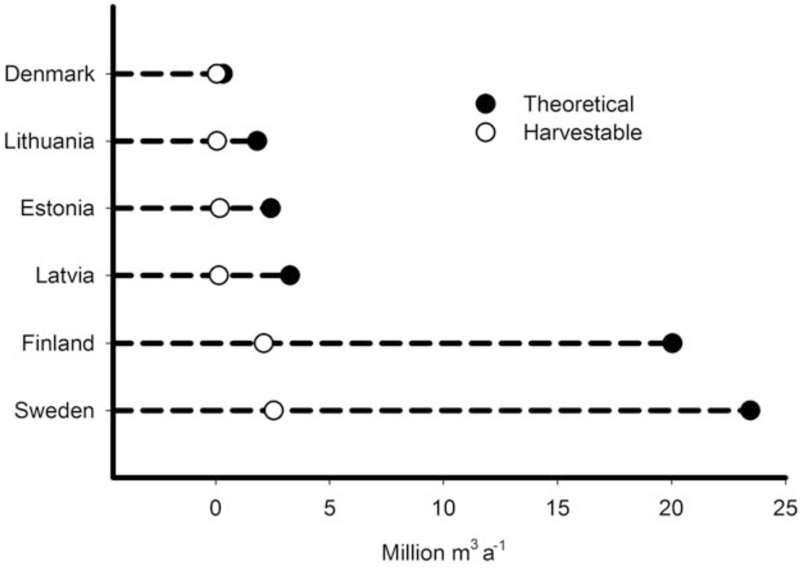

Fig. 9.1 Theoretical and harvestable potential of stump and root biomass in the Nordic and Baltic countries (data from Asikainen et al. 2008). In estimating harvestable (or available) biomass, impacts related to nutrient loss, steep slopes including sensitive sites, were embedded in the reduction factors

is widely used in energy production. The majority of logging residues are harvested in the clear cuts used to harvest timber. Logging residues include the top part of stems, branches and foliage, but stumps and even a part of coarse roots are harvested for energy biomass. In fact, stumps are potentially a major source of raw material (Fig. 9.1), and they may provide even more energy biomass than the other components of logging residues in a clear cut (Hakkila 2004).

Harvesting of stumps (or stump harvesting) for energy biomass refers to the lower part of the stem not harvested for timber, including tap roots and coarse lateral roots (Hakkila 2004). In Finland, stump harvest is currently practiced in 14 % of the clear-cut areas (145,000 ha). Unlike in Sweden and other Nordic and the Baltic countries, harvesting of stumps and coarse roots is increasing in Finland, and about 1 million m^3 of stumps and coarse roots were harvested in 2010 (Ylitalo 2011) (Fig. 9.2). This is about half of the potential amount of stump biomass harvestable regarding the technical and economic limits for using stumps in energy production (Hakkila 2004).

The importance of stump biomass in energy production is widely accepted. However, the use of stump biomass is also questioned due to the impacts of stump harvesting on the structure and functioning of forest ecosystems. Among the main issues, it is claimed that the detrimental effects of stump harvesting on the sequestration and storage of carbon in soils undermine the benefits of the use of stump biomass in energy production as a substitute for fossil fuels and for reducing carbon emissions (e.g., Repo et al. 2012; Vanhala et al. 2012). In this regard, stumps and coarse roots

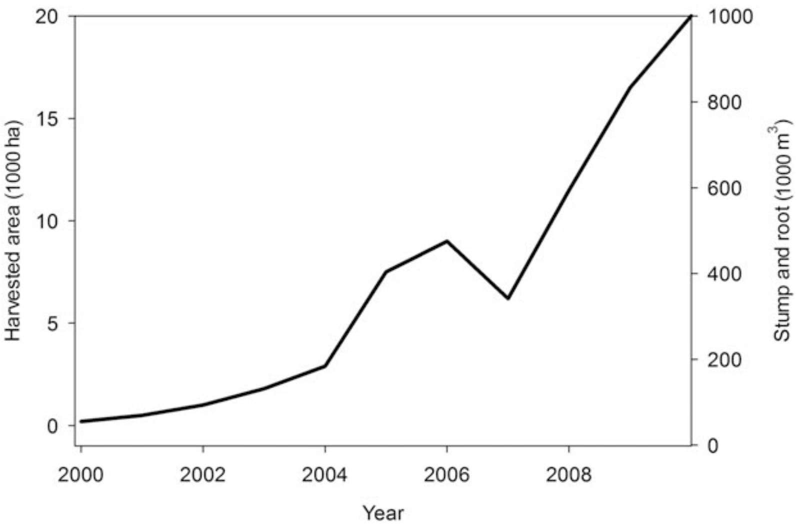

Fig. 9.2 Annual harvest of stumps and coarse roots during 2000–2010 in Finland for the purpose of energy production. (Data from Ylitalo 2011)

are an important part of the total reservoir of carbon in forest ecosystems, which is partly lost when the stumps are harvested and used in energy production. Thus, the carbon storage potential of soil decreases over decades when the difference in time needed for the full decomposition of stumps in the ground and the instant combustion of stump biomass are compared (e.g., Melin et al. 2010; Alam et al. 2013). It is therefore important to understand how sustainable the stump harvesting is, regarding carbon sequestration and reducing carbon emissions by substituting this material for fossil fuels. In this chapter, these questions are discussed in order to identify the advantages and disadvantages of using stumps for energy biomass.

9.2 Stump Harvesting

9.2.1 Potential Feedstock for Forest Bioenergy

The potential of a forest ecosystem to sequester and store carbon depends on the tree species, the growing conditions and the management controling the overall ecosystem dynamics. Figure 9.3 shows the share of biomass in different tree components in a managed forest, with and without stumps in harvest residues, assuming the harvesting of timber alone (pulp wood, saw logs) or harvesting of timber with logging residues. In the former case, the energy biomass harvest is about 30 % of the total biomass stocking, whereas in the latter case it is more than 40 %.

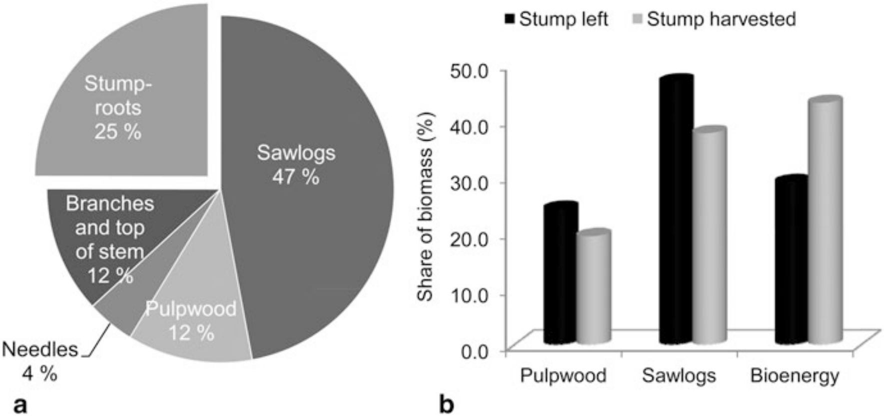

Fig. 9.3 (**a**) Share of pulpwood, sawlogs and energy biomass (needles, branches and top of stem, and stump-roots) at clear-cut areas in Norway spruce stands. (**b**) Share of energy biomass if stumps and roots were left in the forest or harvested. Simulated results are based on the Sima model (Kellomäki et al. 1992, 2008) and followed current thinning recommendation for Finland (Tapio 2006) for the fertile (*Oxalis-Myrtillus type,* OMT) stand located in Joensuu (62°39'N, 29°37'E), Finland

Table 9.1 Biomass (Mg ha^{-1}) of different tree components in a mature and naturally established mixed Scots pine and Norway spruce stand as determined by the sample tree method and the allometric functions of Marklund (1988) (Adapted from Liu and Westman 2009). The stand is located in southern Finland (61°50'N, 24°22'E) on a south-facing slope with an average inclination of 3.4 % and a mean elevation of 152 m above sea level. Site type changed along the slope, from poorer site type (*Vaccinium type,* VT) on the top of the slope, and *Myrtillus type* (MT) to better site type (*Oxalis-Myrtillus type*, OMT) at the bottom (based on Finnish site classification, Cajander 1949). In the region, mean annual temperature is 2.9 °C and mean annual precipitation 709 mm. Of a total of 792 stems in the study area of one ha, 589 are Norway spruce and 203 are Scots pine, and the tree age ranged from 100–140 years

Estimation method	Biomass, Mg ha^{-1}				
	Branches & needles	Stem biomass	Stump & coarse roots	Medium & fine roots	Total
Sample tree method					
Scots pine	15	90	27	2	134
Norway Spruce	19	66	18	3	106
Allometric functions					
Scots pine	14	60	22	–	95
Norway Spruce	23	46	20	–	88

Generally, in a managed and mature boreal forest ecosystem, stumps and coarse roots represent about 20–30 % of the total tree biomass (Hakkila and Aarniala 2004) (Table 9.1). Regarding the harvest of energy biomass, the stumps and roots of Norway spruce are preferred mainly due to its higher biomass in coarse side roots and its loosely anchored rooting system (Fig. 9.4), which makes the extraction of stumps easier compared with Scots pine. However, the heating value of the stump biomass of Scots pine (5.84 kWh kg^{-1} of dry biomass) is slightly higher than that of Norway spruce (5.37 kWh kg^{-1} of dry biomass) (Nurmi 1997).

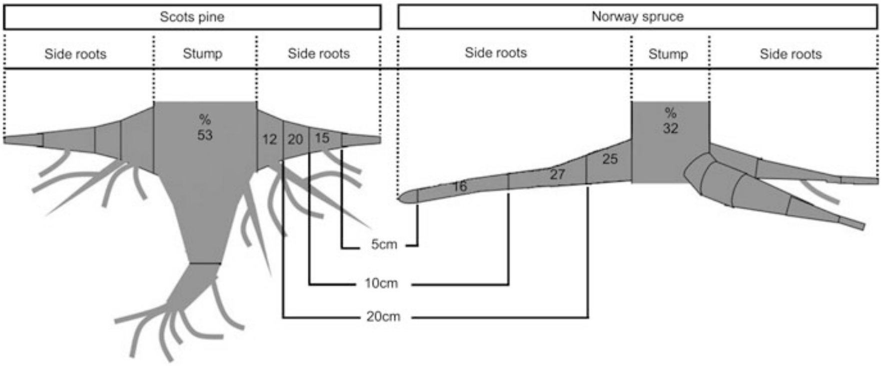

Fig. 9.4 Share of biomass in stumps-root system of mature Scots pine and Norway spruce species. (Redrawn from Hakkila 2004)

In Scots pine, the stump includes more than 50 % of the combined mass of harvestable stump and coarse roots (side roots), whereas in Norway spruce the share of stump is 30 % (Fig. 9.4). This implies that in Scots pine the share of harvestable coarse roots (47 %) is clearly less than in Norway spruce (68 %). Regarding the tap root of Scots pine, the parts deep in the soil profile are not harvestable due to their strong anchorage.

9.2.2 *Quality of Stump Biomass as Bioenergy Feedstock*

Stumps are mainly harvested with excavators equipped with a stump-lifting head to facilitate the extraction. During harvesting, the machine operator places the lifting head under the stump to loosen the root anchorage and take the stump out of the ground when it is ready for lifting. Afterwards, the lifted stumps are shaken in the air with jerky movements helping to dislodge the soil and other contaminants before the stump is cracked into pieces and stored at a nearby site. Usually, the stumps are stored in piles for a season or two before they are forwarded to the road side and finally to the power plant. In between harvesting and shipping to the power plant, several approaches are used to ensure the quality of the stump-root biomass.

The key parameters used to indicate the quality of stump biomass as a feedstock for bioenergy are heating value, and the contents of moisture and ash (Laurila and Lauhanen 2010; Anerud 2012). Heating value (or net calorific value) indicates energy that can be generated without the condensation of vapor. The calculation is done as the gross heating value minus the heat released in the condensed water in combustion (Nurmi 1997). Moisture content is simply the amount of water in relation to the total weight of the biomass, whereas ash content relates to the amount of impurities in the biomass.

The heating value of biomass mostly depends upon its moisture content. About half or more of the fresh biomass consists of water, which substantially decreases the heating value of fresh stumps (Hakkila 1989; Nurmi 2000). There is a linear correlation between the moisture content of biomass and the heating value of forest biomass; i.e. a 1 % increase in moisture content reduces the heating value by 0.0594 MWh Mg^{-1} (Laurila and Lauhanen 2010). Therefore, splitting stumps into pieces may increase substantially the heating value of stump biomass due to enhanced drying compared with that of biomass representing non-split stumps in a single piece (Anerud and Jirjis 2011). Sufficient drying occurs in piles at the harvesting site, where water in the biomass evaporates due to solar radiation and wind. In addition, the open-air on-site storage of stumps facilitates removal of impurities by rain water and partly by the force of the wind.

The presence of impurities (e.g. soil particles) in burning decreases the heating quality of stump biomass due to increased ash content. Therefore, power plants combining the production of heat and power (CHP) may not be interested in stump biomass if the percentage of impurities is too high. The impurities reduce the burning capacity of utilized biomass, even though it is highly dependent on the technology used in burning. In Finland, for example, the most used technology is the fluidized bed boiler where a broad variety of biomass as a bioenergy feedstock is acceptable (IEA 2009, 2010). In general, this is not the case in the small-scale power production plants, which limits the use of stump biomass in them. Harvesting and post-harvesting methods could, however, be developed further in order to improve the quality of the feedstock based on stump biomass. For example, vibrating stumps during harvesting and storing stumps outdoors helps to remove most impurities.

9.2.3 Emission from Supply Chain

The supply chain of forest biomass includes several phases, where fossil fuels are utilized to obtain forest biomass for energy use. Direct carbon emissions from the use of machines in harvesting and logistics affect the total carbon balance of the use of forest biomass in energy production. The typical supply chain includes harvesting by excavator, forwarding to the road side by forwarder, and transporting to the power plant by trucks. Additional emissions are due to the chipping of biomass either before or after the transportation to the power plant. Depending on the specific requirements and machinery used, the supply chain can be different but the magnitude of the difference may not have any considerable effect on the total emissions in the production and management chain. These machine-related emissions have to be counted when the carbon benefits of stump utilization are compared with fossil-fuel energy.

The harvesting and transportation of stump biomass is different from supply chains optimal for other sorts of logging residues. For example, the vibration to reduce impurities is necessary only for stump biomass. Apart from that, the size of truck load may vary extensively whether transporting logging residues other than stumps. The irregular shapes of stumps with attached roots may reduce the mass of the energy biomass in the load size upto 40 %, if the chipping is done at the power plant

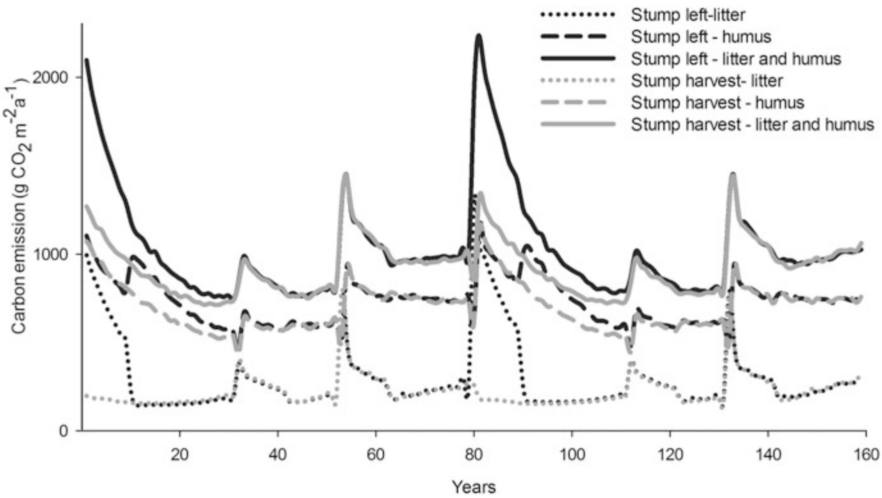

Fig. 9.5 Carbon emissions from soil due to the decomposition of litter and humus in management regimes with stump harvest and no stump harvest. In the simulation, stump-root system was also harvested before starting the simulation for management that collected stump-root system and vice versa. Simulated results are based on the Sima model (Kellomäki et al. 1992, 2008) and followed current thinning recommendations for Finland (Tapio 2006) for the fertile (*Oxalis-Myrtillus type,* OMT) stand located in Joensuu (62°39'N, 29°37'E), Finland

(Väkevä et al. 2004). Regarding stump biomass, the total carbon emissions from the whole supply chain, from extraction through transportation to power plant, are 4–20 kg CO_2 MWh^{-1} (Forsberg 2000; Wihersaari 2005; Alam et al. 2013). Moreover, Alam et al. (2012) estimated that one unit of fossil energy could produce roughly 30–40 units of forest biomass used in energy production, but the values vary substantially depending on tree species and their growing condition, and the moisture content in the biomass.

9.3 Stump Harvesting and Soil Carbon Sequestration

In the Nordic countries, forest soils usually contain more carbon than the aboveground biomass. The build-up of soil carbon storage has probably taken hundreds of years as a result of small litter fall, while slow decomposition has enhanced the accumulation of organic matter on soil. Stump harvesting indicates an enhancement in management and use of forest resources, with impacts on the carbon dynamics in the forest ecosystem including the carbon emissions. Figure 9.5 shows the emission of carbon from the forest ecosystem in two successive rotations assuming no stump harvest or stump harvest before the start of the first rotation and at the end of the first and second rotation. Cutting with stumps left in the ground increased the carbon emissions substantially for several decades due to the decay of stumps compared with cutting with the extraction of stumps. In the latter case, all the carbon in stumps

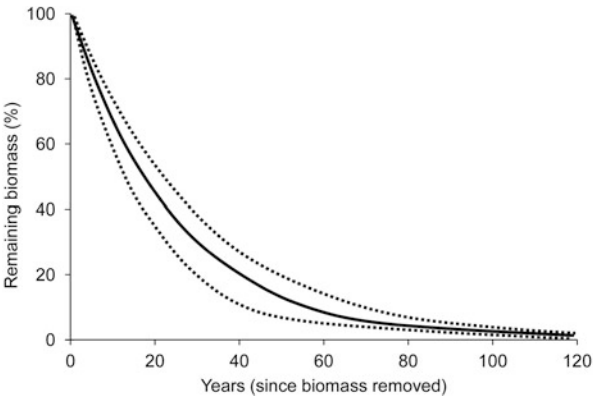

Fig. 9.6 Decomposition functions and annual decomposition rate of logs and stumps of Norway spruce trees in Nordic conditions of Finland, Sweden and north-west Russia. Dotted lines represents higher and lower decomposition rates and solid line indicates the average decomposition rate found in the literature. Age of the oldest sample tree varies from 32 to 60 years. (Adapted from Melin et al. 2009)

will be emitted immediately if they are burnt into energy, whereas the same amount of carbon will also be emitted over a period of decades if stumps are left to decay in the forest ecosystem.

The decomposition rate of stump wood is substantially slower than that of other components of logging residues (stem top part, branches, foliage) mainly due to the larger size of stumps and their high lignin content with high resistance to decay. The available literature shows that the annual relative decomposition rate for Norway spruce logs and stumps is in the range 3.2–5.2 %, which means that almost 80 % of the mass, depending on the studied area, is lost within 30 to 50 years (Krankina and Harmon 1995; Naesset 1999; Harmon et al. 2000; Yatskov et al. 2003; Shorohova et al. 2008; Melin et al. 2009) (Fig. 9.6). Furthermore, the prevailing climatic conditions (temperature, humidity) and the fertility of site affect the decay rate of stump biomass. In the Finnish conditions, the decay rate is slower on the poor sites (*Vaccinium* type) than on the fertile sites (*Myrtillus* type). Obviously, changing climate with higher temperatures and higher precipitation may accelerate the decay process of stump wood and enhance the carbon cycle in the forest ecosystem. In Finland, for example, the temperature increase is expected to be 4 °C in summer and more than 6 °C in winter by the end of this century (Ruosteenoja and Jylhä 2007), which is likely to increase the carbon uptake but also the decay of organic matter in forest soil. These changes in climatic conditions may reduce part of the benefit provided by stumps in carbon storage in forest ecosystems due to the enhanced decay rate of stump wood .

9.4 Concluding Remarks—Advantages and Disadvantages of using Stumps

Stump harvesting has several advantages in biomass supply for energy use, but there are also many disadvantages reducing the benefits of using stump wood in energy production. The advantage, above all, is that the stumps and coarse roots provide large amounts of forest biomass to substitute and impede carbon emissions from fossil energy. The biomass in stumps is energy-rich, providing the energy of 130–250 MWh ha^{-1} of forest land (Hakkila 2004; Eriksson and Gustavsson 2008) depending on the properties of forests. In harvesting stumps and coarse roots the soil surface is disturbed, which enhances the establishment of seedlings and accelerates regeneration (UPM Kymmene 2003, Saarinen 2006). On the other hand, this allows faster weed and scrub growth on the site, which may increase management costs of seedling stands in tending. However, removal of stumps may reduce the damage to seedlings due to pine weevils common in the Nordic countries. Furthermore, the harvesting of stumps and coarse roots is an efficient way to reduce the risks of root rot and thus decrease the risks of reduced growth and damage to stem wood. These benefits are partly undermined by the changes in the structure and dynamics of the forest ecosystem. For example, the extraction of stumps is likely to decrease the biodiversity due to the reduction of decaying wood available for many rare and endangered invertebrates. Furthermore, stumps constitute the largest proportion of deadwood, which is a vital habitat for a variety of fungi, lichens and mosses.

In the Nordic countries, harvesting of stump biomass is recommended mainly in clear-cut areas. This reduces the possible damage to the soil and excludes the risk of damage to remaining trees, which is possible in thinning operations. Nevertheless, the disturbance of the soil may lead to increased mineralization and leaching of minerals along with a decreasing amount of carbon in the terrestrial carbon sink. On the other hand, the content of nutrients in stump wood are relatively much smaller than in foliage and branches, which means the effects of stump removal on soil nutrients are not as great as might be expected. Therefore, it is possible that stump harvesting may affect only marginally the tree growth and carbon sequestration in the rotation following the stump harvesting, as reported by Alam et al. (2012), but contradictory findings are also reported (Jacobson et al. 2000; Walmsley et al. 2009; Mason et al. 2012). Especially, the whole-tree harvest including foliage may reduce the growth of conifers in the next rotation on poor sites (Mann et al. 1988; Proe et al. 1996; Jacobson et al. 2000; Walmsley et al. 2009; Mason et al. 2012).

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Chapter 10
Life Cycle Carbon Assessment of Bioenergy Production

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Abstract The carbon neutrality of forest biomass in energy production is questioned due to high indirect greenhouse gas emissions (mainly CO_2) consequent to the land use and the changes in land use. This is why the effects of forest management on the carbon flows of ecosystem should be linked with the carbon flows due to the use of forest biomass in energy production. In this context, the dynamic life cycle assessment (LCA) is addressed regarding how the atmospheric impacts of energy production based on forest biomass are linked to the forest management used to control the sink/source dynamics in the forest ecosystem. Such integration makes it possible to identify the management strategies for using forest-based biomass in climate change mitigation.

Keywords Attributional · Biomass · Carbon dioxide · Climate impact · Consequential · Ecosystem · Life cycle assessment · Life cycle inventory · Mitigation · Net CO_2 exchange · Regional level · Stand level · Time dynamics

10.1 Forest Biomass in Producing Energy and Mitigating Climate Change

Global warming and growing demands for energy have triggered a wide interest in bioenergy in order to increase energy sources and mitigate climate change. In the Nordic countries, especially in Finland and Sweden, bioenergy is mainly based on forest biomass, which is harvested directly from the forests and utilized as wood chips or in the form of black liquor originating from pulp production and processing waste of sawmilling. The general rationale behind the utilization of forest-based bioenergy is that the use of biomass produced in forest ecosystems will not increase the greenhouse gas (GHG) emissions as much as fossil fuels. This is especially the case for carbon dioxide (CO_2), which is emitted in the combustion of biomass but sequestered in ecosystems in the growth of trees of successive generations.

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The carbon neutrality of renewable biomass including forest biomass has recently been questioned due to high indirect GHG emissions, which are due to the use of land for producing energy biomass (Searchinger et al. 2008; Melillo et al. 2009; Schulze et al. 2012). The impacts of increased use of biomass may lead to long-term losses of carbon from ecosystems that are not reasonable with respect to the mitigation of climate change. The carbon debt (Gibbs et al. 2008; Pingoud et al. 2012) induced by the production of bioenergy can be compensated over time by the future forest growth. In the Nordic countries, a time perspective to cover the debt in re-growth is long due to slow tree growth, as indicated by long rotation periods needed in forestry. The importance and time dependence of indirect carbon emissions have been emphasized in recent studies where emissions were studied based on carbon flux and stock changes in forests (Melin et al. 2010; Repo et al. 2011; Sathre and Gustavsson 2011; Kilpeläinen et al. 2012; Pingoud et al. 2012).

The use of forest biomass in substituting fossil fuels implies that the boundaries of energy system should also cover the potential of forests to produce energy biomass and sequester carbon into the ecosystem. This emphasizes the possibilities to control the accumulation and release of carbon into and from the stocks in the forest ecosystems (Gustavsson and Sathre 2011; Kilpeläinen et al. 2012, 2013). Furthermore, the boreal forest ecosystems may benefit from the global rise of temperature and carbon dioxide, which increases the carbon uptake in trees and the consequent carbon sequestration into the ecosystem (e.g. Melillo et al. 1993; Kellomäki et al. 2008). However, the model simulations show that climate change may increase the frequency and length of drought episodes even in boreal conditions, thus reducing the potential to sequestrate and store carbon in the ecosystem and to produce energy biomass and timber (Kellomäki et al. 2008; Alam et al. 2010; Shurpali et al. 2010). These issues emphasize the need for systematic approaches in assessing atmospheric impacts of utilization of biomass in climate change mitigation.

10.2 Use of Life Cycle Assessment (LCA) for Forest Production

10.2.1 Life Cycle Assessment

Environmental life cycle assessment (LCA) can be used to analyze the environmental impacts of producing energy biomass in land use. LCA refers to a technique to assess environmental impacts associated with all the stages of a product life “from cradle to grave” (ISO 2006). In forestry, the focus in LCA is mainly on the techno-system emissions, including the management, harvesting and logistic operations over the biomass production and utilization chain. Until now, the production phase (forest growing) has only roughly been included in LCA or even left outside the system boundaries due to the assumption that the production and utilization of biomass is carbon-neutral.

Following the basic principles of LCA, all the operations that contribute to the life cycle of the product, production process or activity fall within the system boundaries

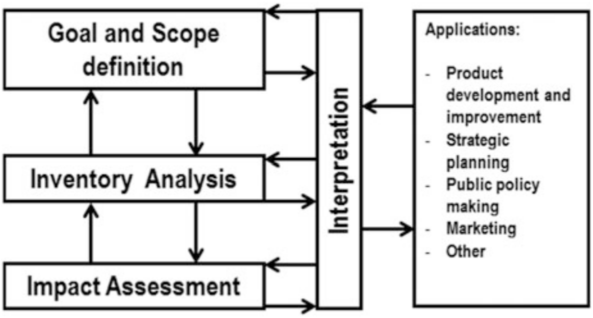


Fig. 10.1 Schematic representation of the phases of LCA. (ISO 2006)

and should be included in the analysis (ISO 2006). In LCA, the goal definition and setting the scope of assessment are the most important components; i.e. they define the purpose of assessment, and the system boundaries and functional unit used in the assessment (Fig. 10.1). In general, the functional unit provides a commensurate unit to which the inventory data is compared or normalized. In forestry studies, the common functional unit is the mass of product under study. Furthermore, other functional units such as energy and the economic value of a product and land area can be used. In case of forest-based bioenergy, the energy unit is a reasonable choice, because the energy content of wood per volume is lower compared to coal, for example.

10.2.2 Life Cycle Inventory

In applying LCA in the forest-based energy production, the production phase has a key role. If it is excluded, it is impossible to include in LCA the indirect impacts of energy biomass induced by the forest production phase (forestry). This is especially the case if the atmospheric impacts are estimated by using the Life Cycle Inventory (LCI). This refers to the inventory of flows including inputs of water, energy, and raw materials to the production system, and the releases to air, land, and water outside the production system (ISO 2006; Finnveden et al. 2009). Regarding the production of energy biomass in forestry, the key flows are CO_2 in growth and CO_2 emissions from soil, management, logistics and manufacturing and using fuels, which control storage and release of carbon in the energy production system. Such a holistic approach is useful also, if the multiple objectives are set for forests and there are possible trade-offs between different objectives (e.g. in carbon sequestration and bioenergy production). This enables us to highlight the parts of the life cycle of forest-based energy where the greatest improvements can be made with respect to atmospheric impacts and forest management.

10.2.3 Life Cycle Impact Assessment

Generally, the impact assessment in LCA consists of classification, characterization, normalization and valuation (weighting). Classification is the process of assignment and initial aggregation of LCI data into impact groups. Characterization is the assessment of the magnitude of impacts of each inventory flow (e.g. impacts of CO_2 on global warming), and it enables us to compare the LCI results within each impact category. Usually, the inventory results are assigned to different impact categories, based on the impacts on the atmosphere and the environment. Impact categories include global effects (e.g. global warming and resource depletion), regional effects (e.g. acidification, eutrophication) and local effects (e.g. toxicity and land use). When the impacts are normalized, the comparisons between various selected situations are enabled. Valuation is the assessment of the relative importance of environmental burdens identified in the various stages of impact assessment. Based on the LCI, the Life Cycle Impact Assessment (LCIA) can be done (ISO 2006), if one aims to evaluate all the environmental impacts (Fig. 10.1).

For the impact assessment, the estimation of net atmospheric impacts of forest bioenergy needs to be integrated with the dynamic carbon sinks and sources covering the whole production and utilization chain. These calculations should also include the simultaneous effects of forest management and the changes in environmental conditions (e.g. temperature, precipitation) affecting the ecosystem dynamics. In assessing the atmospheric impacts of forest-based energy, it is of prime importance to consider the potentials of the forest ecosystem (1) to produce energy biomass, (2) to substitute fossil fuels, and (3) to produce biomass for wood-based products and other ecosystem services (e.g. timber, carbon sequestration). Such a holistic assessment helps in setting the strategies to use forests to reduce CO_2 emissions and substitute fossil fuels and/or increase sequestration of carbon from the atmosphere (Fig. 10.1, 10.2). This implies that the changes in carbon fluxes (i.e. carbon uptake in growth and emissions in decomposition of organic matter) should be included in the LCI for interpreting the inventory results in the LCIA as regards the use of forest biomass in producing bioenergy. The same factors allow us to consider properly the temporal and spatial dimensions in producing energy biomass and substituting fossil fuels in order to mitigate climate change. Temporal and spatial variability indicates how the objectives to use forests in mitigating climate change can be achieved at the regional/landscape level; i.e. how forests composed of separate stands of varying age and structural properties (e.g. tree species composition, stocking) take up and retain CO_2 in the ecosystem and what are the net climate impacts of the use of forest biomass in producing bioenergy.

10.3 Estimation of Net CO_2 Exchange of Forest Production and Biomass Utilization

In the following, the carbon dioxide fluxes in the bioenergy production system assessed by means of LCA. The analysis includes both the ecosystem producing energy biomass and the technosystem used to convert biomass to energy, thus covering the

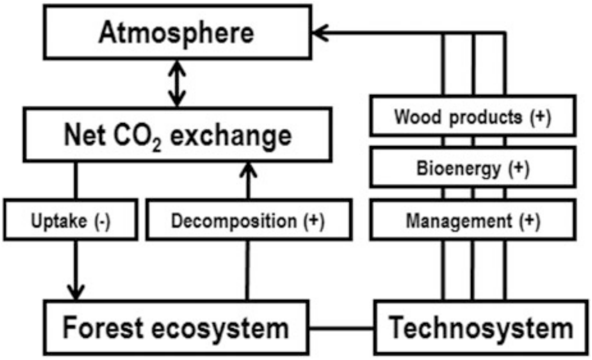


Fig. 10.2 Schematic representations of the links between ecosystem and technosystem CO_2 flows in calculating net CO_2 exchange for the production of forest biomass and the use of biomass for bioenergy

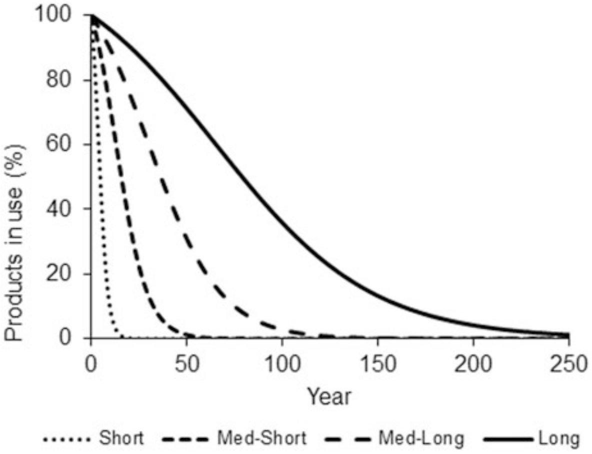
whole production chain "from cradle to grave". In this context, the net CO_2 exchange of the whole forest bioenergy production and utilization system includes an annual sum of the ecosystem and technosystem fluxes (net emissions or sequestration of carbon) at stand or regional level (Fig. 10.2).

The processes of carbon accumulation and release in the forest ecosystem (net ecosystem CO_2 exchange, NEE) are either annual or singular (thinning, final felling). NEE consists of photosynthesis, autotrophic respiration and heterotrophic respiration (for further details, see Chapter 8 in this volume). Eddy covariance technology is commonly used to calculate the values of NEE. However, the time series of measurements are rather short, and the study sites are never, or only seldom, established with the production of energy biomass in mind, with the exception of measurement for agricultural energy crops or peat mining for energy production (e.g. Seppälä et al. 2010; Shurpali et al. 2010; Grönroos et al. 2013). Furthermore, the missing reference situation reduces the applicability of eddy covariance measurements in the LCI and LCIA, especially in long-term analyses. Therefore, modelling by integrating the CO_2 dynamics in the ecosystem with LCA seems to be the only way to assess the atmospheric impacts of bioenergy production and utilization (Cherubini et al. 2012; Kilpeläinen et al. 2011; Helin et al. 2012; Kilpeläinen et al. 2012).

Kilpeläinen et al. (2011) combined a gap-type forest growth model with forest inventory data for using LCA in assessing the atmospheric impacts of a forest-based energy system both at the stand and regional scales. The approach integrated the traditional forestry measures (i.e. growth and timber/biomass yield) along with the consequent carbon fluxes into the forest ecosystem and from the forest to the atmosphere (Kilpeläinen et al. 2011; Alam et al. 2013; Kilpeläinen et al. 2012; Kilpeläinen et al. 2013). The calculation system includes further the CO_2 emissions from technosystem (i.e. emissions from machinery used in management and logistics). In this context, the net CO_2 exchange (g CO_2 $m^{-2}a^{-1}$) is a sum of CO_2 fluxes covering both ecosystem and technosystem following Eq. 10.1:

$$C_{net} = C_{seq} + C_{decomp} + C_{harv} + C_{man} \tag{10.1}$$

Fig. 10.3 Wood products turnovers in various categories



In Eq. 10.1, the carbon flux into the forest ecosystem (growth; above-ground and below-ground) has negative values, whereas the fluxes to the atmosphere have positive values following the practice used in eddy covariance measurements. In Eq. 10.1, the net CO_2 exchange (C_{net}) is the sum of carbon uptake (C_{seq}) in growth, the carbon emissions from decaying litter and humus (C_{decomp}) and from the techno-system, including emissions from the degradation of wood-based materials manufactured from timber and the instant emissions from combustion of energy biomass after harvesting (C_{harv}), and emissions from the management chain (C_{man}). The C_{harv} can also include the carbon emissions from the combustion of waste from processing saw logs and/or pulp.

The CO_2 emissions from degrading wood-based materials represent abandoned material no longer in use. In calculations, the materials are divided into different groups based on their life span. In doing this, timber (pulpwood, saw logs) was converted into wood-based products. The share of different products existing in the technosystem stock at a certain time point was calculated following Eq. 10.2 (Karjalainen et al. 1994):

$$PU = D - \frac{a}{1 + be^{-ct}}, \tag{10.2}$$

where PU is the proportion (0 … 100) of products in use, and a, b, d are fixed parameters (120, 5, 120, respectively). *T* is time (a) and *c* (a^{-1}) is 0.5; 0.15; 0.065 and 0.03 for short, medium-short, medium-long and long lifespan of a product, respectively (Fig. 10.3). In general, pulpwood represents the items with a short or medium-short lifespan and sawlogs represents the items with a medium-long or long lifespan. The carbon released from the products that are no longer in use is completely converted into CO_2.

10.4 Net Atmospheric Impacts of Bioenergy Production and Utilization

10.4.1 Attributional LCA

In forest-based production, the net ecosystem CO_2 exchange is allocated to various biomass products utilizing their share from total production over the rotation period, based on the attributional LCA. This approach assesses the carbon flows over the whole production and utilization chain and they are allocated to each product, including both the ecosystem and technosystem over the whole rotation period of biomass production. Regarding forest-based bioenergy, the carbon in energy biomass is released immediately, representing an instant emission when combusted. In the integrated production of timber and energy biomass, the largest emissions exist at the end of the rotation. This is why the timing of allocation represents a strong producer's point of view. It answers the question: what is the most atmospherically efficient way to produce biomass for energy over one rotation? (Routa et al. 2011a, 2011b; Alam et al. 2012, 2013). The approach is close to the carbon footprint calculation, where the emissions from the technosystem (e.g. chipping, transportation) and biomass utilization are related to the whole production chain.

10.4.2 Consequential LCA

Net CO_2 exchange is also used in calculating the payback time of immediate CO_2 emissions using the consequential LCA (e.g. Repo et al. 2011; Helin et al. 2012; Pingoud et al. 2012). The consequential LCA refers to the assessments of how the CO_2 fluxes in the ecosystem and technosystem will change in response to any actions used to control fluxes. Regarding the forest management and forest-based bioenergy, this approach has a strong focus on the decision whether to produce energy biomass from existing biomass resources or not. With respect to atmospheric impacts, at the beginning of the study period there is a resource of energy biomass, which may be utilized or not. In this situation the energy biomass potential is often at the maximum at the beginning of the study period. Thereafter, there is a difference in the CO_2 emission values, depending on whether the logging residues are left to decay or combusted. This difference is utilized further in assessing the substitution impact of energy biomass utilization. Consequently, it is used to assess which way is preferable in mitigating CO_2 emissions in the context of forests and forest-based bioenergy.

The consequential LCA of carbon should emphasize the temporal and spatial dependencies of carbon sequestration, bioenergy potential and wood utilization in bioenergy use and its atmospheric impacts (Helin et al. 2012). In fact, the forest-based bioenergy system includes forest areas up to the national level composed of a large number of tree stands providing energy biomass when managed for timber alone or in an integrative way for timber and energy biomass. In this case, the energy biomass potential is dependent on the age structure and tree species composition, both of which change over time depending on the growth, management and utilization. On a large spatial scale, the forest-based bioenergy system is a dynamic one, which has strong effects on the possibilities for producing energy and the consequent atmospheric effects. In applying the consequential LCA, a major problem is the reference situation to which one compares the use of forest biomass in producing bioenergy. This is because the optimal production option for timber is probably different than that for producing energy biomass. The selection of reference situation affects the timing of emissions and the future growth and development of tree stands with regard to the carbon sequestration and biomass yield. This further affects the estimation of atmospheric impacts of forest bioenergy production and its substitution capacity over a given time period.

10.5 Case Studies

10.5.1 Case 1—Stand Level Analysis

Figure 10.4 shows the net CO_2 exchange over an 80-year rotation in a Norway spruce stand, which was managed for producing both timber and biomass for bioenergy (i.e. energywood harvest and harvest of logging residues; needles and branches). In the timber production regime, the net CO_2 exchange was close to the net ecosystem CO_2 exchange. The only difference is the emissions due to management and timber harvest (i.e. emissions from machinery). The production of energy biomass is not assumed to affect timber production and thus carbon stored in and released from wood products is not shown in Fig. 10.4. In producing bioenergy, the net CO_2 exchange includes the CO_2 emissions from the management and harvest, and the combustion of energy biomass. The effect of combustion on the net CO_2 exchange in the biomass production system is indicated by the black bars in Fig. 10.4.

Generally, the emissions from combusted biomass are related to the amount of harvested timber, which in turn depends, among other things, on tree species, the productivity of the site and the amount of energy biomass logged in management and harvest. For example, stumps can be left in the cutting area, which decreases the payback time of carbon released from the ecosystem. The timing of thinnings may also change if energy biomass is preferred in management compared to that which is optimal in timber production. On the other hand, any management enhancing the forest growth will also enhance the net CO_2 exchange in the forest production, with the consequent increase of CO_2 emissions in producing bioenergy (Routa et al. 2011a; Routa et al. 2012; Pyörälä et al. 2013; unpublished). In the efficient production of energy biomass, the rotation length is probably different from that optimized for timber production. This further affects the time period, which should be considered in estimating the atmospheric impacts of energy biomass (Pyörälä et al. 2013; unpublished).

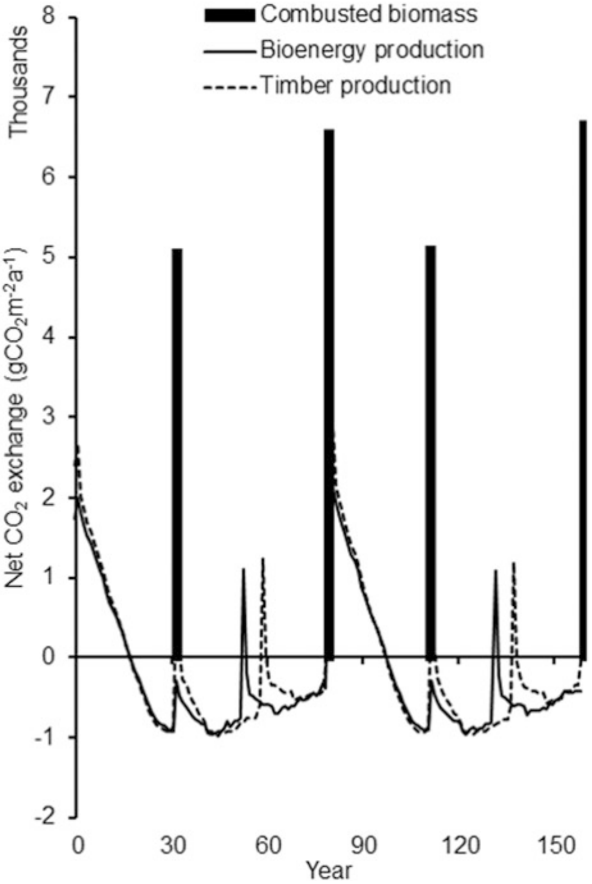


Fig. 10.4 Annual net CO_2 exchange (C_{net}) studied over two 80-year rotation periods both for timber and bioenergy production in Norway spruce. Simulated results for OMT (*Oxalis myrtillus*) site type in central Finland (Joensuu region: 62°39' N, 29°37' E). Energy biomass included small-sized trees (i.e. whole trees) in energywood harvesting and logging residues (i.e. needles and branches) in final felling

10.5.2 Case 2—Regional Level Analysis

In regional level analyses, the atmospheric impacts of bioenergy production include the CO_2 emissions from any stand managed and harvested for producing energy biomass. On the other hand, each stand in the region is growing and developing in its own way depending on the fertility of the site, tree species, and the age and structure of tree stand (e.g. spacing of trees). In such a case, the atmospheric impacts of bioenergy production (i.e., the difference between the net CO_2 exchanges of the production and utilization of energy biomass and timber and only timber in forestry) vary over time, as demonstrated in Fig. 10.5. This difference can be further utilized in calculations of the net climate impacts of bioenergy utilization in substituting fossil fuels. In Fig. 10.5, the energy biomass was harvested in the thinning of stands in early developmental phases (energywood thinning) and in the clear cut of mature stands (logging residues). The atmospheric impacts were assessed by using the bioenergy potentials in the calculations. By using the potentials, one avoids the difficulties in estimating the fluctuations in timber cuttings due to variation in timber prices.

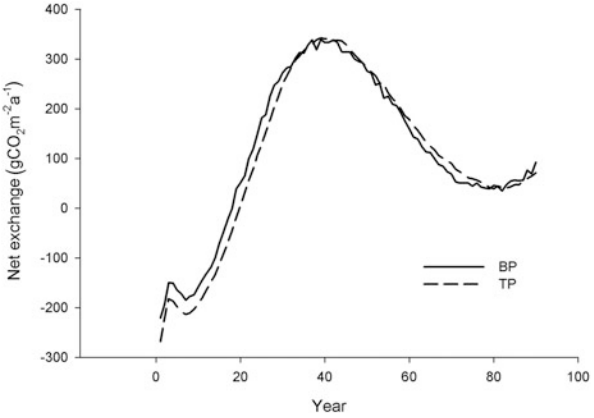


Fig. 10.5 Annual net CO_2 exchange (C_{net}) of bioenergy production (*BP*) and timber (*TP*) production over Finland (Kilpeläinen et al. 2012). Simulated results for the sample plots of National Forest Inventory (*NFI*) over the whole of Finland. TP included harvesting of timber (i.e. saw logs and pulpwood) in thinnings and final fellings. In addition to timber, BP included harvesting of energy biomass of small-sized trees (i.e. whole trees) in energywood thinnings and logging residues (i.e. needles, branches, top parts of the stem, stumps and roots) in final fellings. Copyright (2012), used with permission from John Wiley & Sons

During the first 40 years, the CO_2 emissions into the atmosphere increased due to the increase in the use of timber and energy biomass facilitated by the increasing potential of timber and energy biomass (Fig. 10.5). This was due to the maturing of trees and the shift of age structure from the balance between young and mature stands to the dominance of mature stands. Thereafter, the potentials declined due to harvesting of mature stands and the shift of age structure, where the young stands and mature stands are more balanced than at the end of the first 40-year period. It is evident that the net CO_2 exchange from a forest region fluctuates over longer periods following the dynamics of age structure controlled by the management and harvest cycle defined by the length of rotation and species-specific management both in bioenergy and timber production regimes. This also affects the availability of energy biomass as a substitute for fossil fuels.

10.6 Time Dynamics of LCA in Forest Production

The dynamic development of a forest ecosystem is challenging from the LCA point of view. After the clear cut before planting, the forest ecosystem is a source of CO_2 due to the decomposing logging residue and soil organic matter originating from the previous production cycle (Fig. 10.6). This is the case until the uptake of CO_2 in growing seedling exceeds the CO_2 emissions from decay litter and humus (about 20 years from establishment). Thereafter, the forest ecosystem is a strong sink of CO_2 until the culmination of annual growth at 40–60 years, depending on the species and site fertility. The further maturation of trees reduces the capacity of the ecosystem

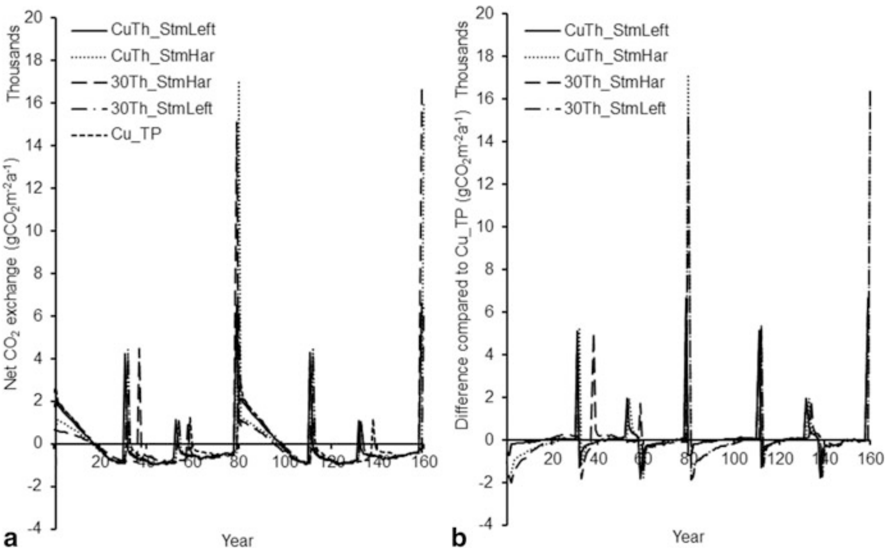


Fig. 10.6 Net CO_2 exchange (C_{net}) (**a**) and their differences (**b**) for four various bioenergy production and utilization regimes compared to the traditional timber production regime. The simulations were extended over two rotations (2 × rotation of 80 years) in order to demonstrate how the management regimes, especially the stump harvest, may affect the net CO_2 exchange over time in integrating the ecosystem and technosystem with each other. *CuTh_StmLeft* = bioenergy harvesting, current basal area thinning thresholds, stumps left on site; *CuTh_StmHar* = bioenergy harvesting, current basal area thinning thresholds, stumps harvested; *30Th_StmHar* = bioenergy harvesting, 30 % increased thinning basal area thinning thresholds, stumps harvested; *30Th_StmLeft* = bioenergy harvesting, 30 % increased thinning basal area thinning thresholds, stumps left on site, *Cu_TP* = traditional timber harvesting, no bioenergy harvested

to sequester CO_2 but the amount of carbon in trees and soil is still high, even though the storage rate declines along with the reducing growth. Thus, the timing of the maximum CO_2 uptake and carbon storages is different. Regardless of the intensity of thinning, the harvest and combustion of stumps will increase the CO_2 emission substantially, which in this case remained smaller than in excluding the harvesting of stumps (Fig. 10.6). However, the carbon in stumps remaining in the cutting area will be emitted gradually. Therefore, the assessment of net atmospheric impacts of forest-based bioenergy is to be extended over the whole production cycle (rotation) and/or assessed in alternative spatial scales in order to avoid the biases due to the temporal variability in the CO_2 uptake and emissions in the forest ecosystems.

10.7 Summary and Conclusions

The carbon neutrality of forest biomass in energy production is questioned due to high indirect greenhouse gas emissions consequent to the land use and the changes in land use. This is why the effects of forest management on the carbon flows of an ecosystem should be linked with the carbon flows due to the use of forest biomass in energy production. These links may be built using the dynamic life cycle assessment (LCA), where the atmospheric impacts of energy biomass production are linked with the forest management controling sink/source dynamics in the forest ecosystem. Such integration makes it possible to define the management strategies for using energy biomass in fossil fuel substitution and in the mitigation of climate change.

The time dependency of the emissions affects the atmospheric impact calculation for the forest biomass used in energy production. In this regard, the attributional LCAs can be utilized in assessing the carbon footprint for the production of forest-based bioenergy over the whole production chain, which includes the CO_2 uptake in forest growth and the CO_2 emissions in management, logistics and combustion of biomass in energy production. Similarly, the consequential LCA can be used to compare the bioenergy production and utilization, and fossil fuel substitution to the situation where bioenergy is not utilized. The reference situation and substituted fuel affects the interpretation of how bioenergy affects the short- and long-term mitigation of climate change. It is still open how best to set the system boundaries to calculate the net climate impacts of production and utilization of energy biomass. Furthermore, is it reasonable to utilize the overall sink of the forest to compensate for emissions from the utilization of bioenergy, and how the carbon credits are allocated to bioenergy and to other wood-based products in such a situation?

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Chapter 11
Time Dynamics and Radiative Forcing of Forest Bioenergy Systems

Roger Sathre, Leif Gustavsson and Sylvia Haus

Abstract In this chapter we explore the temporal dynamics of using forest bioenergy to mitigate climate change. We consider such issues as: growth dynamics of forests under different management regimes; the substitution effects of different bioenergy and biomaterial uses; temporary carbon storage in harvested biomass; the availability of different biomass fractions at different points of a wood product life cycle; and changes in carbon content of forest soils. We introduce the metric of radiative forcing, which quantifies the accumulating energy due to the global greenhouse effect, and we describe a method to estimate quantitatively and to compare the cumulative radiative forcing (CRF) of forest bioenergy systems and reference fossil energy systems. In three case studies, we describe the time dynamics and estimate the CRF profiles of various forest biomass systems.

Keywords Time dynamics · Cumulative radiative forcing · Forest bioenergy · Landscape scale · Carbon balance · Greenhouse effect · Mitigation · Forest fertilization · Residues · Substitution

11.1 Forest Sector in Mitigating Climate Change

The forest sector plays an increasing role in climate change mitigation in some countries, with the prospect of sustainably providing essential materials and services as part of a low-carbon economy. In contrast to fossil energy systems which deplete on a geologic time scale, forest bioenergy systems are cyclically renewable through ecological processes. The dynamic nature of forest systems is complex, and should be understood and incorporated into our modeling, policymaking, and management

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frameworks. There are multiple climatic aspects of biomass, such as fossil fuel substitution (Schlamadinger et al. 1997), material substitution (Gustavsson and Sathre 2011), and carbon stocks and flows within living biomass, wood products and soils (Eriksson et al. 2007).

The carbon flows of forest biofuels from a life-cycle perspective include the phases of growth, processing, utilisation, and return. In the growth phase, carbon dioxide (CO_2) is removed from the atmosphere through photosynthesis, and held as carbon-based compounds in tree tissues. The chemical bonds of these reactions result in accumulated solar energy. In the processing phase, tree biomass is harvested, transported, and refined to give it desired characteristics. The energy sources used for these processes may be fossil fuels. Carbon is temporarily stored in biomass fibre, until the biofuel is finally combusted. Upon combustion, the accumulated energy is released and used, and the carbon is ultimately released back to the atmosphere. From a wider perspective, this energy service substitutes for other energy sources, likely fossil fuels.

There are several approaches to quantifying dynamic carbon flows over time. Flow-based approaches seek to quantify the carbon mass balance, where input minus output minus stock change is constant, over a given time horizon, e.g., annual or rotational. A common method of analysing mitigation options is the greenhouse gas (GHG) balance approach, where all GHG emissions that occur during a given time period are simply summed up regardless of when they occur. Cumulative net emissions are determined, beginning from zero emission in the baseline year. A system with lower cumulative emissions at the end of the time period is considered to have less climate impact than a system with higher emissions. Non-CO_2 GHG emissions (e.g., CH_4 and N_2O) are typically converted to “CO_2 equivalents” (CO_2e) based on the Global Warming Potential (GWP) of the gases, which express the relative climate impact of the gases compared to an equal mass of CO_2 over a fixed time horizon, typically 20, 100, or 500 years (IPCC 2007).

This approach does not fully describe the complexity of the system, however, because the sum is static and does not take into account the temporal patterns of the GHG emissions and the resulting dynamics of atmospheric radiative imbalance. The heterogeneity of stocks and flows over time reduces the effectiveness of static methods of analysing and comparing the mitigation effectiveness of forest systems. Within any finite time period, the accumulated radiative imbalance, and hence the climate impact, depends not only on how much GHG is emitted but also on when it is emitted. Cumulative radiative forcing (CRF), also called integrated radiative forcing or absolute global warming potential, is a metric that more accurately estimates the time-dependent climate impacts of dynamic systems, but is a more complex analytical procedure than GHG balance calculation and requires information on time profiles of emissions and removals (Fuglestvedt et al. 2003).

Several authors have proposed indicators based largely on CRF. For example, Kirkinen et al. (2008) introduced a “relative radiative forcing commitment” defined as the CRF caused by using a fuel divided by the combustion energy of the fuel. O’Hare et al. (2009) proposed a “physical fuel warming potential” defined as the CRF caused by using a fuel, relative to the CRF of using a reference fuel. Kendall et al. (2009) introduced a “time correction factor” to be applied to CO_2 emissions that occur at the beginning of a defined time horizon but are amortised over the entire horizon, such that the CRF is equivalent for both. Levasseur et al. (2010) proposed

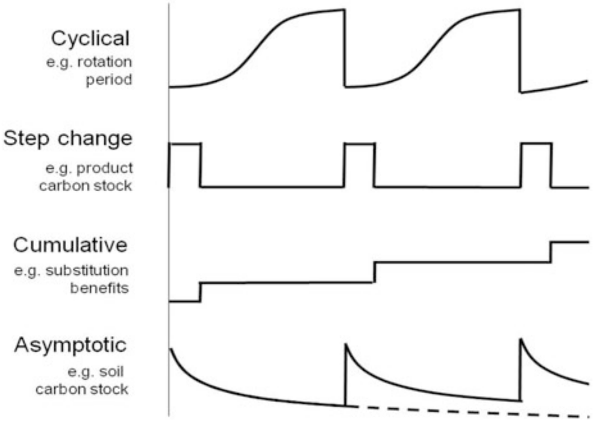


Fig. 11.1 Illustrative examples of temporal patterns of carbon stocks and flows associated with forest bioenergy systems. Positive values show atmospheric net carbon reduction. Vertical axes are not to scale

the use of "dynamic characterisation factors" for the global warming impact category of life cycle assessment, based on the CRF of different GHGs over different time horizons. Cherubini et al. (2011) calculated a "GWP$_{bio}$" defined as the CRF resulting from a unit of bioenergy divided by the CRF resulting from an equivalent amount of fossil energy. The GWP$_{bio}$ notion was expanded by Pingoud et al. (2012) to include substitution effects of wood-based products. As currently applied, however, the GWP$_{bio}$ indicator gives simplified approximations of climate impacts, but does not incorporate comprehensive life-cycle emissions modeling of specific forest stands and landscapes. Bergman (2012) developed a metric called "time-zero equivalent", based on CRF, to consider the effect of timing of GHG emissions on the climate change impacts of building products.

Robust estimates of short- and long-term impacts resulting from GHG emissions are needed for various products, fuels, and management practices. For example, forest growth modeling under simulated environmental conditions may seek to identify changes in future growth patterns (Poudel et al. 2011). In this chapter we explore the temporal dynamics of using forest bioenergy to mitigate climate change. We focus on the metric of radiative forcing, which quantifies the accumulating energy due to the global greenhouse effect, and we describe a method to estimate quantitatively and compare the CRF of forest bioenergy systems and reference fossil energy systems. In three case studies, we describe and compare the time dynamics and estimate the time profiles of CRF of various biomass uses. Although the emphasis of this chapter is on forest biofuels, many issues are equally relevant to agricultural biofuels.

11.2 Time Dynamics of Forest Systems

Time dynamics of GHG stocks and flows associated with forestry and wood product use exhibit a variety of temporal patterns (Fig. 11.1). Cyclical patterns repeatedly accumulate and release carbon stocks, such as a forest rotation cycle. Step changes involve a discrete increase or decrease in carbon stock in a given pool, for example

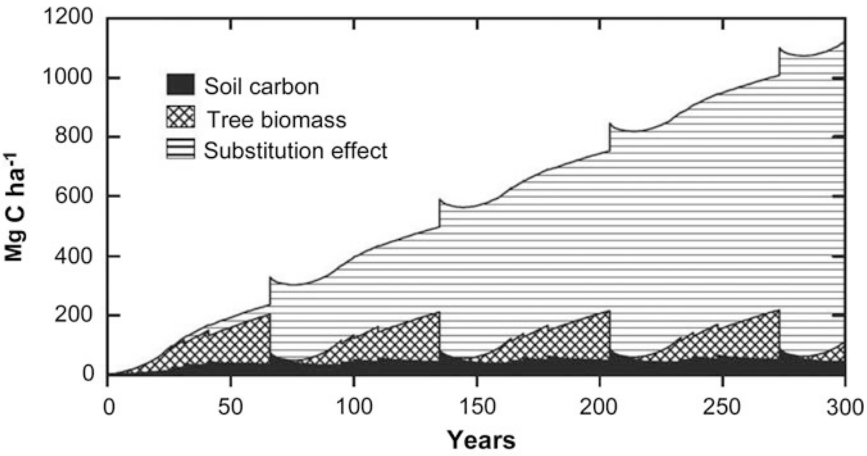


Fig. 11.2 Atmospheric net carbon reduction development over time of a managed forest stand (MgC ha^{-1}), including carbon stocks in tree biomass and forest soil, and cumulative substitution benefits. (Source: Eriksson et al. 2007)

the beginning and end of the life span of a wood product. Cumulative changes are flows that accumulate over time, such as fossil emissions avoided due to ongoing biomass substitution. Asymptotic changes are rapid at first and become slower over time, for example a decrease in soil carbon stock due to decay. Different forest management actions and wood product uses will result in different combinations of GHG emissions and removals over time. The climate effects of different biomass uses may entail material substitution (displacement factors for material use; Sathre and O'Connor 2010), fuel substitution (displacement factors for different fossil fuels; Gustavsson et al. 1995), and multiple or cascaded use (material use followed by energy recovery; Sathre and Gustavsson 2006).

Dynamic analysis of the climate implications of forest bioenergy systems may be conducted across various spatial scales, from the stand level to the landscape level. At the stand level, a unit ha of forest land may be followed throughout multiple rotation periods to generate understanding of the time-dependent changes in stocks and flows in biomass, wood products, litter and soil under various management and use scenarios. The overall climate significance of this stand-level forest management may be put in a larger context by landscape-level analysis incorporating many stands within a region or a country. Carbon dynamics differ substantially as the scale increases from the forest stand level to the landscape level. Within a managed forest stand a characteristic curve can be traced over time: carbon is bound in tree biomass during stand establishment and growth, then eventually accumulates at a decreasing rate, then is removed during harvesting, followed by establishment of the subsequent rotation. With each harvest, the benefits of biomass substitution benefits accumulate, and changes in soil carbon stock depend on relative rates of litter inputs and organic matter decay (Fig. 11.2).

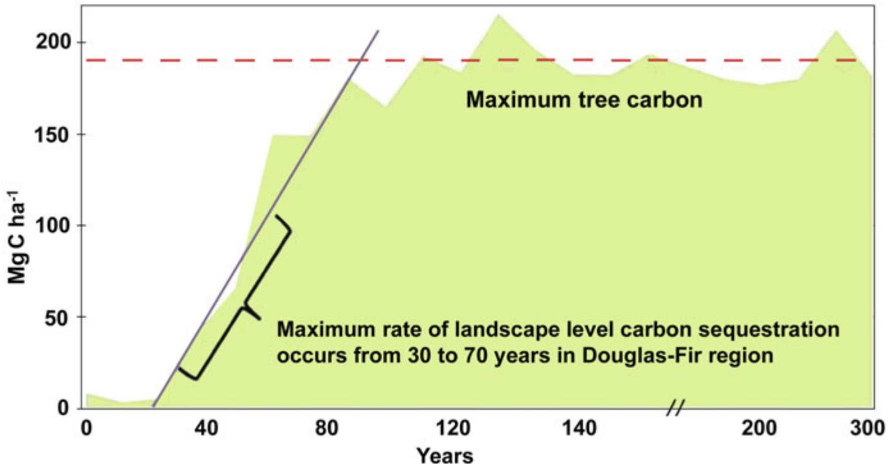


Fig. 11.3 Forest carbon accumulation in Washington State, USA (MgC ha^{-1}). Little net growth occurs beyond the age of 100 years. (Source: Lippke et al. 2011, based on data from US Forest Service Forest Inventory (Bolsinger et al. 1997))

At the landscape level in a managed forest, the total carbon balance at any time is the aggregate of the balances of a multitude of stands. The total carbon stock in living biomass at the landscape level tends to remain fairly stable over time in a healthy forest ecosystem, as the harvesting of some trees during a given time period is compensated by other trees growing during the same period. The maximum carbon stock at the landscape level is lower than the maximum at the stand level, because not all the individual stands will hold the maximum stock at the same time (Kurz et al. 1998). If forests are managed appropriately, the average carbon stock in forest biomass can increase over time, while simultaneously producing a flow of harvested biomass out of the forest that gives continually increasing carbon benefits due to fuel and material substitution (Pingoud et al. 2010).

Understanding of forest biomass growth dynamics is integral to robust system analysis. Managed forestry typically involves cyclical growth curves or rotation periods, with the magnitude and frequency of cycles depending in part on climate and management regimes. If a mature forest stand is not harvested, the growth rate decreases and the carbon stock levels off (Fig. 11.3). Eventually, non-harvested forest stands are subject to natural disturbance regimes such as fire or insect attacks that result in release of stored carbon (Kurz and Apps 1999), damning prospects for permanent carbon storage in living trees. In addition to dynamics of living tree biomass, corresponding patterns of carbon stock change can be identified in wood-based products, dead and decaying biomass, and forest soils.

The significance of temporary carbon storage in harvested biomass varies with the life span of the product. The short time (weeks or months) between harvest and typical use of biofuels is less significant than the longer lifespan (decades or centuries) of wood construction material used as bioenergy after building demolition. Long-term carbon storage occurs in long-lived harvested forest products, which may be used as bioenergy at the end of their service lives. Atmospheric CO_2 concentration is affected by *changes* in the stock of carbon in forest products, not by the magnitude of the stock. The stock of forest products will stabilize if the rate of wood entering the products reservoir is equal to the rate of wood leaving the reservoir. Over time an increase in carbon stock in forest products could occur as a result of general economic implications such as economic growth, whereby more products of all kinds are produced and possessed, or through a societal transition from non-wood to wood-based products.

An integrated life cycle approach using cascaded biomaterial and bioenergy becomes more critical if biomass production rates become limited in relation to demands (Sathre and Gustavsson 2006). The limiting factor of the system can be the available biomass resource for energy or the land available for biomass production. This limit implies that use of biomass in one application will reduce the amount available for other applications. Different biomass uses may have different impacts, and the efficiency of replacing different fossil fuels in different sectors can vary. For example, using biomass in a large stationary plant such as a combined heat and power (CHP) plant to replace fossil-based electricity and heat is more climatically beneficial than substituting fossil transportation fuels with solid biofuels (Joelsson and Gustavsson 2010).

11.3 Radiative Forcing

11.3.1 What is Radiative Forcing?

Radiative forcing is a quantification of the *greenhouse effect*. This effect is caused by particular gases in the atmosphere with a peculiar property that allows short wavelength radiation (for example, visible light and ultraviolet radiation) to pass through, but restricts longer wavelength radiation (for example, infrared radiation) from passing. Solar radiation as sunlight passes through the atmosphere and is absorbed by the earth's surface. The solar radiation that is absorbed by the earth's surface warms up the surface and atmosphere. That heat will then radiate away from the earth, analogous to a fireplace radiating heat felt by those sitting near it. In the absence of GHGs that heat would radiate out into space, but the GHGs block part of the heat and trap it in the atmosphere. The naturally-occurring greenhouse effect is crucial, because it maintains the earth at a suitable temperature for current life. In the absence of GHGs in the atmosphere, the planet would be much colder.

Over the long term, the earth and atmosphere system remains in radiative balance, where the amount of incoming solar radiation absorbed by the system is balanced by the release of the same amount of outgoing longwave radiation (Fig. 11.4). However, as human society has increased the amount of GHGs in the atmosphere in recent decades and centuries, we have created a *radiative imbalance* in which there is less outgoing radiation than incoming radiation. That imbalance is called "radiative forcing". Thus radiative forcing is a measure of rate of flow of excess energy entering the earth system. It describes the state of energy imbalance, where radiative energy

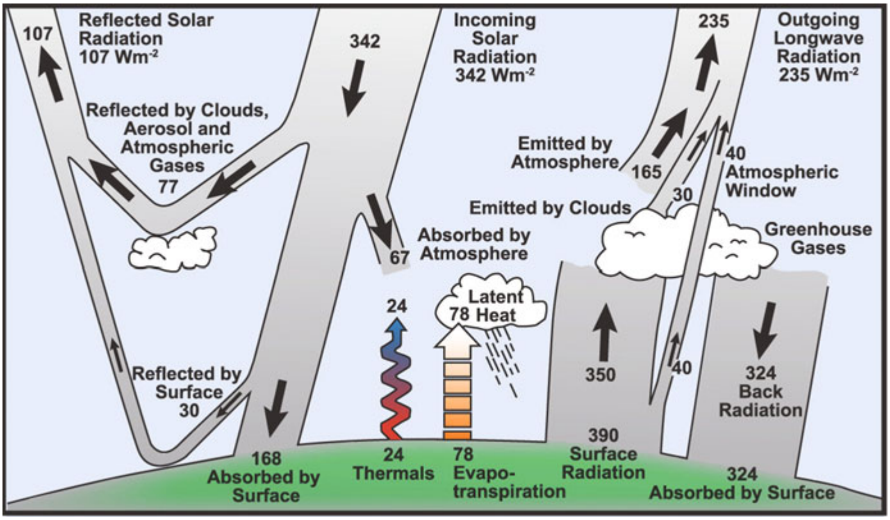


Fig. 11.4 Estimated annual and global mean energy balance of the Earth. (Source: IPCC 2007)

inputs minus outputs yield a positive accumulation of energy in the earth system, leading to climate impacts we experience. When summed over time, the accumulated energy can be termed "cumulative radiative forcing" (CRF), a measure of total excess energy trapped in the earth system, and a suitable indicator of climate change impacts. Hence, positive CRF implies global warming and negative CRF implies cooling.

When a carbon-containing fuel is burned, CO_2 is formed and is released into the environment. Other GHGs may also be emitted at various life-cycle phases, for example methane (CH_4) and nitrous oxide (N_2O). These GHGs mix uniformly throughout the atmosphere and begin to trap heat through the process of radiative forcing. GHGs vary in their radiative efficiency, which determines their ability to accumulate heat. Per unit of mass, N_2O traps the most heat of these three gases, followed by CH_4 and then CO_2.

GHGs are slowly removed from the atmosphere through natural processes, thus over an infinite time horizon a unit of GHG will cause about the same amount of radiative forcing regardless of when it is emitted. However, many policy objectives cover a finite time horizon, for example, reducing climate change impacts during the next 20 or 100 years. Thus the effectiveness of a mitigation activity to fulfil such policy objectives may depend not only on how much GHG is emitted, but also on when it is emitted.

Factors other than atmospheric concentrations of GHGs can alter the Earth's energy balance as well, including aerosols from volcanoes and air pollution and the amount of solar radiation delivered by the sun to the earth. Another potentially important factor in the energy balance is albedo, which is a measure of surface reflectivity. Changes in land surface albedo, e.g., between forested and harvested land, can significantly change the balance of solar radiation and hence radiative forcing, particularly in boreal forest regions (Marland et al. 2003).

11.3.2 Estimating Radiative Forcing

To estimate the radiative forcing implications of various forest management and bioenergy deployment pathways, we integrate three analytical elements: temporally-explicit life cycle system modeling to determine GHG emission profiles; atmospheric decay modeling to determine residence of GHGs in atmosphere; and time-dependent estimates of radiative forcing due to atmospheric concentration changes of the GHGs. Zetterberg (1993) described this method of estimation of radiative forcing, and the parameters have since been updated by the IPCC (1997, 2001, 2007).

To compare bioenergy and fossil energy systems, a detailed life cycle assessment of the elements of both systems is needed. The elements and system boundaries must be described and defined clearly (Schlamadinger et al. 1997). Both the bioenergy chain and the reference fossil chain must be accurately accounted from the natural resources to the delivered energy services. Along the whole chain all processes like transport, conversion, and distribution have to be considered and all GHG emissions must be accounted to compare the full systems rather than the individual components thereof. Comparing bioenergy and fossil energy systems involves both biological and technical features. Clear definition of the reference system and the forest bioenergy system requires delineation of system boundaries in terms of activities, time, and space. Temporal and spatial scales are interlinked through biomass growth rates, and stand-level analysis gives a different perspective than landscape-level analysis. The land that is used in the bioenergy system should be considered also in the fossil reference system. Definition of systems should be done to produce a comparable functional unit of goods and services (Gustavsson and Sathre 2011). Clear definition of the initial boundary conditions, or starting point of the analysis, is needed as it affects later results, though a long-term analysis will identify persistent trends.

Once emitted, a GHG will continue to cause radiative forcing and trap heat in the earth system as long as it remains in the atmosphere. GHGs are removed from the atmosphere by natural processes, at time rates that vary with the GHG. Net emissions of CO_2, N_2O and CH_4 that occur during a year are typically treated as pulse emissions. The atmospheric decay of such pulse emissions can be estimated using Eqs. 11.1, 11.2 and 11.3 (IPCC 1997, 2001, 2007):

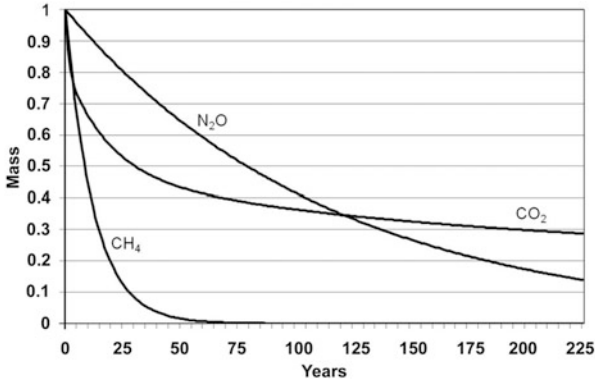
$$(CO_2)_t = (CO_2)_0 \times \left[0.217 + 0.259e^{\frac{-t}{172.9}} + 0.338e^{\frac{-t}{18.51}} + 0.186e^{\frac{-t}{1.186}}\right] \quad (11.1)$$

$$(N_2O)_t = (N_2O) \times \left[e^{\frac{-t}{114}}\right] \quad (11.2)$$

$$(CH_4)_t = (CH_4)_0 \times \left[e^{\frac{-t}{12}}\right] \quad (11.3)$$

where t is the number of years since the pulse emission, $(GHG)_0$ is the mass of GHG emitted at Year 0, and $(GHG)_t$ is the mass of GHG remaining in the atmosphere at year t, where GHG represents CO_2, N_2O and CH_4, respectively. The estimated decay over time of unit mass emissions of CO_2, N_2O and CH_4 is shown in Fig. 11.5.

Fig. 11.5 Estimated atmospheric decay of unit emissions of CO_2, N_2O and CH_4



The time profiles of atmospheric mass of each GHG can then be converted to time profiles of atmospheric concentration, based on the molecular mass of each GHG, the molecular mass of air estimated at 28.95 g mol^{-1}, and the total mass of the atmosphere estimated at 5.148×10^{21} g (Trenberth and Smith 2005). Marginal changes in instantaneous radiative forcing due to the GHG concentration changes can then be estimated using Eqs. 11.4, 11.5 and 11.6 (IPCC 1997, 2001, 2007):

$$F_{CO_2} = \frac{3.7}{\ln(2)} \times \ln\left\{1 + \frac{\Delta CO_2}{CO_{2ref}}\right\} \tag{11.4}$$

$$F_{N_2O} = 0.12 \times \left(\sqrt{\Delta N_2O + N_2O_{ref}} - \sqrt{N_2O_{ref}}\right) - f(M,N) \tag{11.5}$$

$$F_{CH_4} = 0.036 \times \left(\sqrt{\Delta CH_4 + CH_{4ref}} - \sqrt{CH_{4ref}}\right) - f(M,N) \tag{11.6}$$

where F_{GHG} is instantaneous radiative forcing in W m^{-2} for each GHG, ΔGHG is the change in atmospheric concentration of the GHGs CO_2, N_2O and CH_4, respectively (in units of ppmv for CO_2, and ppbv for N_2O and CH_4), $CO_{2ref} = 383$ ppmv, $N_2O_{ref} = 319$ ppbv, $CH_{4ref} = 1774$ ppbv, and $f(M, N)$ is a function to compensate for the spectral absorption overlap between N_2O and CH_4 (IPCC 1997, 2001, 2007).

CRF occurring each year in units of W s m^{-2} (or J m^{-2}) can then be estimated, by integrating the instantaneous radiative forcing occurring through each year. This operation converts the energy flow per unit of time of the radiative imbalance caused by GHGs into units of energy accumulated in the earth system per m^2 per year. Values of radiative forcing are annual and global averages, at the outer surface of the troposphere, estimated here based on earth radius of 6371 km and height of tropopause of 12 km.

The case studies in this chapter are expressed as CRF per unit of forest land (W s m^{-2} ha^{-1}) and CRF per unit of biomass dry-matter (W s m^{-2} Mg^{-1}). Firstly, radiative forcing is estimated in units of Watts of radiative imbalance per square meter of surface area of the troposphere (W m^{-2}). Secondly, this is integrated over time to become Watts × seconds (or Joules of accumulated energy) per square meter (W s m^{-2}). Thirdly, this is normalized to a unit hectare of forest land or a unit ton of dry biomass, to determine the climate mitigation efficiency of forest management on the basis of land area or biomass production.

This type of analysis involves relatively simple models of complex natural processes, thus is subject to uncertainty. The calculations of radiative forcing assume relatively minor marginal changes in atmospheric GHG concentrations, such that radiative efficiencies and atmospheric decay patterns of the gases remain constant. However, significant increases in the atmospheric concentration of CO_2 can be expected during the coming decades and centuries. Increased atmospheric CO_2 concentration will decrease the marginal radiative efficiency of CO_2, but will also decrease the marginal atmospheric decay rate of CO_2. These will have opposite and therefore offsetting effects on radiative forcing, thus we expect this uncertainty to be minor (Caldeira and Kasting 1993). More sophisticated modeling could account for expected future trajectories of GHG concentrations and its effects.

Furthermore, the estimation of CRF by integrating instantaneous radiative forcing over time ignores the feedback effect that the accumulated energy will have on future outgoing radiation. Radiative forcing is a measure of the radiative imbalance given that atmospheric temperatures remain unchanged. In fact, positive radiative forcing will increase the heat energy accumulated in the earth system, leading to more outgoing longwave radiation and an eventual restoration of radiative balance. Since instantaneous radiative forcing does not account for this feedback effect, the described approach may therefore slightly overestimate the amount of accumulated heat caused by a unit of radiative forcing.

11.4 Case Studies

Here we illustrate the effects of the issues discussed above, by quantifying the GHG flows and radiative forcing over time for several illustrative forest biomass system scenarios.

11.4.1 Case Study: Using Forest Residues to Substitute Fossil Fuels

The time dynamics of carbon flows can be significant to the climate impacts of forest residues used for energy. Residues that are removed from the forest and burned release CO_2 into the atmosphere immediately, whilst residues left to decompose naturally in the forest slowly release CO_2 over a time scale of years and decades. If residues are used for energy to replace fossil fuel, the biogenic carbon is emitted immediately, and some fossil emissions also occur from fossil fuels used to recover and transport the biomass. If residues are left in the forest and not used as fuel, a corresponding amount of fossil fuel will be used instead resulting in immediate fossil emissions, followed by gradual emission of biogenic carbon from the decaying residues.

Sathre and Gustavsson (2011) conducted system modelling to estimate the CRF that results from recovering, transporting, and burning forest residues, compared with the forcing that would have occurred if the residues were left in the forest and fossil fuels used instead. Harvest slash and stumps were differentiated as to their recovery difficulty and natural decay rate. CRF was significantly reduced when forest residues were used instead of fossil fuels. Fig. 11.6 shows the reduction in CRF over a 240-year period when slash and stumps are used to replace fossil coal or fossil gas at year 0. The assumed baseline for decay of residues if not removed from the forest follows either natural decay patterns, or is instantly oxidized to determine the significance of this common default assumption. The CRF reduction is greater if the biomass left in the forest is instantly oxidized than if it decays naturally. Natural decay occurs slowly over many years, thus the biogenic carbon is temporarily stored out of the atmosphere, delaying CO_2 emissions and reducing CRF. The natural decay curves are different for slash and stumps because of their different decay rates. Stumps decay slower than slash, thus if the biomass is not used as biofuel the carbon in stumps will remain stored out of the atmosphere for a longer time than the carbon stored in slash. The instant oxidation curves are almost identical for slash and stumps because their stored carbon is assumed to be fully released during the year of harvest.

The type of fossil fuel replaced influenced the result, with coal replacement giving the greatest CRF reduction. Replacing oil and gas also gave long-term CRF reduction, although CRF was slightly positive during the first 10–25 years when these fuels were replaced (Fig. 11.7). Globally, about 27 % of primary energy is supplied by coal, 34 % by oil, and 21 % by gas (IEA 2012). Coal emits more CO_2 per unit of energy than other fossil fuels, thus substitution of coal should be prioritized if possible. Greater climate benefits occurs when logging residues are transported longer distances and used to replace coal than when residues are transported shorter distances and used to replace oil or gas (Gustavsson et al. 2011). Biomass productivity also affected CRF reduction, with more productive forests giving greater CRF reduction per hectare. The decay rate for biomass left in the forest was found to be less significant. Fossil energy inputs for biomass recovery and transport had very little impact on the estimated CRF.

11.4.2 Case Study: Forest Fertilization and Biomass Substitution

More complex systems can also be modelled to estimate overall climate impacts and benefits of forest management and forest product use. Intensive forest management is time dependent, such that actions (e.g., forest fertilization) early in a rotation period result in management-related emissions, followed later by increased biomass growth and substitution potential. Harvested forest biomass is rarely used exclusively for bioenergy, and climate implications of other biomass uses should be considered. Important temporal issues include the duration of carbon storage in forest products, the fate of post-use forest products and the availability of residues at different times.

Over the life cycle of a wood product, biomass residues will become available at different times. Thinning residues may be generated during the growth phase of the forest. Later, forest residues are created when the forest stand is harvested, processing residues are available when the roundwood is transformed into wood products, and construction site residues are left when a building is assembled using wood-based

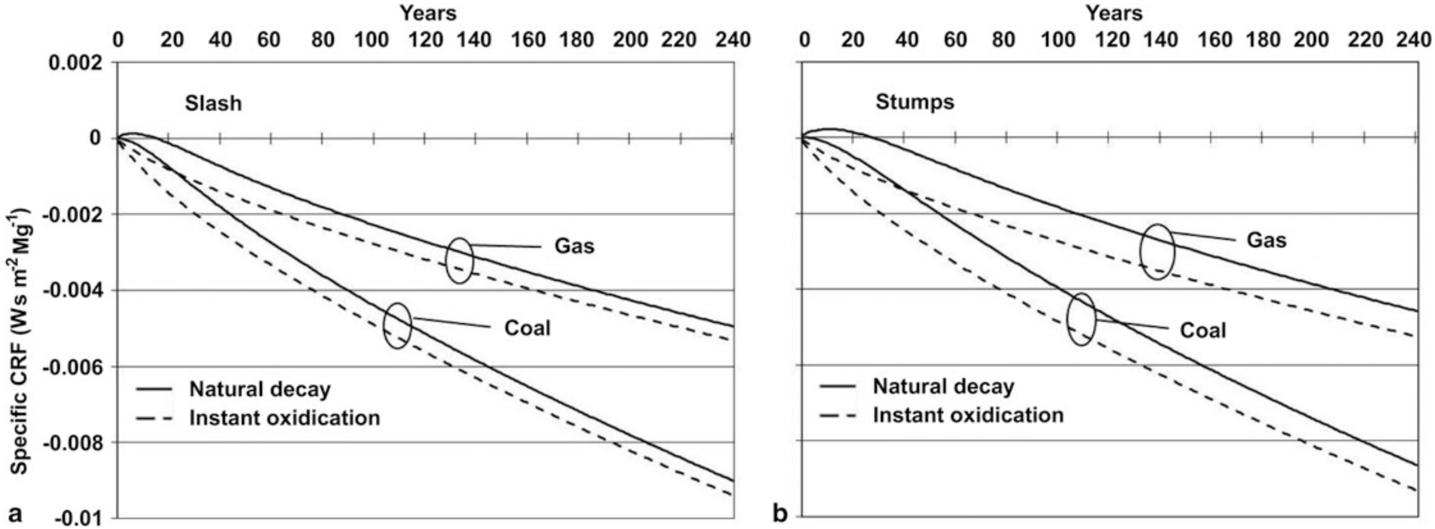


Fig. 11.6 Specific CRF (W s m^{-2} Mg^{-1} dry matter) when slash (**a**), and stumps (**b**) replace fossil coal or gas, assuming baseline of either natural decay or Instant oxidication (Source: Sathre and Gustavsson 2011)

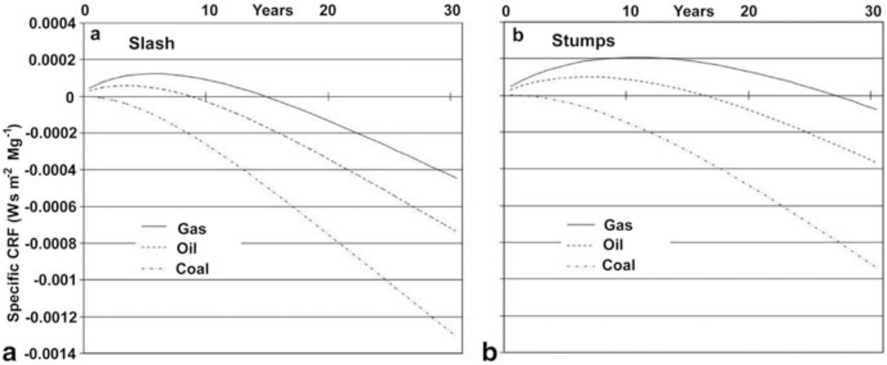


Fig. 11.7 First 30 years of specific CRF (W s m^{-2} Mg^{-1} dry matter) when slash (**a**), and stumps (**b**) replace coal, oil or gas, assuming baseline of natural decay. (Source: Sathre and Gustavsson 2011)

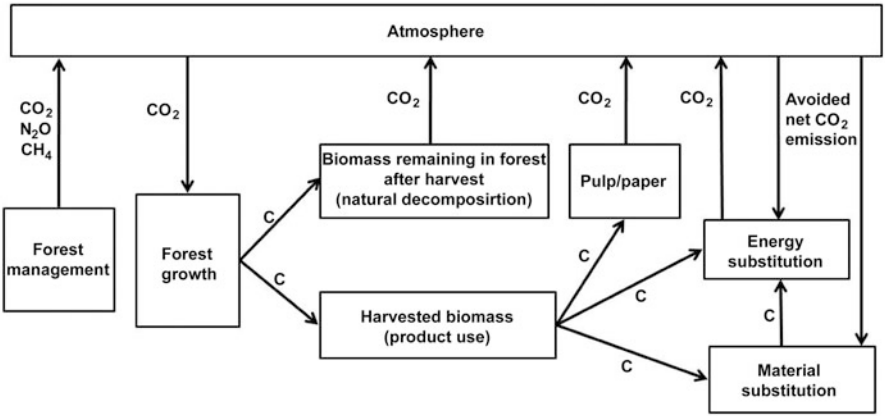


Fig. 11.8 Schematic diagram of GHG flows and stocks tracked on an annual basis. (Source: Sathre and Gustavsson 2012)

materials. Later still, post-use residues are produced at the end of a wood product life cycle. The use of these residues to replace fossil fuel results in reduced fossil carbon emissions at different times in the life cycle of the material.

Sathre and Gustavsson (2012) conducted a broad system analysis of the time-dependent climate implications of various forest management regimes and forest product uses. They tracked the annual GHG flows and stocks for each system (Fig. 11.8), including the fossil CO_2 emissions from forest operations, the CO_2, N_2O and CH_4 from production and application of fertilizer, N_2O emission from fertilized soil, CO_2 from fossil fuels used for biomass harvest and transport, avoided CO_2 emissions from using the biomass to substitute for materials and fuels, and carbon stock changes in living trees, forest products, and in soil and decaying biomass. The annual net emissions of CO_2, N_2O and CH_4 were calculated for each system based on life-cycle modeling, as well as the resulting annual changes in instantaneous

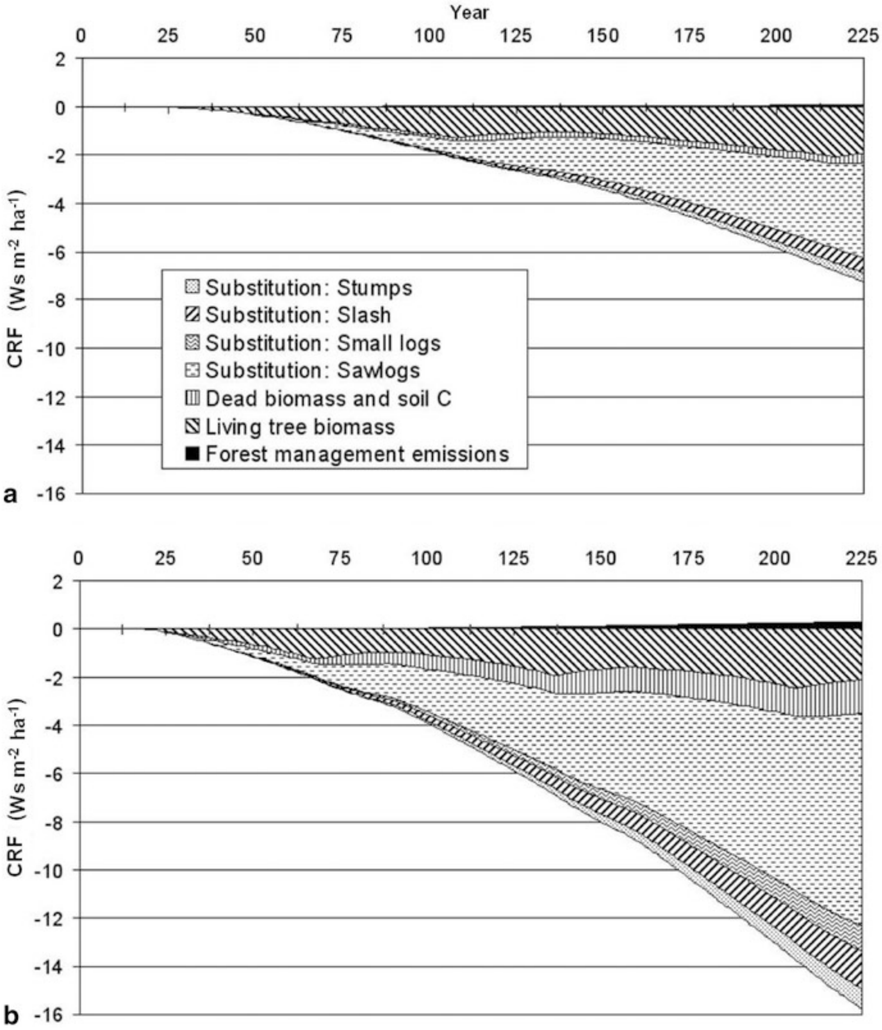


Fig. 11.9 CRF (W s m^{-2} ha^{-1}) of conventional (**a**), and fertilized (**b**) stands, broken down by biomass types and forest management actions. Energy substitution effects are based on replacing coal. Reference unmanaged land use is not considered. The potential impact on CRF due to the use of less forestland in the reference fossil scenario is assumed to be zero

radiative forcing and CRF over a long-term time horizon. As indicators of the efficiency of climate change mitigation of each scenario, Sathre and Gustavsson (2012) estimated the CRF reduction per hectare of forest and the CRF reduction per ton of dry matter of biomass.

The modelled forest management regimes include conventional production forestry in northern Sweden with a rotation period of 109 years, and more intensive fertilized management with a rotation period of 69 years. The CRF of the conventional and fertilized stands is shown in Fig. 11.9. The CRF is broken down into contributions from different biomass types and forest management, with coal as the reference fossil fuel. During the first 15 years the CRF is the same in both managed stands. It begins slightly positive in sign due to fossil emissions from stand establishment, and by Year 10 becomes negative due to carbon uptake in growing trees. The radiative forcing of the fertilized stand increases during Year 16 due to GHG emissions from fertilizer production and application. By Year 18, however, the CO_2 removal due to the increased biomass growth of the fertilized stand compensates for these emissions, and the cumulative radiative forcing of the fertilized stand becomes less than that of the conventional stand. By the end of the 225-year study period, the avoided CRF of the fertilized stand is over twice that of the conventional stand.

Increasing biomass production through forest fertilization combined with substitution of non-wood products and fuels can significantly reduce CRF. The emissions from intensified forest management, including manufacture and application of fertilizer, resulted in very little radiative forcing in comparison with the negative radiative forcing from using the increased forest growth for biomass substitution. The biggest single factor causing reduction in radiative forcing was using sawlogs to produce wood material to replace energy-intensive construction materials such as concrete and steel. Another very significant factor was replacing fossil fuels with wood residues from forest thinning, harvest, wood processing, and post-use wood products. The fossil fuel that was replaced by the biofuels affected the reductions in GHG emission and radiative forcing, with carbon-intensive coal being most beneficial to replace. The climate benefits of fertilization were proportional to the increased rate of biomass production, in terms of shortened rotation lengths and increased harvest volumes.

Here we extend the analysis of Sathre and Gustavsson (2012) to include scenarios of unmanaged forest land in the reference fossil system. If a forest is not managed, at least three effects on carbon balance can be distinguished: (1) The forest biomass would continue growing until the stand is mature. At this point a dynamic balance would be reached, where natural mortality equals growth and average carbon stock remains near-constant. In the long-term, the living forest biomass may be reduced due to natural disturbances (wind, insects, etc.) and reach a new equilibrium lower than that of a managed forest at the time of harvest. (2) The soil carbon stock would vary somewhat depending on the amount of biomass litter from natural disturbance events. (3) No forest products would be produced and other, more carbon-intensive, materials and fuels would be used instead. The substitution effect of forest product use is cumulative; i.e., carbon emission is avoided during each rotation period due to the substitution of fossil fuel and material by the harvested biomass. Thus, not harvesting the forest would cumulatively increase the carbon emission over what would otherwise be possible if the forest stand were harvested and used on a regular rotation period.

The carbon stock in living biomass is here assumed to reach a level of 20 % greater than the harvest level of the conventionally managed stand. Due to uncertainties regarding the long-term development of the living carbon stocks in an unmanaged stand, we consider two scenarios: (a) the carbon stock remains constant at the 20 % greater level, and (b) the carbon stock reaches the 20 % greater level, and then is slowly reduced by natural disturbances to be 20 % less than the harvest level of the

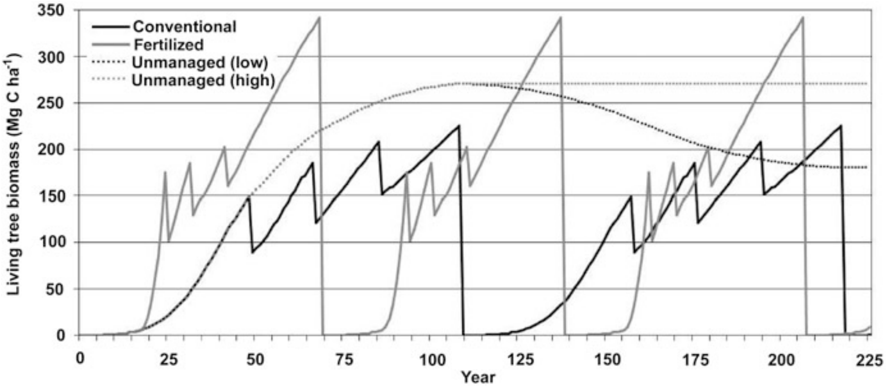
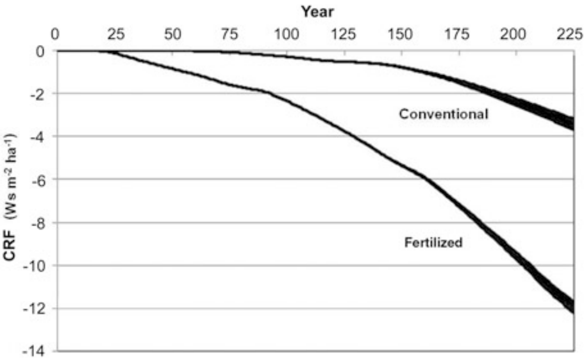


Fig. 11.10 Carbon stock in living tree biomass (MgC ha^{-1}) over time in forest stands with four different management regimes: (1) conventional management with 109 year rotation period, (2) fertilized management with 69 year rotation period, and unmanaged and unharvested with carbon stock stabilising at (3) 20 % above or (4) 20 % below conventional harvest level. For all management regimes, Year 0 begins with a cleared stand

Fig. 11.11 Net CRF (W s m^{-2} ha^{-1}) of forest use, calculated as CRF of conventional and fertilized stands minus CRF of unmanaged stands with high and low levels of equilibrium carbon stock. Energy substitution effects are based on replacing coal



conventionally managed stand. Fig. 11.10 shows carbon in living tree biomass in the conventional and fertilized forest stands, and in unmanaged stands with assumed high and low levels of equilibrium carbon stock.

The net climate benefits of forest management are quantified in Fig. 11.11, which shows the difference in CRF between the managed stands and the reference unmanaged stands. The shaded areas represent the difference in CRF between the low and high scenarios of the unmanaged stand. The CRF reduction of the unmanaged stands is due entirely to accumulated carbon stocks in living biomass and soils, with no ongoing substitution benefits from harvested biomass. The fertilized stand produces earlier climate benefits, and by the end of the 225-year study period it results in over three times greater reduction in CRF than the conventional stand.

The substitution benefits of forest product use are cumulative while the carbon sink in the forest biomass and soil is limited. Therefore, non-management and non-use of forest biomass becomes less attractive as the time horizon increases. Over the long term, an active and sustainable management of forests, including their use as

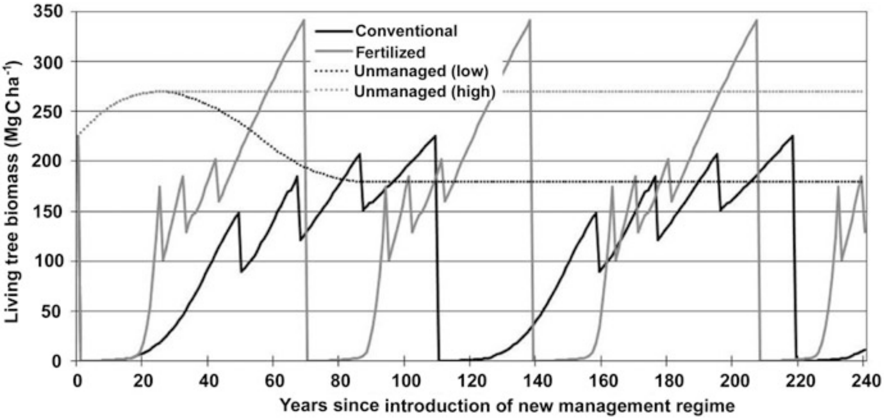


Fig. 11.12 Carbon stock in living tree biomass (MgC ha^{-1}) over time in forest stands with four management regimes (see Figure 11.10 for description). For all management regimes, Year 0 begins with a mature conventionally-managed stand

a source for forest products and biofuels, allows the greatest potential for reducing net carbon emission.

11.4.3 Case Study: Carbon Dynamics at a Landscape Level

A managed forest landscape typically consists of a number of even-aged forest stands, each representing a different age class. At the landscape level, the total carbon balance at any time is the aggregate of the balances of the stands, each of which is at a different stage of its rotation. Final felling occurs by definition only once for each rotation period of a single stand. Because each stand has a different age, only a fraction of the full area is harvested at a time, giving relative stability to productivity and carbon stock at the landscape level.

Here we present a case study of carbon accounting at the stand level and landscape level. The analysis begins with a landscape in northern Sweden composed of 109 separate forest stands of the same area. The stands have been managed conventionally with a 109-year rotation period, and the age structure of the stands is a uniform distribution of stand ages from 1 to 109 years; i.e., one stand becomes mature and is harvested each year. The mature stands contain a carbon stock in living tree biomass of 225 tC ha^{-1}. In this analysis, as each of the 109 stands becomes mature, one of four different management regimes is implemented on the stand: (1) clear-cut harvest followed by continuation of conventional management with 109 year rotation period; (2) clear-cut harvest followed by fertilized management with 69 year rotation period; (3) the stand is left unharvested and unmanaged with carbon stock stabilising at 20 % above or (4) 20 % below conventional harvest level. Stand-level carbon stock trajectories for each of these four regimes are shown in Fig. 11.12. The net effects

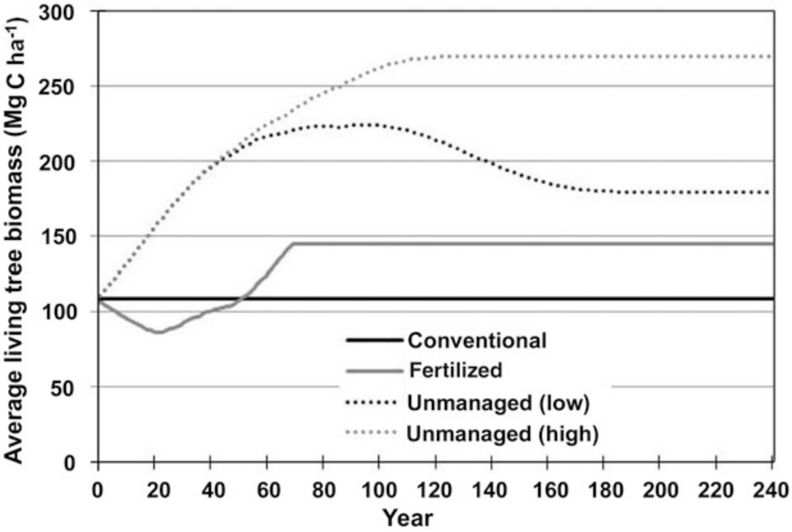


Fig. 11.13 Average landscape level carbon stock (MgC ha^{-1}) in living tree biomass of 109 forest stands under four different management regimes. During each of years 0 through 108, one of the 109 stands reaches maturity under conventional management and then continues under the four different management regimes (except transition to fertilized regime, which occurs from year 0 to 68; see text)

of forest management are calculated as the direct effects of a managed stand minus the reference effects that would have occurred if the stand had not been managed.

The carbon stock in living biomass over time on the landscape level including all 109 stands is shown in Fig. 11.13. If conventional management is continued, the landscape level carbon stock remains constant over time, because the same fraction of the forest land is harvested each year so the age structure remains constant. If fertilized management is adopted, to allow compatibility of the 69-year rotation period across the 109 stands and ensure an even distribution of stand ages on the landscape, we assume that 1.58 stands are harvested each year. This implies that during the transition from conventional management to fertilized management, some existing conventionally-managed stands are harvested before they are fully mature. This results in an initial temporary reduction in landscape level carbon stock, followed by an increase to a stable carbon stock at a higher level than the conventional management. If the forest is not harvested and left unmanaged, the reference landscape level carbon stock increases as each of the stands reaches mature or over-mature conditions. Depending on the equilibrium carbon stock in unmanaged stands, the reference landscape level carbon stock either remains at a high equilibrium or slowly decreases to a lower equilibrium level.

Annual net CO_2 emissions on the landscape scale are shown in Fig. 11.14, calculated as emissions of conventional and fertilized stands minus emissions of reference unmanaged stands with high and low levels of equilibrium carbon stock. Emissions of CH_4 and N_2O due to fertilization are minor and are not shown. The net emissions

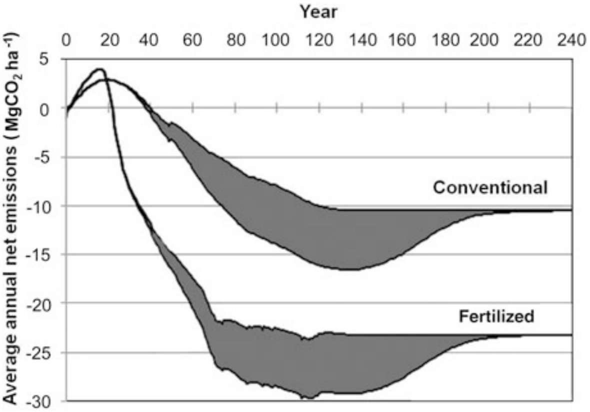


Fig. 11.14 Average annual landscape level net CO_2 emissions ($MgCO_2$ ha^{-1}) of the 109 managed forest stands, calculated as emissions of conventional and fertilized stands minus emissions of reference unmanaged stands with high and low levels of equilibrium carbon stock

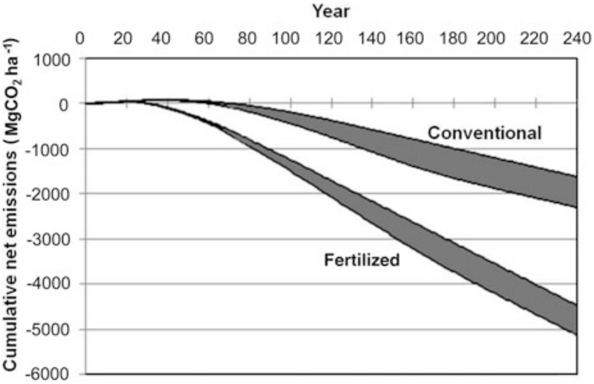
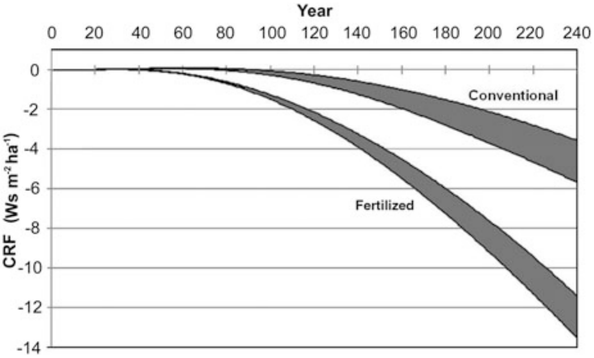


Fig. 11.15 Cumulative landscape level net CO_2 emissions ($MgCO_2$ ha^{-1}) of 109 managed forest stands, calculated as emissions of conventional and fertilized stands minus emissions of reference unmanaged stands with high and low levels of equilibrium carbon stock

include changes in biomass carbon stock, changes in soil carbon stock, changes in wood product carbon stock, emissions from forest management activities, and avoided emissions due to substitution of non-wood material and fuel, estimated using the methodology of Sathre and Gustavsson (2012). The managed regimes result in negative emissions, mainly due to avoided emissions resulting from substitution of carbon-intensive non-wood fuels and materials. The fertilized regime has more avoided emissions than the conventional regime, because the biomass production is greater which allows more substitution of non-wood products. The unmanaged regime ultimately results in zero net emissions, as the forest ecosystem reaches dynamic equilibrium and no forest products are produced and substituted for non-wood products. If the unmanaged forest stabilises at a lower carbon stock, there is a period of net positive CO_2 emissions from dying biomass before reaching long term equilibrium at zero net emissions.

Cumulative net CO_2 emissions on the landscape scale are shown in Fig. 11.15. The unmanaged stands eventually stabilise and make no further contributions to the carbon balance. The managed stands, on the other hand, continue to accumulate

Fig. 11.16 CRF (W s m^{-2} ha^{-1}) over time of 109 managed forest stands, calculated as CRF of conventional and fertilized stands minus CRF of reference unmanaged stands with high and low levels of equilibrium carbon stock



avoided CO_2 emissions due to the use of harvested biomass to substitute for carbon-intensive non-wood fuels and materials. By the end of the 240-year study period, the avoided emissions from the fertilized stand are about double those of the conventional stand, due to the higher biomass production rates of the fertilized regime.

The net landscape level CRF over time is shown in Fig. 11.16, calculated as CRF of conventional and fertilized stands minus CRF of reference unmanaged stands. The fertilized regime results in significantly less climate impacts than the conventional regime, with a net CRF of about -11 to -13 W s m^{-2} ha^{-1} after 240 years, depending on the equilibrium carbon stock of the unmanaged reference case. The conventional regime results in about half as much the avoided net CRF as the fertilized regime, with about -4 to -6 W s m^{-2} ha^{-1} after 240 years. The difference between the CRF of the unmanaged regimes shown in Fig. 11.16 and that shown in Fig. 11.11 is due to the different starting conditions of the scenarios (compare Figs.11.10 and 11.12). One scenario begins with a mature forest, and the other begins with a recently harvested stand. The carbon emission differences between the unmanaged (low) and unmanaged (high) scenarios have a greater impact on net CRF in Fig. 11.16 compared with Fig. 11.11 because emissions occurring earlier in time cause more radiative forcing because they reside in the atmosphere for a longer proportion of the study time horizon. This shows the importance of methodological choices in defining temporal system boundaries, which may be more significant for radiative forcing analysis than for carbon balance analysis.

11.5 Conclusions

In this chapter we have discussed the temporal dynamics of forest bioenergy systems, and have described the development and application of models to estimate the CRF associated with various forest management and forest product use options. This type of analysis provides insight into the potential trade-offs between short-term carbon sequestration benefits and long-term substitution benefits. It may be used to determine optimal strategies for managing forests and using forest products so as to minimise CRF and climate change impacts. Implementing these strategies will require the integration of progressive practices of forest management, fertilization, soil science, forest products industries, energy systems, construction, and waste management.

Future steps are to further develop, improve, and use modeling tools to estimate the climate and energy implications of forest systems over varying spatial and temporal scales. The interactions of the following three important factors linked to forestry should be better understood: (1) Forest management intensity, e.g., conventional, fertilized and unmanaged regimes; (2) Geographic aggregation of analysis, i.e., stand level versus landscape level dynamics; and (3) Initial boundary conditions, e.g., a mature or harvested forest stand, or a range of age classes across a landscape. The patterns of these complex systems are difficult to compare quantitatively using conventional, static carbon balance approaches. Rigorous analysis of temporal dynamics may reveal subjective valuations of time preferences, e.g., the favor given to current population and desires. Radiative forcing analysis may be used to further our understanding of the appropriate roles for forests in the future, and appropriate management strategies for forests as part of a transition to a low-carbon society.

Acknowledgements We gratefully acknowledge the support of Nordic Energy Research and Swedish Energy Agency.

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Part III
Adaptation

Chapter 12
Climate Change and Forest Management: Implications for Energy Biomass Production in Boreal Forest Ecosystems

Ashraful Alam

Abstract Changes in climate interact with the development processes of forest ecosystem in a complex way, which may bring new dimensions to the sustainable management of forest resources. It is important to identify the sensitivity of forest ecosystems to changing climate, and further to understand how to utilize the opportunities and to avoid the risks under climate change. The issue is especially important for the boreal forests in northern Europe, where nearly all the forests are managed. In these conditions, rapid changes in climate may create situations where the traditional guidelines for forest management may no longer be appropriate. In this chapter, these issues are discussed, focusing on the Finnish boreal forests and their management under climate change when producing timber and energy biomass concurrently but still maintaining large carbon stocks in forest ecosystems.

Keywords Carbon stocks · Climate change · Energy biomass · Forest ecosystem · Forest management · Timber production

12.1 Global Climate Change

12.1.1 Emissions of Greenhouses Gases and Climate Change

The global climate is changing rapidly in many and complex ways. In addition to increasing temperature, the frequency of extreme climatic events is increasing, too. The International Panel for Climate Change (IPCC) defines climate change as any changes in climate over time, whether due to natural variability or a result of human activity (IPCC 2007). Recent changes in climate are mainly human-induced due to increasing the global concentration of greenhouse gases (GHG) in the atmosphere, which is causing global warming. The primary sources of the increase in GHG are the excess use of fossil fuels in energy production and changes in land use (conversion of forests to other land). These actions emit GHGs to the atmosphere, which is trapping

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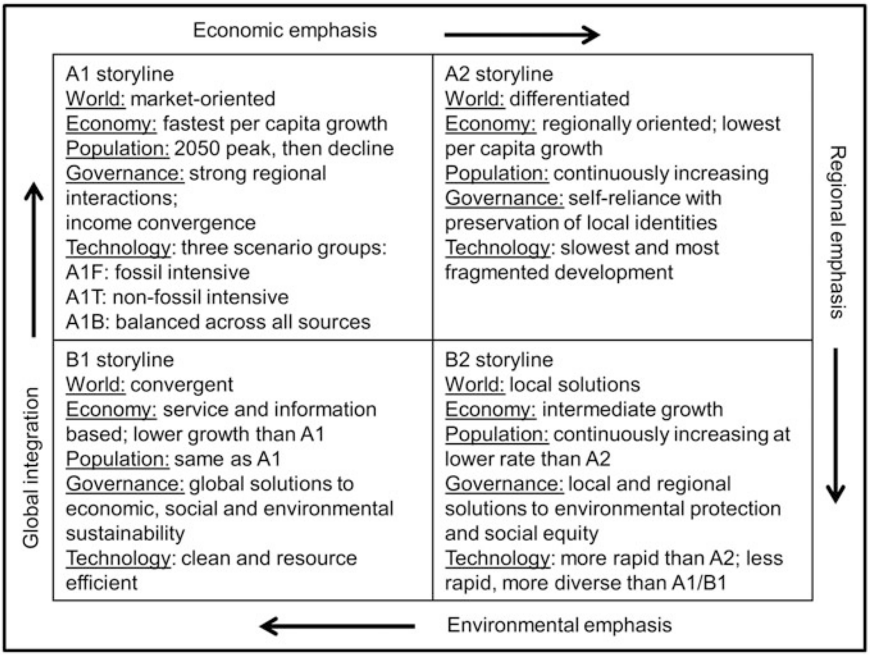


A1 storyline World: market-oriented Economy: fastest per capita growth Population: 2050 peak, then decline Governance: strong regional interactions; income convergence Technology: three scenario groups: A1F: fossil intensive A1T: non-fossil intensive A1B: balanced across all sources	A2 storyline World: differentiated Economy: regionally oriented; lowest per capita growth Population: continuously increasing Governance: self-reliance with preservation of local identities Technology: slowest and most fragmented development
B1 storyline World: convergent Economy: service and information based; lower growth than A1 Population: same as A1 Governance: global solutions to economic, social and environmental sustainability Technology: clean and resource efficient	B2 storyline World: local solutions Economy: intermediate growth Population: continuously increasing at lower rate than A2 Governance: local and regional solutions to environmental protection and social equity Technology: more rapid than A2; less rapid, more diverse than A1/B1

Fig. 12.1 Illustration of greenhouse gas storylines and processes based on those developed by IPCC (2007)

more heat in the lower atmosphere thus making the Earth's climate warmer as well as altering the precipitation patterns. The expected changes in both of these climatic factors are based on the scenarios on the future growth of GHGs in the atmosphere using the scenarios on the growth of economy and human populations, and changes in technology and other related processes behind the emissions of GHGs (Fig. 12.1).

12.1.2 Expected Changes in Climate

Carbon dioxide (CO_2) is the most important among GHGs, with the strongest influence on global warming. The atmosphere consists mainly of nitrogen and oxygen (together 99 %), with smaller proportions of other gases such as CO_2 (0.037 %). The amount of CO_2 in the atmosphere is maintained through a balance between uptake and emission processes such as photosynthesis and respiration in plants, and combustion of carbon-rich materials. The reason for concern is that the level of CO_2 concentration in the atmosphere has increased from pre-industrial times, from 280 parts per million (ppm) in 1750 to 398.40 ppm in 2013 (April). Currently, the concentration of CO_2 in the atmosphere is increasing by 2.54 ppm per year (data from Mauna Loa Observatory, NOAA). The future level of concentration differs among the scenarios, but there is an increasing trend estimated between 550 and 1000 ppm by 2100 (IPCC 2007).

Following the increase in atmospheric CO_2, global *air temperature* has been increasing at a rate which has been $0.13 \pm 0.03\,°C$ per decade during the last 50 years (IPCC 2007). Warming is probably the highest over the land surface at the upper northern latitudes, where the annual mean temperature may increase by 6–8 °C by the end of this century (i.e. 2080–2099) regardless of the climate scenario (Fig. 12.2). At the same time, the *precipitation* throughout the world is likely to change, but the change is highly variable spatially and temporally. For example, precipitation has increased in the eastern parts of North and South America, northern Europe and northern and central Asia (IPCC 2007). Conversely, the drought periods have increased in the Sahelian and Mediterranean regions, southern Africa and in some parts of southern Asia. In general, precipitation is likely to increase substantially at high northern latitudes and to decrease in the subtropics as the past observations indicate.

12.2 Impacts of Climate Change on Forest Growth and Management

12.2.1 Distribution of Global Forests and Climate

The global distribution of major vegetation types in relation to temperature and precipitation is shown in Fig. 12.3. In the high latitudes against arctic tundra, the boreal forests occupy the latitudes, where annual mean temperature is from $-5 \ldots +5\,°C$, and the annual mean precipitation is at least 150 mm. In these conditions, the forests are mainly dominated by coniferous species, and only seldom do broad-leaved species dominate the forest landscape. The productivity of forest ecosystem is from 1 to $10\,m^3\ ha^{-1}\ a^{-1}$ in the boreal zone depending on growing conditions and tree species. In the middle latitudes, the temperature is $+5 \ldots +15\,°C$, with clear seasonality in precipitation. In this region, these temperate forests include a mixture of various broadleaved and coniferous species. In the tropics, the annual mean temperature exceeds 22 °C, and with an ample supply of water the productivity exceeds $20\,m^3\ ha^{-1}\ a^{-1}$. However, the short supply of water seriously limits forest growth in the dry tropics, e.g. only scattered trees occupy savanna grass lands. In summary, the climate has a dominant influence on the structure and functioning of forest ecosystems throughout the world, and it plays an important role in controlling forest growth and consequent carbon sequestration.

12.2.2 Climate Change and Productivity of Forest Ecosystems

Forests take up CO_2 in photosynthesis, thus carbon is accumulated in forest biomass and forest soils. The stomatal functions control the flow of CO_2 into stomatal cavities, where it is available for photosynthesis in mesophyll cells. Open stomata allow

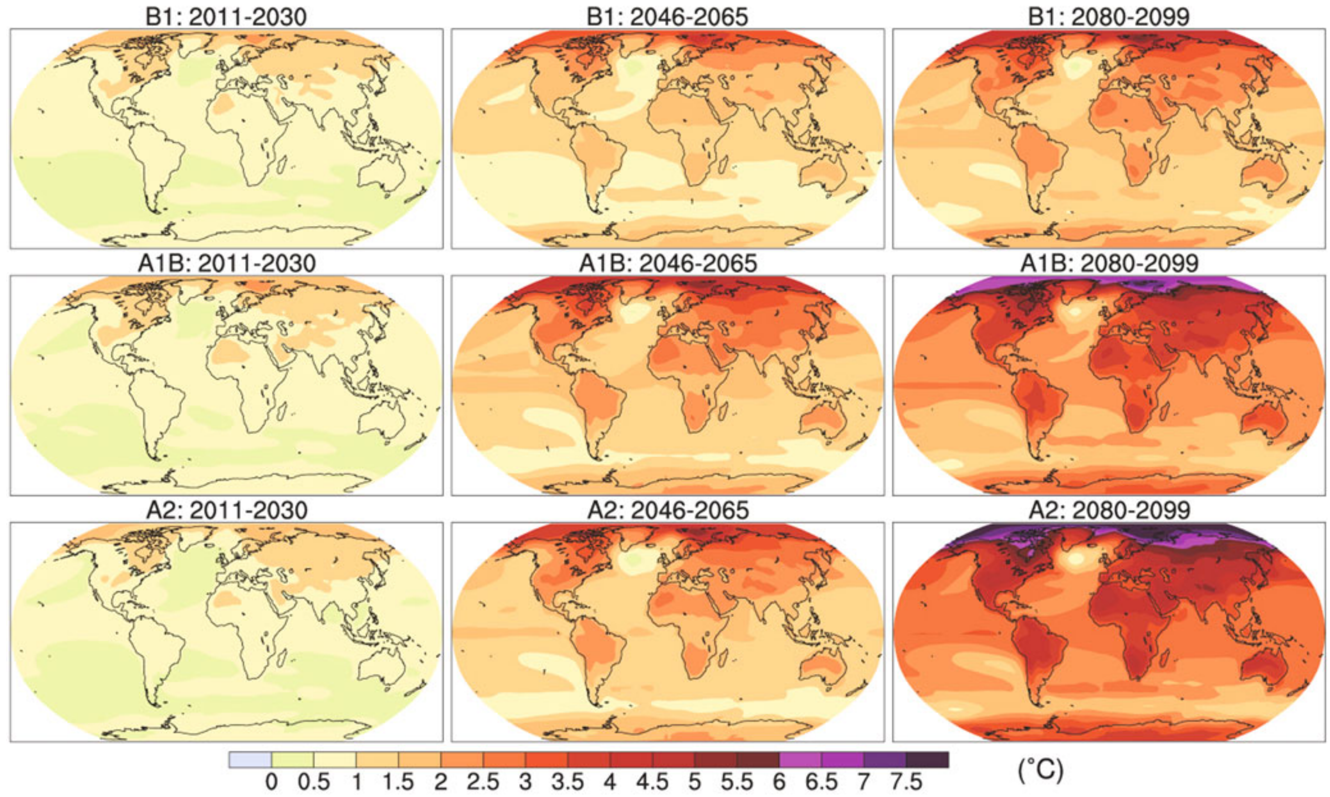
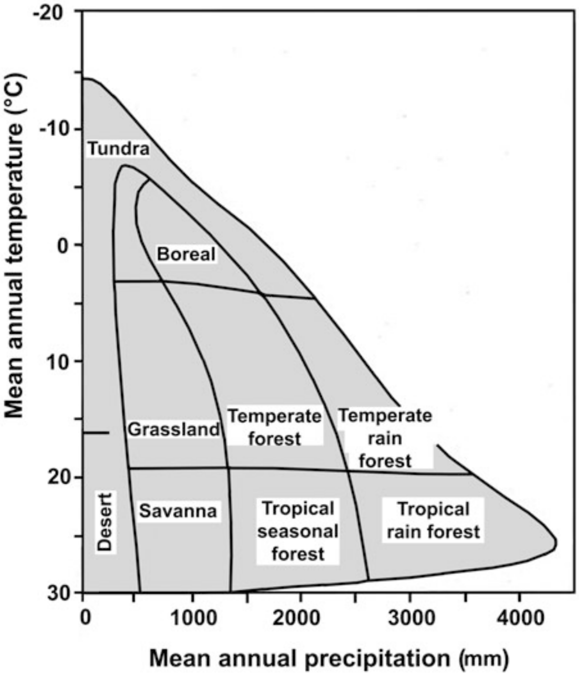


Fig. 12.2 World-wide projected changes in surface air temperatures, relative to 1961–1990 averages, under three different emissions scenarios (*rows*, *top* to *bottom B1*, *A1B*, *A2*) for three different time periods (columns, *left* to *right*: *2011–2030*, *2046–2065*, *2080–2099*). Projected warming is much stronger over land areas and high latitudes. (Meehl et al. 2007)

Fig. 12.3 Distribution of major vegetation types against mean annual temperature and precipitation (Redrawn from Whittaker 1975)



water to evaporate to the atmosphere. Regardless of temperature conditions, higher levels of CO_2 act as a fertilizer and increase the growth of trees. Experiments show that the net photosynthesis may increase up to 50 %, on an average, if the CO_2 concentration increases from 350 to 700 ppm (Cure and Acock 1986). Such high increases in photosynthesis is most probably due to the fact that most of the CO_2 enrichment experiments use plants which are at their early developmental stage, with high capacity for uptake of CO_2. On the other hand, experimental plants are growing under no competition from other plants. The strong effect of high CO_2 concentration on photosynthesis (at leaf level) is probably not repeated to the same extent in the growth of single plants or in the productivity of ecosystems, which are affected by many other factors such as competition over nutrients, water and light (Körner 1993, 2006; Fig. 12.4). However, the water use efficiency of a tree may increase where there is elevated CO_2 concentration (e.g. Battipaglia et al. 2013), which may partly mitigate the effects of reduced availability of water on the uptake of CO_2 if the climate change reduces water supply.

In the northern latitude or boreal conditions, increased temperature alone would increase the CO_2 uptake rate and elongate the growing season, thus stimulating the growth and increasing the productivity (e.g. Alam et al. 2008). The case is different in the Mediterranean region, for example, where increased temperature without sufficient precipitation is expected to cause a decline in productivity due to increasing droughts (Lindner et al. 2010). However, the warming climate may increase the risks of drought even in boreal conditions due to enhanced evaporation

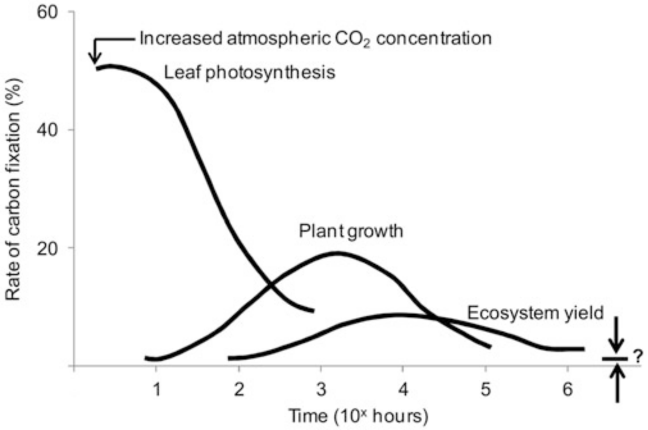


Fig. 12.4 Over short time scales (hours), an increase of atmospheric CO_2 concentrations (doubling from current) causes a significant increase in leaf photosynthesis and carbon fixation. Over medium time-scales (days–years), the increase in leaf photosynthesis causes a gradual increase in plant growth, followed by a gradual decline. Over medium to long time scales (years–centuries), the increase in plant growth leads to an increase in the overall yield of a forest ecosystem. It is important to note, however, that the response of leaf photosynthesis is much larger than the response at the forest level. The effect of increased CO_2 over time scales greater than a century is currently unknown. (Time scale on the *X-axis* is in powers of 10; $1 = 10$ hours, $6 = \sim 114$ years). (Redrawn from Körner 1993; Permission is obtained from the author by personal communication)

regardless of the increase in precipitation. Furthermore, the warming climate would make snow melt earlier thus affecting the availability of water later in the growing season (Kellomäki et al. 2008).

The warming climate may, thus, have varying effects on the growth and productivity of forests depending on the location, growing condition and factors limiting tree growth. In Finland, for example, the growth of forests decreases towards the north (less than $1\,m^3\ ha^{-1}\ a^{-1}$) compared with that (more than $6\,m^3\ ha^{-1}\ a^{-1}$) in south due to the lower temperatures in the north (above 64° N) than in the south (below 64° N). In these conditions, the climate change is likely to increase the growth of forests due to elevated temperature and CO_2 concentration, longer growing seasons and enhanced mineralization of nitrogen bound in soil organic matter (SOM). In the southern boreal conditions, the drought episodes may, however, increase to such an extent that the reduced supply of soil water may locally reduce the forest growth (Berg 2000; Karhu et al. 2010; Ge et al. 2012).

12.2.3 Management of Forest Ecosystems and Climate Change

Climate system, forest ecosystems and forest management are interlinked and they interact with each other in several direct and indirect ways (Fig. 12.5). In Europe and elsewhere, the majority of the forest ecosystems are managed and they play an

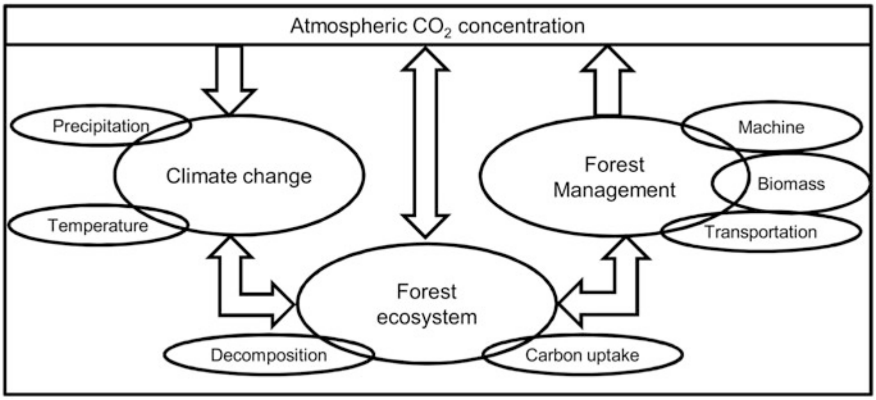


Fig. 12.5 Illustration of interaction between climate change, forest ecosystem and forest management

important role in the economy and the environment. Managed forests provide several ecosystem goods and services based on managing the structure of the ecosystem for making it function as desired. Forest-based production is thus a result of the interaction between the environment and genotype (Kellomäki et al. 2009), which is modified in management to produce biomass (e.g. energy biomass, saw logs and pulpwood) and other ecosystem services (e.g. biodiversity, carbon sequestration etc.). In this context, climate change will bring new dimensions to the management, with a need to revise the current management guidelines in order to meet the changes in structure and functioning of the ecosystem. For example, a warmer climate with changes in growth may make it necessary to change the timing and intensity of thinnings and the length of rotation to meet the possible changes in growth. Forests may have the ability to adapt autonomously to changing climate, but this may not be enough if the climate changes rapidly compared with the past changes, thus emphasizing the need for planned adaptation (Kellomäki et al. 2009).

12.3 Modeling Production and Utilization Potentials of Energy Biomass Under Climate Change—The Case of Finland

12.3.1 Outlines

In Finland, the forests are of the boreal type, where growth is mainly limited by low temperature, short growing season and limited availability of nitrogen. Therefore, climate change is likely to increase the forest growth throughout the country, thus increasing the sequestration and accumulation of carbon in forest biomass and SOM. Furthermore, climate change will have effects on the competition between different tree species, with impacts on tree species compositions. Especially on fertile sites, deciduous species may become more dominant than they are currently. The increasing growth and changes in competition between coniferous and deciduous species may require modification of the current management practices to meet the changes in growing conditions in order to fully utilize the positive effects of climate change and to avoid any harmful effects which climate change may have on growth and development of boreal forests (Kellomäki et al. 2008).

In the following, model simulations are used to demonstrate how the elevating temperature and CO_2 and changes in precipitation may affect the growth and development of managed boreal forests which produce timber and biomass for energy. Regarding the whole territory of Finland, different climatic scenarios (current and changing climate) and management regimes are employed in simulations which utilize a process-based ecosystem model (see Box 4.1 in Chap. 4). The model has parameters for Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* L. Karst.), silver birch (*Betula pendula* (Roth)) and downy birch (*Betula pubescens* (Ehrh)) growing between the latitudes N 60° and N 70° and longitudes E 20° and E 32° within Finland.

12.3.2 Climate Scenarios

The simulations utilize the downscaled climate data for Finland provided by the Finnish Meteorological Institute. Scenarios for the climate projection are based on the average responses calculated using nineteen global climate models, where variables such as minimum and maximum temperature, precipitation, solar radiation, air pressure, snow depth, soil moisture and wind velocity have been provided (Jylhä et al. 2004, 2009; Ruosteenoja et al. 2007). The grid for current climate (1971–2000) is 10×10 km, whereas the scenario for changing climate the utilized grid is 50×50 km. The climate change scenarios are given in three tri-decadal periods; i.e. near-term, 2010–2039, mid-term, 2040–2069 and long-term, 2070–2099. In the simulations, the A2 scenario is utilized (see Fig. 12.1), where CO_2 concentration is expected to rise to 840 ppm by 2100, whereas the mean temperatures for January is expected to increase by 7.6 °C and for July by 3.4 °C by the end of this century over the whole of Finland. In winter, warming is strongest in the north, while in summer the greatest warming takes place in the south. Precipitation is estimated to increase about 30 % in the winter and about 10 % in late summer by 2100.

12.3.3 Management Scenarios

The Finnish National Forest Inventory data is used as an input in simulations (Fig. 12.6c). In the simulations, bioenergy thinning (or energy biomass thinning), commercial thinning, final felling and regeneration are used in management. The thinning rules, based on the development of basal area and dominant height, are

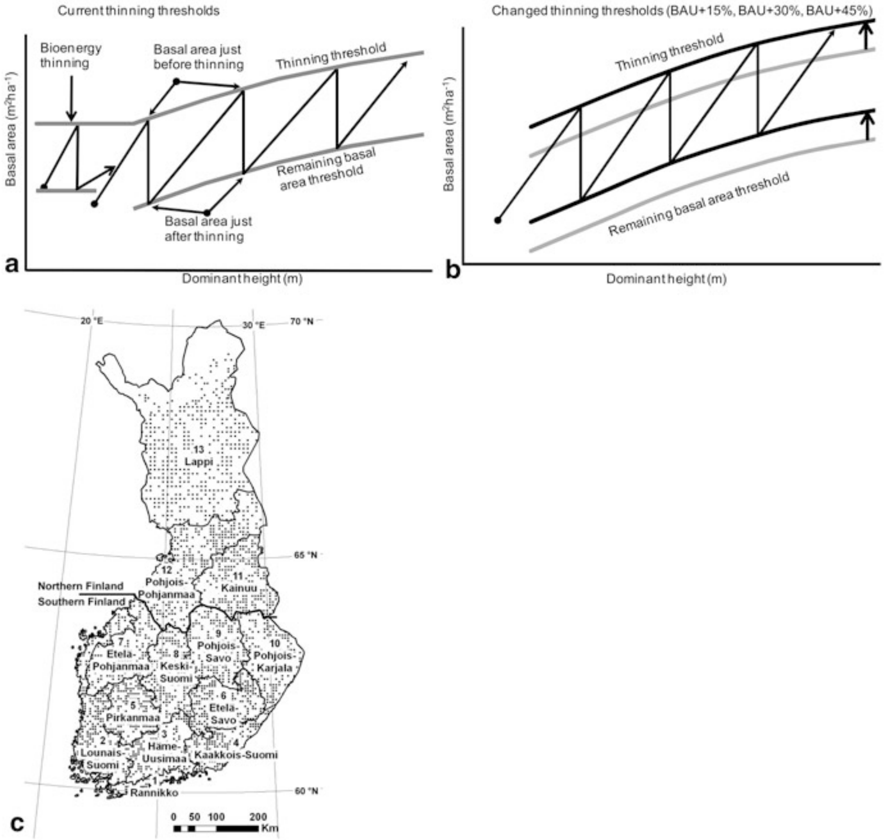


Fig. 12.6 **a** Principles defining the thinning regime based on development of dominant height and basal area in Finland, current recommendation is considered as business-as-usual thinning regime. Before the commercial thinning, bioenergy thinning is done and followed the site- and species-specific recommendation. **b** Thinning regimes are changed in terms of increased thinning thresholds, compared to business-as-usual recommendation. **c** The management regimes are applied for the whole of Finland as seen in the map. The line in the middle of the map separates southern and northern Finland, and number indicates different Forestry Centers (South: 1–10; North: 11–13) in Finland. *Dots* on the map show the permanent sample plots used in the Finnish National Forest Inventory

those currently recommended for different tree species, site fertility types and regions (Tapio 2006). Thinning is done from below (suppressed trees are removed first, and then from among the dominant trees) to such a level that the remaining basal area is reduced to the expected value at a given dominant height (Fig. 12.6a). Bioenergy thinning is also done based on the site- and species-specific recommendations (Tapio 2006). The final felling is made whenever the mean diameter of the trees in the plots exceeds the given value indicating the maturity for regeneration.

The modified management regimes are constructed by changing the basal area value triggering thinning and the basal area value to be retained in thinning (see

Fig. 12.6b). Both thresholds were increased by 15, 30, and 45 % (thereafter used BAU + 15 %, BAU + 30 % and BAU + 45 %) compared with the current recommendations (BAU + 0 %). The thresholds for bioenergy thinning are the same for both current and increased thinning thresholds. In all the management regimes, energy biomass and timber were produced. Energy biomass represents small-sized trees from bioenergy thinning and logging residues (tops of stem, branch, foliage and stumps) from final felling, whereas timber represents pulpwood (top diameter 6–15 cm) and saw logs (top diameter > 15 cm) produced in commercial thinnings and final felling. The rest of the stem (top diameter < 6 cm) was included in logging residues.

12.3.4 Effect of Climate Change on Forest Growth and Carbon Sequestration

In Finland, forest growth (stem wood increment, $m^3\ ha^{-1}\ a^{-1}$) may increase substantially under a warming climate compared with the current climate (Fig. 12.7). The largest relative changes may occur in the north, although the absolute values are still highest in the south (see also Bergh et al. 2003; Briceño-Elizondo et al. 2006; Alam et al. 2008; Lindner et al. 2010). Currently, temperature sum in the north is about 700 degree-days, while in the south it is about 1,230 degree-days. Under changing climate, temperature sum increased 74 % in the north part by the end of the century (2070–2099) compared with the current climate (1971–2000) (Fig. 12.7). Responding to increased degree-days, the growth increased 50 % in the south, but doubled in the north (Fig. 12.7). These changes indicate the dominant role of temperature in controling growth in the north, even though the absolute growth was still the largest in the south. However, the potential increase in growth was reduced in the south due to water limitations. Especially, the growth of Norway spruce may be reduced on the sites with low water-holding capacity (Briceño-Elizondo et al. 2006; Kellomäki et al. 2008) due to the increase of drought episodes (frequency, duration) as is likely to occur in the central-European lowlands (e.g. Lasch et al. 2002; Lindner et al. 2010).

12.3.5 Effect of Climate and Thinning on Energy Biomass, Timber and Carbon Stocks

The production of energy biomass and timber increased along with the current and warming climate when applying current management (Fig. 12.8). The production was also affected by the management regime by increasing the thinning thresholds. Even during the first period (2010–2039), the production of timber and energy biomass increased by climate change regardless of the management regimes. During the second and last period, both the climate change and increased thinning thresholds enhanced the production of energy biomass further, but only the climate change enhanced

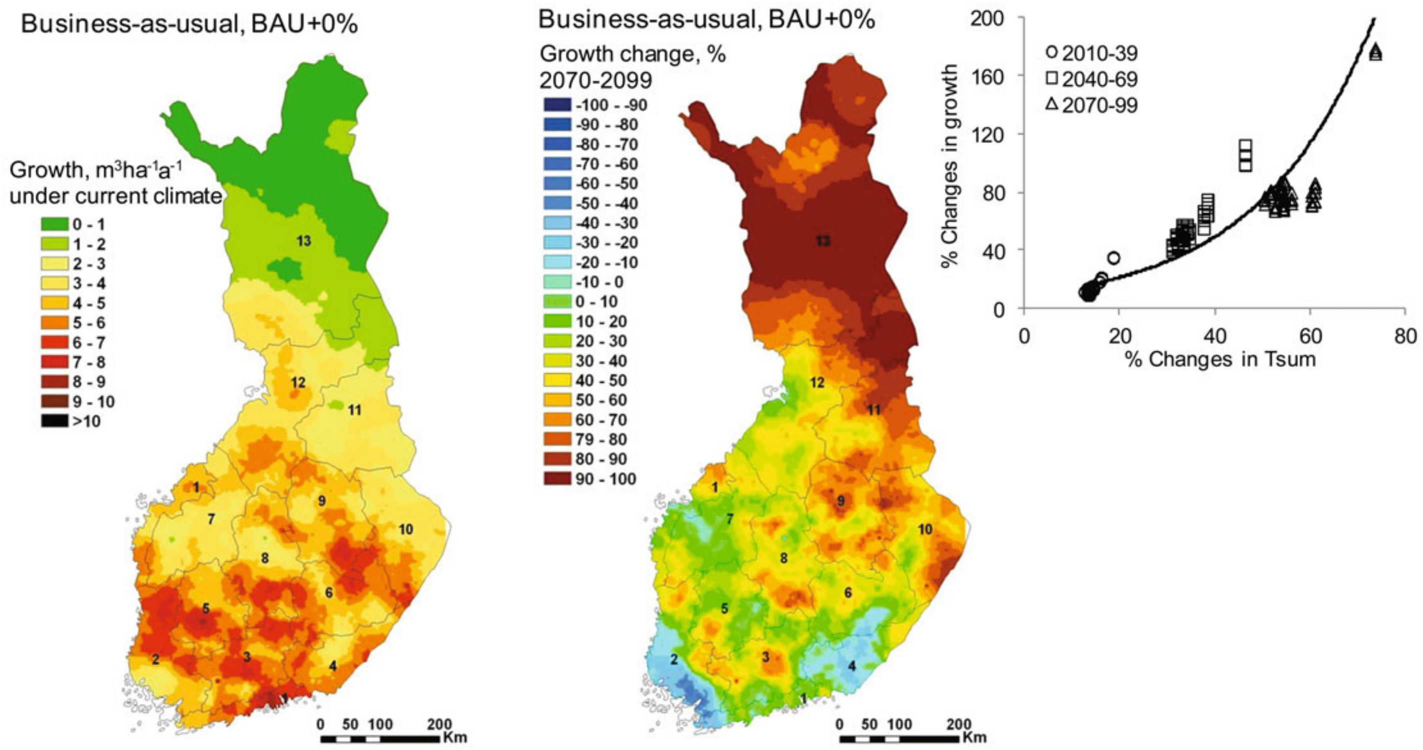


Fig. 12.7 Maps: Forest growth in current climate (left) and its relative changes in changing climate compared to current climate (right) under business-as-usual management (BAU + 0 %). Insert in the *upper right* panel indicates the relative effects of growth against temperature sum under changing climate in three 30-year periods between 2010 and 2099 over the whole of Finland for all the management regimes (BAU + 0 %, BAU + 15 %, BAU + 30 % and BAU + 45 %). (Alam 2007)

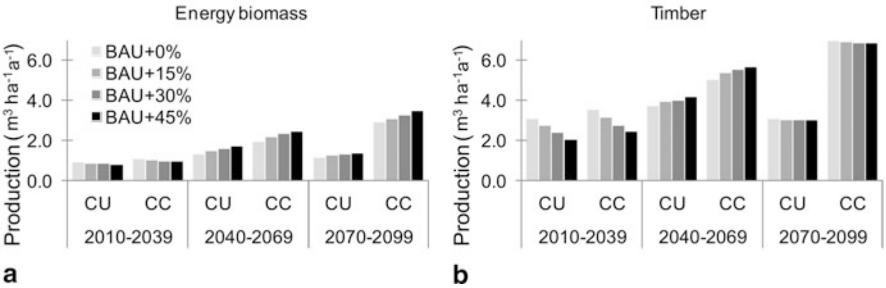


Fig. 12.8 **a** Effects of thinning regimes on energy biomass production from energy biomass thinning and final felling and **b** timber production from commercial thinning and final felling, in three 30-year periods between 2010 and 2099 under current climate (*CU*) and changing climate (*CC*) in the whole of Finland. BAU + 0 % represents the current thinning regime (see Fig. 12.6)

the timber production (Fig. 12.8). Similar findings are reported in some previous studies as regards the effects of thinning regime and rotation length on the timber and biomass yields and sequestration of carbon in forest ecosystems (Kaipainen et al. 2004; Alam et al. 2008).

However, it is still open how to manage forest to combine the carbon sequestration and the production of timber and energy biomass, because these two objectives are partly conflicting (Kirschbaum 2003). For example, undisturbed forests store the most carbon, while utilization of wood requires removal of wood thus reducing on-site carbon storage. In this regard, increased thinning thresholds may partly mitigate this conflict (e.g. Garcia-Gonzalo et al. 2007; Alam et al. 2010). Furthermore, the increases in the initial stand density combined with the increased thinning thresholds could increase the production of timber, energy biomass and maintain high carbon stocks in the ecosystem (Alam et al. 2012). Regarding rotation length, the current length or a moderate elongation seems to be appropriate in achieving these management aims (Liski et al. 2001; Pohjola and Valsta 2007).

12.4 Conclusions

In boreal conditions, the expected climate change, as a function of elevated temperature, longer growing season and higher precipitation with concurrent elevation of CO_2, may increase forest growth and biomass production. A concurrent increase in energy biomass and timber production as well as carbon stocks under climate change and even under current climate can be possible if the stocking remains high throughout the rotation, may be slightly longer than currently recommended. In addition, proper choice of management may increase energy biomass production on one hand, and may reduce management related emissions on the other hand. However, the growth predictions for the future are still uncertain due to the uncertainties in climate scenarios. Therefore, the likely reduction of growth in some places in the southern boreal areas at the end of this century is probable but much depends on the climate scenario used in the simulations. In this particular case, the annual precipitation increased but the summer precipitation remained the same as currently or even slightly reduced. At the same time, the evaporation increased substantially under a warmer climate, thus reducing the soil water in the growing season. The simulations showed that the responses of the forest ecosystem to the changing climate is a dynamic process, which may increase or reduce the forest growth depending on how optimal the prevailing conditions are for different tree species and how forests are managed to meet the changes in climate.

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Chapter 13
Risk and Uncertainty in Adaptive Forest Management Under Climate Change

Rasoul Yousefpour and Bo Jellesmark Thorsen

Abstract This chapter deals with the complex decision-making problems involving uncertainty and risk that climate change implies for forest owners and managers. Modeling approaches that could support adaptive management strategies are needed because climate change implies increased economic uncertainty. On the other hand, new and reliable information becomes available as time passes and improves our understanding of the impacts of climate change on the dynamics of forest ecosystems. We demonstrate the need for updating decision makers' beliefs and knowledge about the climate state and apply this in adaptive decision-making.

The approach explicitly allows evaluation of the effect of learning on decision-making as future climate is gradually becoming known based on the observations of climate state and its impacts. The time that will elapse before this depend on the divergence among climate trajectories, the long-term rate of change, and the short-term variability in climate. Relying on updated beliefs, adaptation to climate change at each decision point in time may call for a change in forest management regime. This is the case if it ensures optimal utilization of forest resources, e.g. for biomass and bioenergy production. Moreover, optimal decisions may differ locally depending on the risk attitude of forest owners and the bio-geography of their forest resources.

Keywords Adaptive decision making · Beliefs · Behavior of decision makers · Local adaptation · Bayesian update · Biomass production

13.1 Handling Uncertainty and Risks in Management

In general, decision-making in forestry has its focus on how to handle the hazards and risks in management (Blennow 2008; Hanewinkel et al. 2010) or the risks in prices and markets (Buongiorno and Zhou 2011; Hildebrandt and Knoke 2011). In

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this respect, we are facing new uncertainties still little addressed in research, namely those related to climate change and their likely impacts on forest ecosystems. A key word is “likely”, because this issue is highly uncertain due to a lack of complete knowledge or historical parallels of how climate change impacts on ecosystems. More fundamentally, the degree of climate change itself is uncertain. This situation calls for modeling that would support adaptive management strategies.

The core of adaptive management is to collect and integrate the necessary knowledge (as it becomes available) on how ecosystems are likely to respond to alternative management schemes and changes in environmental conditions (Yousefpour et al. 2012). This leads to a continuous decision process (Prato 2008; Probert et al. 2010), which involves a chain of state-dose-response-impacts, where management actions affect all the individual links. The outcome of such a chain of events is a set of flows of forest-based goods and services. Potentially, this leads to a final or steady state of forest ecosystem, which is in balance with the changed climate. A good adaptive forest management strategy is thus designed to pursue the best possible overall outcome in terms of a specific performance measure (Zhou et al. 2008; Yousefpour 2009).

Regarding forest management, the climate change will induce changes in risks and uncertainties (Jacobsen et al. 2010). The changes in risk variables (and other state variables) are likely to follow the development of changes in temperature, precipitation, wind and atmospheric carbon dioxide (CO_2). This is likely to result in a transition, where the growth and risk dynamics under the current climate will change to a new one which is stable under the new climate (Yousefpour et al. 2012). This implies that the distributions of all variables are non-stationary in means and higher orders over the transition period. Distributions may not be adequately captured, e.g. by the Geometric Brownian motion processes or similar ways. Thus, an alternative model for the possible development of future climate could be a model that simulates the development of core variables in the form of trend-stationary processes, with a hetero-scedastic and time dependent variance (Yousefpour et al. 2013).

Box 13.1 Concepts of Risk and Uncertainty

Risk and uncertainty are sometimes considered to have distinctly different meanings. In the Knightian sense, the former implies a form of empirical or objective measure of risk, e.g. probabilities and/or impacts, whereas uncertainty does not (or it relies on subjective probabilities). However, this distinction is far from being used consistently in the literature. Many papers and even books (e.g. Dixit and Pindyck 1994) apply the term “uncertainty” to refer, e.g., to the variation in prices, but clearly apply empirical (objective) measures and models to measure this uncertainty (e.g. Thorsen 1999). In fact, there is a tendency to use the term “risk” mainly for detrimental events like storm and fire. This is related strongly to the risk aversion, where risk has the broader definition of variation in outcomes. For this reason, we apply both the terms of risk and uncertainty, and try to remain true to their use and application in the literature.

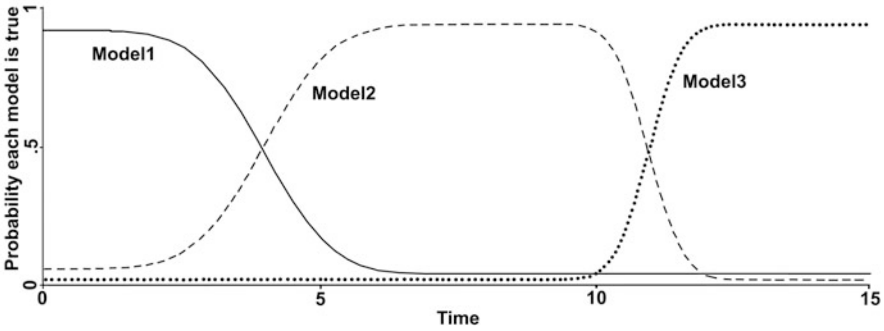

Fig. 13.1 Idealized Bayesian inference, where the posterior probability of different models changes over time with the expansion of the likelihood when more data are entered into the analysis. Depending on the problem context, the time scale on the x-axis may vary, but it is long enough for information to be gathered and analysed, replacing hypothesis 1 in favour of hypothesis 2. This may further be replaced by model 3 as more evidence is accumulated. (Gelman and Shalizi 2012)

13.2 Adaptive Forest Management Under Climate Change

In forest research, risk dynamics is widely studied in the context of prices; i.e., how decision-makers can optimally design adaptive management schemes with decision rules sensitive to forthcoming information (Thorsen 1999; Jacobsen and Thorsen 2003; Yousefpour 2009). Most interesting are the approaches that deliberately integrate the forthcoming information in the analysis of decision alternatives by using stochastic dynamic programming (McDonald-Madden et al. 2010; Probert et al. 2010; Iverson and Perrings 2012). In this case, the stochastic process of relevant variables and their development are assumed to be known for a given period. For example, the estimates of empirical distributions and data generating processes for prices are assumed valid for future decisions, when constructing and applying transition probability matrixes in closed form models. Acknowledging the inherent uncertainty in climate change, we resort to an alternative approach, namely the updated Bayesian approach (Box 13.2, Fig. 13.1) correspondingly applied in biological conservation (McDonald-Madden et al. 2010, Probert et al. 2010). In this approach, possible extreme changes in climate (from no change to high change) and the stochastic nature of changes in future climatic development are addressed in exploring proper ways to manage risks associated with climate change. These guidelines and the modeling framework are the major concern in this chapter.

Box 13.2 Outlines of Bayesian Interference Used in Modeling
According to Gelman and Shalizi (2012), Bayesian statistics starts with a prior distribution, obtaining new data and information, and uses this to create the posterior distribution. The accumulation of evidence is summarized in a posterior distribution. This process implies that as information arises over time, there may be a rise and fall in the posterior probabilities of various

models (cf. Fig. 13.1). The goal is to learn about general laws and trends, as expressed in the probability that one model or another is correct.

13.3 Decision Making in Forest Management Facing Climate Change

Implicitly, most state-of-the-art models assume that forest managers are rational within the limits of the decision-making model, e.g. the forest managers are knowledgeable about forest growth, risk and uncertainties, and impacts of various changes and shocks. However, there is much evidence that forest managers base their decisions on different sets of information and use it in quite different ways than assumed (Blennow 2008; Hoogstra 2008). The successful decision-making in adaptive forest management depends on that managers are aware of the changes in the development of the state of forest and available management strategies (Jacobsen and Thorsen 2003; Linkov et al. 2006; Yousefpour 2009).

Strategies to manage climate changes include the mitigation of climate change and the adaptation to climate change (Yousefpour and Hanewinkel 2009). Mitigation implies benefits at the global scale, whereas adaptation is highly local (Füssel and Klein 2006). Adaptation is focused on a particular location under a particular ownership (Füssel and Klein 2006), and it integrates the beliefs of decision-makers on the potential realization of climate change at the local level (Yousefpour et al. 2012). In this respect, decision-makers may be categorized as determinists and active-adaptives. The determinists believe in no climatic change, and they are stuck to their absolute belief in no change (Jacobsen et al. 2010). The active-adaptive look forward and monitor actively the properties of climate, and they update their knowledge and beliefs about the actual climate realization (Jacobsen et al. 2010). General attitudes of decision-makers (determinists/active-adaptive) have large impacts on decisions, although they do not intend to include this behavior in their decision-making process.

13.4 Modeling Forest Management to Adapt to Climate Change

13.4.1 Models of Possible Climate Trajectories

Climate development and future climate trajectories are multi-dimensional by nature and climate state variables, e.g. temperature and precipitation, could be interpreted as indicators of climate change (Allen et al. 2000). We model the deterministic part of the trajectory from the past to current climate state. This includes stochastic shocks (independent and identically distributed, i.i.d.) according to a Wiener noise process with the variance, σ^2, across state, time, and climate models. The deterministic part of the absolute climate change for climate trajectory i and time t is $\boldsymbol{x}_{it}$. We consider

I models of how the climate may develop such that the observed state of climate related variables, $\hat{\theta}_t$ at time ***t*** given climate model ***i*** is given by Eq. (13.1):

$$\hat{\theta}_t(\text{model}_i, t) = x_{it}(\text{model}_i, t) + \varepsilon_{it} \approx \text{N}(x_{it}, {\sigma_i}^2) \tag{13.1}$$

where, $\boldsymbol{t} = 1, .., \boldsymbol{T}, \boldsymbol{i} = 1, \ldots, \boldsymbol{I}$, $\mathbf{x}_{it}$ the mean trajectory by model ***i*** at time ***t*** and ε_{it} is an error with normal distribution around the mean **0** and scenario specific variance, ${\sigma_i}^2$. Finally, any observation of climate state belongs to a noncommittal probability distribution on climate change: $\text{N}(x_{it}, {\sigma_i}^2)$.

13.4.2 Decision-Maker's Beliefs and Processing of Information

We define a set of potential scenarios of climate development based on various models, of which only one can be true in any specific case and analysis. Let us set up a decision framework, where the decision maker holds a set of beliefs regarding the likelihood of each possible climate model being the true one. The decision maker may change his/her beliefs using Bayesian updating depending on any new observations. Let $\boldsymbol{w}_{it}$ be the belief in a particular climate change scenario such that beliefs are complete Eq. 13.2:

$$\sum_{i=1}^{m} w_{it} = 1, w_{i,t} \geq 0 \tag{13.2}$$

for all $\boldsymbol{i}, \boldsymbol{t}$. Thus, $\boldsymbol{w}_{it}$ is the decision-maker's perceived probability at time ***t*** that a climate change scenario ***i*** is the true representative of the climate state $\{\boldsymbol{w}_{it} = \mathbf{Pr}\ (\mathbf{model}_{\boldsymbol{i}}, \boldsymbol{t})\}$ given all the information are available. As time passes and new knowledge about the climate, as given by $\hat{\theta}_t$, is obtained through monitoring, the plausibility of each model is reassessed and the weights, $\boldsymbol{w}_{it}$, are updated. A complete faith in a model is indicated by $\boldsymbol{w}_{it} = \mathbf{1}$, whereas $\boldsymbol{w}_{it} = \mathbf{0}$ indicates no faith. For each period the climate itself or its observable consequences are monitored. Consequently, an estimate of the climate state $\hat{\theta}_t$ is obtained. We make use of this information to update our beliefs in each of the alternative models, using Bayes' theorem (Bayes and Price 1763) Eq. 13.3:

$$\boldsymbol{w}_{i,t+1}(\hat{\theta}_t) = \Pr\left(\text{model}_i | \hat{\theta}_t\right) = \frac{\Pr\left(\hat{\theta}_t\, |\text{model}_i\right) \Pr\left(\text{model}_i, t\right)}{\sum_{i=1}^{I} \Pr\left(\hat{\theta}_t\, |\text{model}_i\right) \Pr\left(\text{model}_i, t\right)} \tag{13.3}$$

where $\Pr(\hat{\theta}_t|\text{model}_i)$ is normal distributed with the probability density function ($\boldsymbol{PDF}_{it}$) Eq. 13.4:

$$\Pr\left(\hat{\theta}_t\, |\text{model}_i\right) = PDF_{it} = \Phi\left(\frac{\hat{\theta}_t - x_t}{\sigma_i}\right) = \frac{1}{\sigma_i\sqrt{2\pi}} e^{-\frac{1}{2}\left(\frac{(\hat{\theta}_t - x_t)^2}{{\sigma_i}^2}\right)} \tag{13.4}$$

The weights at time $\boldsymbol{t}+\mathbf{1}$ depend on the applied climate model and the observed climate state at time $\boldsymbol{t}$. The observed $\hat{\theta}_t$ is an estimate of the realized climate state and as in Eq. (13.1) includes the process variance. The values of $\hat{\theta}_t$ could be drawn by the Monte Carlo technique and from a normal distribution, i.e. $\mathrm{N}(x_{it}, {\sigma_i}^2)$ to simulate particular sequences of climate realizations for given true models.

Climate change may introduce an extraordinary variation to the properties of climate (e.g. increase/decrease in temperature, precipitation, duration of growing season). Therefore, monitoring of different climatic properties under potential changes is necessary to recognize the new climate developments.

In general, meteorological monitoring produces time-series, where the variation in different properties is independent of each other. However, the combination of the most significant climatic parameters on a developmental pathway of climate could serve in defining a particular climate change realization. This may be especially useful when considering how different climate change realizations are built in terms of increase and decrease in the properties of climate. To generalize the two-dimensional consideration of climate state as in Eqs. (13.4) and (13.5) summarizes the modeling procedure developed for the multi-dimensional state of climate based on meteorological monitoring:

$$PDF_{it} = \frac{1}{(2\pi)^{K/2}\left|\sum\right|^{1/2}} e^{-\frac{(\hat{\lambda}_{It}-\lambda_{it})^T \sum^{-1} (\hat{\lambda}_{It}-\lambda_{it})}{2}} \tag{13.5}$$

where $\hat{\lambda}_{It}$ is the observation vector of climate parameters under the actual climate realization $\boldsymbol{I}$. It is based on the normal distributions with the matrix of averages λ_{It} and the matrix of realization errors ε at time $\boldsymbol{t}$. The probability distribution function (***PDF***) of the matrix of climate realizations $\boldsymbol{i}$ can be calculated from Eq. (13.5), where $\boldsymbol{K}$ is the number of variables and $|\Sigma|$ is the determinant of Σ, the covariance matrix, i.e. $\Sigma_{i,j} = Cov(\lambda_i, \lambda_j)$, $\boldsymbol{i}$, $\boldsymbol{j}$ are properties of two different climate realizations. In addition, ***PDF*** is highly dependent on the new information, $\hat{\lambda}_{It}$, and the difference between $\hat{\lambda}_{It}$ and eventual observations under an alternative climate realization record λ_{it}. We make use of this information in Eq. (13.5) to update our beliefs about the entire set of realizations.

However, the effective belief about climate realizations at time $\boldsymbol{t}$ may consider information looking further back than just a period ($\boldsymbol{t}$ − 1) but put more weight on the recent knowledge. Therefore, we integrate the former beliefs in an autoregressive function to conclude the effective current belief $\boldsymbol{W}_{i,t+1*}$. According to Eq. (13.6), $\boldsymbol{W}_{i,t+1*}$ is possible to retain the memories about former beliefs and knowledge extending back five decades i.e. 50 years ($p = 1, 2, .., 9$).

$$w^*_{i,t+1} = \frac{\sum(1 - 0.2(P_t - p))w_{i,t}}{\sum(1 - 0.2(P_t - p))} \quad \text{and} \quad \mathrm{p} = \{1, 2, \ldots, P_t\} \tag{13.6}$$

13.4.3 Decision-Maker's Objective

The decision-maker evaluates the decision alternatives to achieve an objective, which could follow any economic, social, ecological interest or their mixtures. We consider a decision-maker with economic interest, who aims at maximizing an objective (***OBJ***) of the forest by choosing at each time step the decision alternative that implies the best expected ***OBJ***. The ***OBJ*** measure can reflect a particular management, which is optimized conditionally on the different climate change scenarios being true. This method is applicable to the forest management scenarios where a set of sequential decisions must be made and the underlying system dynamics are Markovian. We determine the management action, depending on the objective, time, and the current state of the system. In our problem, the state variable is an information state or belief in each model, $\boldsymbol{w_{it}}$. For each time step, all possible decisions are evaluated for every possible combination of a discretized set of model weights, $W_t = \{\boldsymbol{w}_{1t}; \boldsymbol{w}_{2t}; \ldots; \boldsymbol{w}_{It}\}$. Thus, the decision process depends on defining probabilities of transition from one information state to another and the value of being in that state. These transition probabilities are calculated based on the information obtained at each time step using Bayes' theorem (see Eq. (13.3)). $\boldsymbol{E}\,(\boldsymbol{w_t}, \boldsymbol{t})$ is used to denote the expected value of a management strategy, $\boldsymbol{a_{tj}}$, from time $\boldsymbol{t}$ to the end of planning period $\boldsymbol{T}$ so that the optimal action $\boldsymbol{a_{tj}}$ satisfies Eq. 13.7:

$$\max_{a_{tj}}/\min\ E(W_t, t) = \sum_{i=1}^{I} w_{it} OBJ_{it}(a_{tj}) \tag{13.7}$$

The value function $\boldsymbol{E}\,(\boldsymbol{W_t}, \boldsymbol{t})$ is the weighted sum of the expected ***OBJ***s from the action $\boldsymbol{j}$ given by model $\boldsymbol{i}$, and the reward received at the decision point $\boldsymbol{t}$. The updated model weighting $\boldsymbol{w_{it}}$ comes from Eq. (13.6), and it is this updating and combination process that ensures our management is adaptive in nature.

13.5 Application of the Model for Adaptive Management—Case Study

13.5.1 Study Area and Management Schemes

The study area of 10,000 ha is located at the southeastern part of Veluwe in the centre of the Netherlands. The area is characterized by an undulating topography with pushed moraines and drifts. The elevation of the landscape ranges from 20 to 100 meters above sea level, with a deep ground water table. The annual mean precipitation is 880 mm and the annual mean temperature of 9.6 °C. The forests in the area have been used for different purposes for centuries. This has lead to the degradation of forest to heath land, which is further enhanced by grazing. The research area consists of ca. 75 % forest, 22 % heath land (mainly *Calluna vulgaris* L. Hull), and small patches of drift sand and some agricultural land.

The majority of the current heath land is kept open by grazing cattle, horse, red deer, fallow deer, roe deer and wild boar, which also limits regeneration of trees in the area. The study area represents a diverse ownership with ca. 275 landowners. There are seven large owners (possessing a total of 9700 ha) and in addition, there are 250 small private forest owners, who own altogether 300 ha of the area. The forest area is dominated by Scots pine (*Pinus sylvestris* L., > 50 %), mainly planted. Natural mixed forests with endemic tree species, including European beech (*Fagus sylvatica* L.), Pedunculate oak (*Quercus robur* L.), Sessile oak (*Quercus petraea* L.) and Silver birch (*Betula pendula* Roth.), make up nearly 20 % of the forest. Douglas fir (*Pseudotsuga menziesii* Mirb. Franco), Norway spruce (*Picea abies* L.), larch (*Larix decidua* Mill. and *Larix kaempferi* Lamb. Carr.), and Red oak (*Quercus rubra* L.) are introduced species, which comprise 15 % of the forest.

At the regional level, management is regulated by the program "Veluwe 2010", which combines all landscape functions important for the region (economic, historical values, nature conservation, biodiversity, recreation). The management schemes are forward looking, and they consider the potential impacts of climate change, emphasizing the precautionary adaptation to climate change (Yousefpour et al. 2012). Under multifunctional management, the composition and age structure of tree species are maintained with diversity regardless of forest owner. Currently, the harvest rate is 40–80 % of the growth. The highest harvest rates are on the properties where the timber production is a dominant element of the multifunctional management.

There is conflict between nature management, including large grazers, and timber production goals in parts of the area. Moreover, climate change might lead to the decline or loss of specific components of biodiversity. Increased attention to risky situations with regard to fire calls for management actions on a landscape scale. Owners and managers will need to co-operate more closely to avoid or minimize areas with high fire risk. Forest management might be adapted to influence the microclimate to minimize expected climate change impacts. Such measures could include choosing different tree species, or keeping the forest more dense. Probably no general solution is possible to solve conflicts between different uses and goals of the forest. These should be solved case by case, involving changes in management strategies and perhaps a change in goals.

We simulated four management schemes for every case; Multifunctional forestry (M1), Multifunctional forestry with a focus on production (M2), No intervention (M3), and No intervention with removing non-native tree species (M4). The four alternatives M1–M4 indicate the management with the focus on the adaptive, adaptive and productive, precautionary adaptive, and conservation, respectively. We assume that under a particular climate realization one of these schemes will be optimal subject to the goals and the structural specifications of the ecosystem under adaptive operation.

13.5.2 Forest Landscape Model "Landclim" and Future Climate Realization

13.5.2.1 Landclim Model

We used a forest landscape simulation tool "LandClim", which simulates forest succession and equilibrium conditions as expected under natural conditions

Table 13.1 Regional circulation model realizations for the IPCC AR4 A1b emission scenario at the Veluwe study area in the Netherlands. Summer refers to the period from April to September

Climate scenario	Temperature (°C)		Precipitation (mm)	
	Mean	Min	Annual	Summer
HC (2081–2100)	12.64	3.28	734.5	331.3
SMHI (2081–2100)	11.24	3.02	809.6	372
MPI (2081–2100)	12.27	4.59	807.5	344

SMHI model realization by the Swedish Meteorological and Hydrological Institute at the Rossby Center; *MPI* model realization by the Max-Planck Institute (MPI) in Germany; *HC* model realization by Met Office Hadley Centre (HC) in UK

(Schumacher and Bugmann 2006). Major European tree species (33 species) are parameterized for LandClim. The simulation tool operates at the landscape level, which is represented by a network of grid cells, representing plots of 30 × 30 m. Moreover, LandClim uses water-holding capacity to indicate differences in soil site conditions across the landscape. Regarding trees, the model input includes species, biomass and age of individual trees on a plot.

13.5.2.2 Climatic Realization

Table 13.1 shows the properties of the three climate realizations used in the study (Yousefpour et al. 2013). The climatic data for the transition from the current climate to the future climate was based on the A1B scenario of the Fourth Assessment Report of the IPCC (Intergovernmental Panel on Climate Change, 2007, see chapter 12 for details). This regional climate data were derived from three climate models from the Max-Planck Institute (MPI), the Hadley Centre (HC) and the Rossby Center (SMHI). According to the climate change projections for the Veluwe region, the SMHI model projects a moderate change in climate resembling the current climate, whereas the MPI and HC models are extreme with less precipitation during the growing season and a larger increase in temperature than in the SMHI.

13.5.3 Modeling for Adaptive Management

13.5.3.1 Updating Beliefs

To support the multi-dimensionality of climate change, we include three climate properties, namely, mean temperature, minimum temperature and precipitation for the realizations of future climate states. We draw the future observations from a multivariate normal distribution of the three climate properties applying Monte Carlo simulation in the MATLAB-R2011a environment. Assuming an actual climate realization (i.e. SMHI, MPI or HC), we make use of the drawn Figures over time to update our beliefs about the actual climate realization against alternative realizations.

13.5.3.2 Adaptation Goals

The global objective of adaptive management of natural resources under climate change is to avoid any disturbances to the ecosystem, in general, and the reduction of the productivity of ecosystem, in particular. Therefore, we aim at maintaining the steady state of the forest ecosystem and define our objective within a goal programming approach. We define a goal to hold the total biomass production in a steady state minimizing deviations from the initial state at any decision point (Eq. (13.8)). Total biomass production includes harvest, mortality and remaining biomass in the forest area per ownership (single or multiple) from the decision point to the end of the planning horizon:

$$\min_{M_{ty}} Goal_{+}^{-}\,(W_{t^*}, t) = \sum_{r=1}^{R} W_{it^*} BIOM_{it+}^{-}(M_{ty}) \tag{13.8}$$

$\boldsymbol{M_{ty}}$ refers to the management action $\boldsymbol{y}$ starting from time $\boldsymbol{t}$ and for the time span of the planning horizon. $Goal_{+}^{-}\,(W_{t^*}, t)$ denotes the expected average deviations from the initial state (decision point) and after the implementation of the management action $\boldsymbol{M_{ty}}$ (to the end of the planning horizon) and based on the set of beliefs $\boldsymbol{W_{t^*}} = \{\boldsymbol{W_{1t^*}}, \boldsymbol{W_{2t^*}}, \ldots, \boldsymbol{W_{It^*}}\}$ about various climate realizations $\boldsymbol{I} = \{1, 2, \ldots, I\}$. The performance measure of the management actions is $BIOM_{it_{+}^{-}}$ representing the absolute deviation of total biomass production from time $\boldsymbol{t}$ to the end of planning horizon $\boldsymbol{T}$.

13.5.3.3 Implementation of the Modeling Concept

We implement the concept for the planning horizon of 2010–2100 and make decisions in three points of time: 2010, 2020 and 2050. Updated beliefs after ten years (2020) or at the middle of the planning horizon (2050) can be integrated into the decision-making process and may consecutively require a switch in adaptive decisions made at an earlier decision point. For this purpose, we take into consideration permutations of the simulations for the four management schemes and in two decision points 2010 and 2020 and iterate the simulations for each of three climate realizations. Simulations were conducted in forest management unit i.e. stand level, which provides the opportunity to make decisions for different ownerships (single or multiple, i.e. locally or regionally).

13.5.4 Results

13.5.4.1 Beliefs Update, Climate Realizations and Decision-Makers

Table 13.2 shows the value of updating beliefs under three assumptions about the actual climate realization, i.e., SMHI, MPI and HC and over time (2010–2090 = $\boldsymbol{W}1 - \boldsymbol{W}8$).

Table 13.2 Beliefs' update over time vs. decision-makers vs. the actual climate realization

Climate realization	Updated belief (%)								Effective belief (%)	
	W1	*W2*	*W3*	*W4*	*W5*	*W6*	*W7*	*W8*	*W2**	*W5**
SMHI	33	83	93	51	82	99	99	✓	61	73
MPI	33	13	13	2	54	99	✓	✓	21	25
HC	33	4	1	17	3	0	0	0	17	2

W Belief (e.g. *W2* = belief at the second decision stage 2020),
*W** effective beliefs aggregated from Eq. 13.6 to retain the memories about former beliefs and knowledge backing up to five decades i.e. 50 years (e.g. *W2** = effective belief at 2020), *SMHI* Model realization by the Swedish Meteorological and Hydrological Institute at the Rossby Center; *MPI* Model realization by the Max-Planck Institute (MPI) in Germany; *HC* Model realization by Met office Hadley Centre (HC) in UK. ✓ No doubt about the actual climate realization i.e. $W = 100\,\%$

In general, the active-adaptive decision-maker, denoting equal probabilities to future climate realizations, could update the knowledge continuously over time to recognize the underlying actual climate realization. If HC is the actual climate realization, even the active-adaptive decision-maker is in doubt and may not arrive at perfect certainty. The determinist decision-maker shows an extreme behavior and comes to either a complete confidence or negligence of the actual climate realization. The time needed to come to an absolute confidence is highly dependent on the actual climate realization. It is longer if there is a considerable change (MPI) and hardly achievable if the change is very high (HC). According to Eq. (13.6), we applied an autoregressive function to model a rational belief making process and rely more on the recent information and beliefs. The effective beliefs set at decision times (i.e. $\boldsymbol{W_{2*}}$ and $\boldsymbol{W_{5*}}$) are the same figures as initial beliefs for the determinist decision-maker. However, these effective beliefs bring a great deal of information for the active-adaptive decision-maker and can actually use at the decision points for adaptation (e.g. 2020).

13.5.4.2 Decisions on Adaptive Actions

Figure 13.2a shows the result of process for an active-adaptive decision-maker at the initial stage 2010 and indicates that: M1 is the decision for the total region and Owner 6. This is not the decision for every individual owner and different management actions may be chosen based on the specifications of the area. For example, the regional decision M1 increases the minimum expected deviation for Owner 7 from 5.2 to 8.9, i.e., by 30 %. Moreover, the minimum deviation to be expected from the most desirable decisions is varying to a considerable extent, where the least is expected from M2 for Owner 7 and the highest from M4 for Owner 4. Similarly, Fig. 13.2b shows the case for a determinist decision-maker. There is an obvious overlapping between both sides of Figs. 13.2a, b. This indicates that some owners will make the same decision regardless of their nature of decision-making, i.e. active-adaptive vs. determinist. However, this is not constant and two determinist Owners

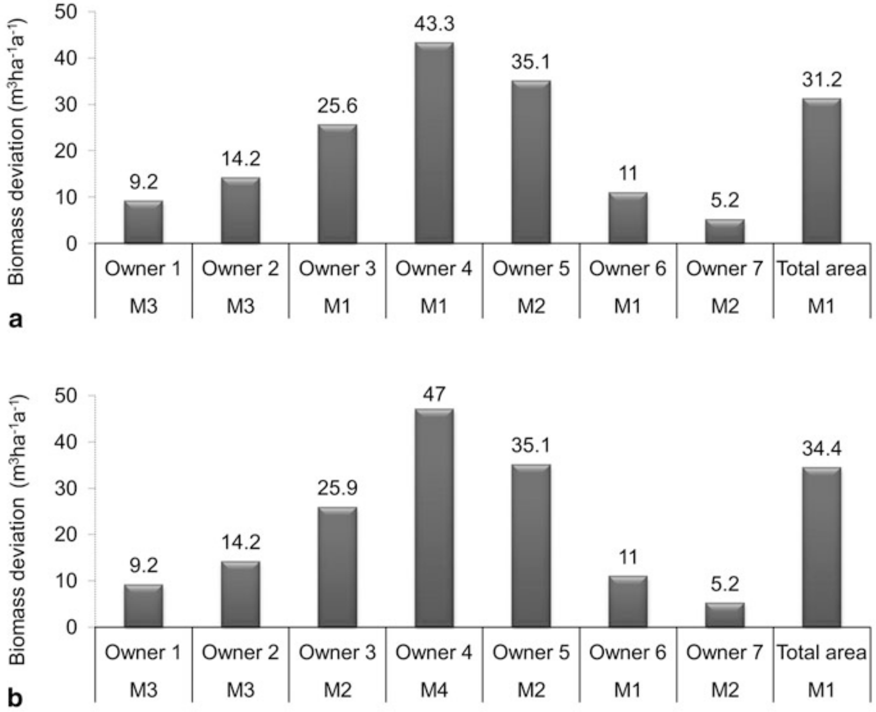

Fig. 13.2 Decisions of an active-adaptive decision-maker (**a**), and decisions of a determinist decision-maker (**b**) at the initial decision stage 2010

3 and 4 decide for M1, whereas they chose M2 and M4 if they were active-adaptive decision-makers.

After a decade of climate observations, the initial beliefs were updated for both active-adaptive and determinist decision-makers. Determinists, however, stick to their initial beliefs ($\boldsymbol{W}_{2010}$ (SMHI) = $\boldsymbol{W}_{2020}$ (SMHI) = 100 %) and evaluate the management actions for the decision horizon 2020–2100. Table 13.3 shows the adaptive decisions of different owners (local decisions) and the decision for the entire area (total, unique management action for all ownerships in southeastern Veluwe). There is no need to switch the management actions for Owners 1, 3 and 5 (Fig. 13.3). These owners make the decisions depending on the initial state of their properties and performance of alternative management actions under climate change uncertainty, i.e. $BIOM_{2020^-_+}$ (SMHI, MPI, HC). It is obvious that by reconsidering the adaptive decisions at 2020, the determinist decision-maker can reach a better minimum deviation for the most of the local areas and especially for the total area (i.e. 31.2 m^3 ha^{-1} a^{-1} at 2010 and 19.4 m^3 ha^{-1} a^1 at 2020).

The response of the active-adaptive decision-maker is substantially different from that of the determinist decision-maker. While the determinist decision-maker remains

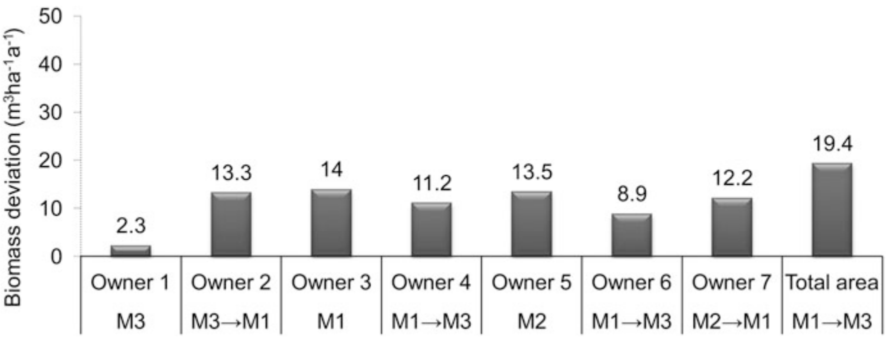

Fig. 13.3 Decisions of determinist decision-makers (owners) at the second decision stage 2020

Table 13.3 Decisions on the adaptive management action; owners vs. the actual climate realization at the second decision stage 2020

	Action	Biomass[a], m^3 $ha^{-1}a^{-1}$ per climate realization		
		SMHI	MPI	HC
Owner 1	M3	10.3	9.9	9.0
Owner 2	M3→M1	2.6	2.4	1.7
Owner 3	M2	5.7	6.1	6.8
Owner 4	M4→M3	6.2	6.9	7.3
Owner 5	M2	16.6	16.5	16.1
Owner 6	M1→M3	16.6	16.3	15.4
Owner 7	M2→M1	11.0	11.2	11.3
Total	M1→M3	16.6	21.6	21.4

[a] Deviation from initial biomass production for the planning horizon 2020–2100

SMHI model realization by the Swedish Meteorological and Hydrological Institute at the Rossby Center, *MPI* model realization by the Max-Planck Institute (MPI) in Germany, *HC* model realization by Met office Hadley Centre (HC) in UK, *M1-M4* management alternatives for adaptive, adaptive and productive, precautionary adaptive, and conservative management of forest areas

indifferent to the new observations, the active-adaptive owner integrates the novel information into the updating system and changes the set of beliefs about the climate realizations (see Table 13.2). This affects the outputs of decision-making and means the decisions are based on the most recent knowledge about the actual climate realization as in Table 13.3. Owners 1, 3 and 5 do not need to change their initial decision, and they continue with the same management action, whereas other owners need to switch to a new management action. However, this is surprisingly not depending on the assumption about the actual climate realization. In this stage, the effect of actions' performance $BIOM_{it_{+}^{-}}(M_{ty})$ is much stronger than effective beliefs $\boldsymbol{W_{It^{*}}}\}$ (Table 13.2). Furthermore, the expected deviations of biomass production under diverse assumptions about the actual climate realization are just slightly different. Changing the regional decision from M1 to M3 decreases the deviations to a considerable extent, i.e. from 34.4 m^3 ha^{-1} a^{-1} at 2010 to 16.6, 21.6 and 21.4 m^3 ha^{-1} a^{-1} at 2020, and assuming the actual climate realizations **SMHI**, **MPI** and **HC**, respectively.

13.6 Discussion and Conclusions

Regarding climate change, the adaptive management involves spatial aspects from local to regional levels, and from owner to owner. Thus, the adaptive decisions are dependent on structural properties of the biological system and the management actions. Modeling the adaptive decision-making is made more complex by the variability of how decision-makers perceive (behavioral system) the climate change. The more precise knowledge about biological and behavioral systems engaged in decision-making is the most efficient and rational approach is the adaptive decision making (Yousefpour et al. 2012). As time passes, new and more reliable information becomes available about climate change. Making use of the new information and updating the adaptive decisions is a recognized procedure in conservation biology and used in management of threatened species, e.g. the Tasmanian devil (*Sarcophilus harrisii*) (McDonald-Madden et al. 2010; Probert et al. 2010). These experiences were utilized in this chapter in studying adaptive forest management in response to climate change. The novelty of this study relies on the multi-dimensionality of climate change as a stochastic phenomenon and the effective beliefs at decision points based on the ultimate and past knowledge. We make use of the updated knowledge and beliefs in the process of decision-making, which is sensitive to the behaviour of decision-makers and subject to goals and considerations.

Beliefs of individual decision-makers about the realization of climate change increase the complexity of the decision-making process (Bleda and Shackley 2008; Blennow and Persson 2009). We found that the determinist and active-adaptive decision-makers showed different behaviour in perceiving climate change. This was even the case when they received the same information about climate change as a multi-dimensional physical phenomenon. Efficient beliefs at decision points may, however, differ from the recent knowledge and belief. This is because the memory about the past observations effect on the beliefs about climate change (Bleda and Shackley 2008). Therefore, making use of accumulated knowledge to update beliefs is an important element to be incorporated into the models for the adaptive decision-making process (Bleda and Shackley 2008; Blennow and Persson 2009; Yousefpour et al. 2012). The updating procedure is widely developed in the probability theory and applied in Bayesian statistics (e.g. Prato 2008; Probert et al. 2010; McDonald-Madden et al. 2010; Ticehurst et al. 2011). Therefore, we adapted the Bayes' theorem (Bayes and Price 1763) in order to model adaptive forest management in response to climate change and the behaviour of decision-makers.

Decisions on adaptive actions may be diversified in two different levels. *First*, decisions should be adapted to the spatial and structural properties of the biological system under management, including actions at local and in regional scales. *Second*, decisions should be reassessed and necessarily switched to new actions at each decision point, if new knowledge and beliefs about climate change are available. Taking into account the economics of adaptive policies in forest management, Zhou et al. (2008) recalled that the post-harvest state and the timing of harvest depend on the states of stands and markets at the time of decision. Therefore, we revisited the decisions at the second decision stage 2020 and for individual ownerships.

Adaptive decisions made at regional level may yield sub-optimal decisions for local applications as found by Yousefpour (2009). Adaptation actions are indispensable not only because of climatic changes but also because of the evolution of the biological system and the need to adapt the management action accordingly. This is the reason why both determinist and adaptive decision-makers at 2020 choose the same actions under different assumptions about the actual climate realization (Table 13.3). Blennow and Persson (2009) concluded that seeing local adaptations to climate change leaves no explicit role for the strength of personal beliefs in climate change and adaptive capacity. Recently, Tompkins and Eakin (2012) emphasized that a neglected point in the analysis of adaptation actions is the relationships between the providers and beneficiaries of adaptation services. This is especially the case dealing with public goods. They concluded that there is a need for mechanisms of governance coupled with knowledge processes to identify the public adaptation needs and to orchestrate incentives, rewards and penalties for individual actions.

The quantification of adaptive decision-making concerns modeling the dynamics of biological systems by growth models, physical projections of climate change realizations, updating knowledge and beliefs about the actual climate state, behavioural economics of decision-makers, and subject to certain goals and considerations. This may become more complex, if forest mitigation actions and the production of bioenergy are added to the adaptation measures (Yousefpour and Hanewinkel 2009). For example, Yousefpour (2012) found that maximizing bio-energy production from Norway spruce plantations in Denmark may severely degrade forest soils. Therefore, future studies may focus on deriving balanced decisions to fulfill multiple purposes and think through sustainability criteria. If there are great trade-offs among management goals, it is necessary to diversify management actions in a forest area or distribute the goals among ecosystem parcels. Thus, it is recommended to make decisions locally to avoid suboptimal utilizations.

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Chapter 14
Economic Competitiveness of Forest Biomass Energy

Sepul K. Barua and Renato Bonilha

Abstract This chapter analyzes the economic competitiveness of energy based on forest biomass in comparison with energy based on fossil fuels. Under current market conditions, forest biomass is not cost competitive with fossil fuels. Improving technologies regarding production and energy conversion could make forest biomass economically more attractive. In the policy sphere, incentivizing the production of forest biomass, its energy conversion and use, as well as taxing fossil fuels for carbon emissions, could also improve the competitiveness of this renewable resource. The long-term prospects of energy based on forest biomass, including policy measures such as carbon taxation, are still highly uncertain, mainly due to large uncertainty in the future developments of carbon prices. Moreover, accounting for the carbon emissions of energy based on forest biomass itself could further restrict the effect of such measures.

Keywords Carbon neutrality · Cost improvement · Economic incentives · Forest biomass · Fossil fuel

14.1 Forest-based Biomass—an Option for Substituting Fossil Fuels in Energy Production

Forest-based bioenergy, mainly in the form of fuel-wood or charcoal, held a dominant position together with animal power as the main source of energy in the pre-industrial era. Wood and its derivatives provided energy in the form of heat, while animals were the main source of mechanical power. The heat was used to meet domestic needs, such as warming of the living environment and cooking, as well as to dry grains and manufacture goods such as glass and iron. It was not until the industrial revolution

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that fossil fuels became the dominant source of energy around the globe. Since then, coal and other fossil sources have dominated the energy sector as cheap and flexible options for most of our energy needs, promoting economic growth and social development. This crucial position taken by fossil fuels has led to increasing concerns over social dependency on non-renewable resources that are unevenly distributed around the world. These concerns were materialized during the oil crisis of 1970s' which had a major negative impact on the global economy.

A more recent source of emerging concerns about the unrestricted use of fossil fuels is the increasing levels of greenhouse gases (GHGs) in the atmosphere and the effect they are having on the global climate. Fossil fuels are a major source of GHG emissions, making reactive substances that could otherwise be trapped underground in a stable form available in the atmosphere. According to the Intergovernmental Panel on Climate Change (IPCC), these changes in climate may overwhelm natural processes of adaptation and jeopardize the livelihood of certain life forms, including humans, in different parts of the globe (Parry et al. 2007). The 1997 Kyoto Protocol, recently extended until 2020, marked the initial global efforts to mitigate and adapt human interaction in order to minimize the impact of global climate change in the future (World Bank 2009).

Aligned with efforts to minimize the threats imposed by fossil fuels, countries have been promoting the development and use of alternative renewable sources of energy.[1] According to the latest global status report from the Renewable Energy Policy Network for the Twenty-First Century (REN21), renewables accounted for 16.7 % of global energy in 2010. Biomass accounts for over 10 % of the global primary energy supply. The majority of the biomass energy is still used in heating, directly burned in an inefficient way, and only about 14 % of the biomass energy comes from combined heat and power (CHP) stations and liquid biofuels (REN21 2012).

Biomass may come from different sources, such as waste material, crop residues and by-products, energy crops and forests. Forest biomass mainly supplies the raw material for direct burning and the manufacture of solid fuels such as charcoal and pellets, despite efforts to produce second-generation biofuels. Different harvesting systems can be used to obtain biomass from forests. A considerable part of forest biomass destined for energy comes from thinning operations (see, e.g., Petty and Kärhä 2012). The choice of species and management are crucial to ensure the supply of this raw material and to improve its cost competitiveness when compared with other sources of energy[2]. The long-term sustainability of the model must also be considered in order to turn forest-based biomass into an economically sound alternative source of energy (Helmisaari 2011; Wall and Hytönen 2011).

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[1] Renewable energy includes a heterogeneous bundle of technologies that use resources such as biomass, wind, solar energy, geothermal heat, tidal power, wave power and other hydraulic power sources to produce energy in the form of electricity, heat, or fuels to be stored and used for power later.

[2] Interested readers are referred to Part 1 of this book to explore these issues further.

This chapter focuses on the economic competitiveness of forest energy biomass in comparison to fossil fuels based on a literature review. More specifically, it analyzes the economic and policy aspects of promoting the production and use of forest-based biomass for energy purposes. We concentrate on the economics of forest biomass energy and different potential measures to improve the economic competitiveness of forest energy biomass. In addition, the carbon neutrality of forest biomass in energy production, its policy implications, and the long-term prospect of forest biomass energy are discussed. The geographic scope of the chapter is Europe in general and Scandinavia in particular. Nevertheless, examples from the elsewhere are also given wherever relevant.

14.2 Economics of Forest Biomass Energy

14.2.1 Economic Competitiveness

The economic competitiveness of energy based on forest biomass, compared with fossil fuels such as coal, is impaired because of its production costs[3]. This holds true for the two main markets of forest biomass energy, i.e. Europe (e.g. Lauri et al. 2012; Mäkelä et al. 2012; Petty and Kärhä 2012) and the USA (e.g. Gan and Mayfield 2007; Gan and Smith 2006a, b). Petty and Kärhä (2012) reported that the whole-tree chips produced in Finland from early thinning of forest stands under average harvesting conditions (i.e. average stem size of removed whole trees is 0.4–0.5 m^3) is not economically profitable. Pellet production for the CHP plants is also not currently profitable in the European countries, especially in Finland (Mäkelä et al. 2012). Likewise, on average, logging residues may be the most cost efficient source of energy compared with biomass from energy plantations and forest thinning, but are nevertheless not economically competitive with fossil fuels such as coal (Table 14.1).

Efficient production of electricity from forest biomass is also still generally in its infancy and is not yet economically competitive. Gan and Smith (2006) reported that generating electricity from forest biomass would cost almost double that from fossil fuels, such as coal or natural gas. There are two main reasons for this cost difference. The first is related to the efficiency of energy production. Biomass fuel is more expensive than fossil fuel per unit of energy produced. The second reason concerns the investment cost. The initial cost for a biomass gasification system is almost 50 % higher than a conventional coal gasification system (EIA 2001). It can be noted here that the production of cellulose ethanol for electricity is not competitive, for example, with corn ethanol. However, cellulose ethanol is much more energy efficient than corn ethanol in terms of the energy output-input ratio throughout the life cycle of the product where the energy is used (Gan and Mayfield 2007).

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[3] These include the costs of producing or procuring forest biomass itself and converting it into energy.

Table 14.1 Comparison of energy production costs using forest biomass and coal (Sources: Adapted from Gan and Smith 2006a, b; USDA Forest Service 2005)

Biomass/energy source	Production cost	
	Biomass (€$^{-\mathrm{dryMg}}$)	Energy (€$^{-\mathrm{MWh}}$)
Short-rotation hybrid poplar plantation	38.91	8.08
Forest thinning	22.45–35.92[a]	4.64–7.41[a]
Logging residues[b]	20.95–24.69[c]	4.34–5.09[c]
Coal	–	*3.98*

[a] Depending on whether chipping is included in the thinning operation

[b] With a maximum transportation distance of about 100 km

[c] Depending on whether marginal or full cost method is applied (In the marginal cost method, only the additional cost from the conventional logging operation is counted as the bioenergy production cost. This method allows the sharing of the cost between timber harvesting and logging residue production. However, the full cost method allocates the total production cost across biomass and conventional wood products such as sawn timber and pulpwood (Gan and Mayfield 2007))

14.2.2 Measures to Improve Economic Competitiveness

14.2.2.1 General

Even though forest biomass energy is currently not competitive with fossil fuels, policy- and technology-based measures can improve the situation. Through a comprehensive review of literature (e.g., Gan and Smith 2006; ECF et al. 2010; Lauri et al. 2012; Petty and Kärhä 2012), we identify five specific types of measures for improving the economic competitiveness of forest biomass energy compared with fossil fuels: (1) incentivizing forest biomass production, (2) penalizing carbon emissions from the use of fossil fuels, (3) incentivizing energy production and use based on forest biomass, (4) reducing forest biomass production costs, and (5) improvement and innovation in biomass-energy conversion technology. The measures are thoroughly discussed below.

14.2.2.2 Incentivizing Forest Biomass Production

Policy incentives to improve competitiveness, such as subsidies, could help enhance the economic competitiveness of the production of forest biomass for energy. One such incentive programme is already in place in Finland. The Sustainable Silvicultural Foundation Law (Kemera, for short), was launched in Finland in 2008. Through the Kemera, about € 20 million worth of economic incentives are offered to private forest owners annually as subsidies for operations pertaining to the production of energy biomass from forests. The aim of this Kemera system is to encourage the recovery of small-diameter thinning wood for energy generation. The operations include thinning young stands, small-sized energy harvesting and chipping. The operations may also include harvesting stands on extremely poor sites. Petty and Kärhä (2012) reported that the Kemera subsidy makes the harvesting of young stands to supply energy biomass profitable in Finland. Moreover, the subsidy system ensures the efficient use of young stands by reducing the occurrence of young stands at an unintended level and thus enhancing the forest growth as well as timber quality and value for pulpwood and sawn wood in the long run[4]. Nordfjell et al. (2008) reported that Sweden, where no incentive programme like the Kemera is in place, is currently facing significant silvicultural problems with young, dense and small-diameter stands.

14.2.2.3 Penalizing Carbon Emissions of Fossil Fuel Use

The use of fossil fuels such as coal, natural gas and oil results in emissions of CO_2, the main GHG responsible for climate change. When the use of such fuels is penalized through, for example, carbon emission tax, these fuels become more expensive to consumers. Therefore, the tax narrows the price gap between biomass and fossil fuel energies, assuming that the former is carbon neutral and hence not taxed. Lauri et al. (2012) reported that forest biomass would not be competitive with fossil fuels in Europe unless the carbon emission tax is above € 20 per megagram (Mg) of CO_2 (Mg CO_2).[5] At a carbon tax below € $50^{-\mathrm{Mg\ CO_2}}$, mainly forest chips, recycled wood, bark and black liquor would be economically viable to use for energy instead of fossil fuels in the European countries. For example, forest chips and material wood would already start to compete with coal in heat and power generation at a carbon tax of € 30 per Mg CO_2. Lauri et al. (2012) also found that at a carbon tax of € 50 per Mg CO_2 or above, wood for industrial use such as round wood would start to become more profitable in heat and power production than in industrial uses. Moreover, wood for material uses such as house construction would constitute about one-third of the forest biomass used for heat and power generation in Europe with a carbon tax on fossil fuels at the rate of € 110 per Mg CO_2.[6]

14.2.2.4 Incentivizing Forest Biomass Energy Production and Use

Since fossil fuels result in carbon emissions, the use of forest biomass as a substitute for fossil fuels should avoid carbon emissions, assuming that biomass energy is carbon neutral. These avoided emissions could be measured and transformed into

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[4] Incentive programmes like Kemera could also encourage the harvesting of more stands at a young age, reducing the potential timber supply in future. This issue is, however, outside the scope of this chapter.

[5] Gan and Smith (2006) reported that at a carbon emission tax of about € 18.71 per Mg CO_2 emissions, biomass recovered from logging residues would be economically competitive with coal in electricity production in the USA. Biomass from hybrid poplar plantation would be competitive at a carbon emission tax between € 56.12 and € 93.54 per Mg CO_2.

[6] This would cause some adverse effects on the availability of wood for material and industrial uses, since the availability of wood is limited. Discussion of this issue is, however, outside the scope of this chapter.

carbon credits (Mayfield et al. 2007). These credits would become an additional source of income to forest owners for supplying harvesting residues and other sources of biomass that are used to produce electricity, or combined heat and power. Such credits could also be acquired by the energy plants that substitute fossil fuel with biomass for generating energy. Co-firing of coal and biomass is already cost-efficient in some places. The carbon crediting option for biomass use could make co-firing even more competitive and profitable.

14.2.2.5 Improving Forest Biomass Production Costs

Feedstock costs usually constitute almost half of the cost of producing energy from biomass. Therefore, driving the feedstock cost path down is the key to making biomass energy economically competitive with fossil fuels and other alternative energy sources. Studies (for example, Gan and Mayfield 2007; ECF et al. 2010) have identified the increase in scale of the biomass supply chain, improvements in feedstock productivity and harvesting and gathering techniques, as well as efficient transportation systems as the main factors for driving feedstock costs down in future. ECF et al. (2010) reported that the supply chain of biomass for energy in Europe could be scaled up by a factor of 10 to 100 in the future. This would significantly contribute to reducing feedstock costs in Europe.

The Scandinavian countries are developing cost-efficient systems to produce wood chips from forest residues. The application of a new harvesting system that piles or bundles the residues while harvesting could drastically reduce the marginal harvesting cost of producing biomass for energy from the forest residues in parallel with the biomass for other commercial uses (e.g. Laitila and Väätäinen 2011). These costs currently make up about 5 % of total biomass production costs in Scandinavia. Furthermore, this new harvesting method uses modified forwarders capable of removing both logs and residues[7] from the stand. This new system would reduce the present cost of forwarding residue to one-fifth of that of the traditional system. Transporting bundled or piled residues to the site of a bioenergy plant and chipping there instead of chipping at the roadside, as currently practised, would also bring the chipping cost down to half of the current level[8]. Moreover, the loading of forest residues for transportation to the site of the energy plant during harvesting is expected to become more efficient in the future in Scandinavia. This would reduce the cost of transporting forest harvest residues to the energy plant and thus the chipping cost (ECF et al. 2010).[9] The cost of production of energy from energy crop plantations could also be

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[7] Forwarders currently used in Scandinavia forward logs only.

[8] Note that this cost reduction estimate is adjusted for the fact that the higher moisture content of the residues slightly increases their transportation cost. Indeed, chipping at the plant was reported by Röser et al. (2011) to be the cheapest option of forest biomass energy production in Scotland, where conditions are not much different from those in Scandinavia.

[9] With the increase in demand for forest biomass, the stumpage price for forest residues could rise in the future. An estimate shows that the price could double in Finland and Sweden in the next-

significantly reduced, especially in establishing plantations and harvesting biomass (Ericksson et al. 2009).

14.2.2.6 Improving Biomass Energy Production

Co-firing woody biomass with coal may be a low-cost option for producing electricity and heat. Co-firing is already economically competitive with fossil fuel energy in some local markets in the USA (Nienow et al. 1995; Perlack et al. 1995). In Europe, it also seems to be an efficient option for producing energy from forest biomass. Currently, in all coal-based power plants in Europe, up to 5 % of the feedstock can be changed to forest biomass (i.e. with the remaining 95 % being coal) without any major investment or financial losses. Overall, about 50 to 90 terawatt hours (TWh) of energy, which is about 8–15 % of the energy currently produced in coal-fired plants, could be produced by biomass-coal co-firing without incurring any financial losses (Hansson et al. 2009).

To increase the use of forest biomass, some degree of conversion and thus additional investments are needed. In current market conditions, such investment costs could range from € 5 to € $10^{-\text{MWh}}$ in Europe. However, due to the availability of good quality fuel and efficient conversion technology, and thus lower conversion costs, the co-firing investment could be reduced to almost half (€ $3^{-\text{MWh}}$) of the current level by 2020 in Europe. Likewise, the additional operation costs of co-firing could also fall to just one-third of the current upper limit of € $3^{-\text{MWh}}$ (ECF et al. 2010).

In the case of energy plants fully dedicated to using biomass energy, the capital cost is currently € 20 to € $35^{-\text{MWh}}$. As more such plants are built, capital and operational costs are expected to decrease by 20 % in the near future (ECF et al. 2010). The complete conversion of coal-fired power plants to biomass fired plants is also an option, but the prospect of cost reduction is not as promising as co-firing and building completely new biomass power plants in Europe (see, for example, ECF et al. 2010).

14.3 Carbon Neutrality of Forest Biomass and its Policy Implications

Bio-energy is usually considered to be carbon neutral (see, e.g., CFS 2010). This stems from the understanding that the conversion of forest biomass to energy actually captures the carbon energy stored in the biomass itself throughout the biological cycle

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20 years if the historical price path is followed and demand continues to grow. However, the price of fossil fuels fluctuates from time to time for both economic and non-economic reasons (Gan and Smith 2006). The historic trend is upward. This could counter the increasing stumpage price, as the relative price rise for energy could be higher than the stumpage price. In addition, hauling costs could also increase with the demand for forest biomass, as harvesting would move to more remote forest areas (ECF; Sörda, Sveaskog and Vattenfall 2010). However, improving the efficiency of loading and transportation of forest residues could neutralize this increase in costs.

of a forest. Even though the conversion itself emits CO_2 and other GHGs, biomass energy replaces fossil fuels and thus helps avoid emissions from those fuels. Thus, even though forest biomass provides energy, the emissions that occur during the conversion are taken out of the atmosphere by the growing forests (CFS 2010). In other words, the neutrality of forest biomass energy is based on a backward looking perspective that the forest biomass being used for generating energy had actually been a forest growing in the past, thus releasing the same amount of carbon it stored before (Lundgren and Marklund 2012).

Policies in place (e.g. Lundgren and Marklund 2012) or being suggested (e.g. Gan and Smith 2006) for promoting forest biomass energy are based on the assumption of the carbon neutrality of bio-energy. For example, renewable energies that include forest biomass energy are exempt from taxes for both carbon and energy content in Sweden. This implies the carbon neutrality of renewable energies (Lundgren and Marklund 2012). The implicit assumption of the Kemera subsidy in Finland, which incentivizes forest biomass production through thinning and harvesting, is that forest biomass is a carbon-neutral source of energy.

However, a forward looking perspective instead of the current backward looking one implies that forest biomass is not at all readily carbon neutral (CFS 2010; Lundgren and Marklund 2012). From a forward looking perspective, CO_2 emitted during the conversion of biomass energy may take a long time to be recaptured by the new vegetation, depending on a number of factors. These factors include forest growth rates, as well as the type of fossil fuel being replaced, the conversion technology used and the alternative use and life cycle of forest biomass if not used for bioenergy (CFS 2010; Lundgren and Marklund 2012). The energy content of forest biomass is less than that in fossil fuels (CFS 2010; Law and Harman 2011), and thus carbon emissions per unit of energy produced are greater for bioenergy than in the case of fossil fuels. McKechnie et al. (2011) reported that it might take more than 100 years in Canadian conditions to neutralize the CO_2 and other GHGs emitted during the energy generation, depending on the type of forest biomass used and form of energy produced. The neutralization time in Scandinavia might be as long as that in Canada, since the forest growth rate and other determining factors are not so different between the two regions.

The time period that long-term policies cover is typically much shorter than 100 years; thus, bioenergy policies without full CO_2 accounting and understanding underlying carbon cycles and processes may actually cause more carbon emissions rather than reducing them (see, for example, Fargione et al. 2008). Thus, not only do the positive outcomes of forest bioenergy have to be assessed, but all carbon emissions also have to be considered, as discussed in Chap. 10. Otherwise, the climate impacts of using forest bioenergy may be overestimated, leading to unintended policy measures being taken (Lundgren and Marklund 2012). This is especially crucial for achieving the Europe 2020 Strategy target. Carbon emissions should be penalized, for example, through taxes. These measures will, however, increase the cost of bioenergy use, thus making it less competitive compared with other energies.

14.4 Long-term Prospect of Forest Biomass Energy

The European emissions trading scheme (EU-ETS) is probably the most mature mechanism of its kind in the world. It is being implemented in three phases and in 2013 it enters its third and final phase, which will extend until 2020; i.e., the year when the second commitment period of the Kyoto Protocol may also end[10]. In the first three-year phase of the EU-ETS, defined as the pilot phase, efforts were made to develop and establish a regional carbon market together with procedures for monitoring, reporting and validating. From 2008 until 2012 (Phase II), the focus was to expand the scope of the scheme and reach the targets set by the Kyoto Protocol. The third phase aims to strengthen the EU-ETS and achieve the targets set by the EU for emissions in 2020.

The legal framework behind the EU-ETS scheme aims to ensure the long-term effectiveness of the mechanism as a tool to amend the market failure exploited by fossil fuels and promote the use and establishment of cleaner technologies. One good example of this is the creation of a carbon floor price (CFP) in the United Kingdom that came into force in 1 April 2013. The objective is to determine a minimum price for carbon and mechanisms to collect and stream resources to top up the difference between the CFP and the market price, thus minimizing fluctuations and supporting the country's effort to reach the agreed carbon reduction targets (World Bank 2012). Long-term sustainability of renewable energy, more specifically biomass energy, depends on the success of mechanisms such as the EU-ETS, among others. In the following paragraphs we will review studies done in Europe and the United States assessing the long-term sustainability of the forest biomass for energy purposes, and its impact on the forest sector.

The Global Forest Products Model (GFPM) and the European Forest Institute Global Trade Model (EFI-GTM) are spatial partial equilibrium forest sector models with imperfect foresight that aim to maximize utility over a period of time. Both models are structurally similar and mainly differ in how they describe the global forest sector[11]. In a study conducted using the GFPM model, Raunikar et al. (2010) found that the increase in global production of fuel wood in the period between 2006 and 2060 is more than five-fold higher in scenario A1B from the IPCC (see Chap. 12 for description of the scenarios). The slower economic growth presented in scenario A2 resulted in a still significant increase of about 3 times in the use of fuel wood. According to the authors, both scenarios go against the past trend presented by most countries, which is a slight decrease in fuel-wood production. Pricewise, the scenario A1B would lead to a price convergence of fuel and industrial wood by 2025 at about € 64 m^{-3}. In the scenario A2, the same price conversion trend would follow, but at a lower rate. Buongiorno et al. (2011) reported similar results from analysing two scenarios, low and high fuel-wood demand, both based on the scenario

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[10] The second commitment period of the Kyoto Protocol could also be 2013–2017.

[11] Interested readers are referred to Buongiorno et al. (2003) and Kallio et al. (2004) for further information on the GFPM and EFI-GTM, respectively.

A1B from IPCC. They concluded that the increase in fuel-wood demand will lead to regional specialization in the manufacture of different wood products and a rise in wood prices due to increasing competition between different segments of the sector. Fuel and industrial wood prices will converge at a level between € 46 and € 58 m^{-3} in the period from 2020 to 2025, depending on the scenario.

Comparable studies using the EFI-GTM model have also estimated a rise in overall prices and increasing competition between fuel wood and other industrial uses of wood such as pulp. Moiseyev et al. (2011) stressed the uncertainty in the assumptions made by the model and relied on sensitivity analysis using different fuel-wood prices to evaluate the impact of the increase in energy biomass use promoted by the European Union's Renewable Energy Sources (EU-RES) targets in the region's forest sector. According to the study, forest biomass and related products are likely to represent less than one-third of the EU-RES target for 2020 and a significantly smaller share in 2030. Lauri et al. (2012) also stress the influence of carbon prices in improving the long-term competitiveness of forest biomass for energy and heat. A study by Kallio et al. (2011) used a simplified model, focusing only on the Finnish forest chips market. They examined the feasibility of the national forest biomass energy use targets used to achieve the 2020 GHG emission reductions. The study suggests that the carbon price range should be around € 25 to € $50^{-\mathrm{Mg\ CO_2}}$ to allow Finland to reach the 25 TWh target from forest chips.

According to the latest forecast of the UK Department of Environment and Climate Change (DECC 2012)[12], the carbon price in 2020 could range from € 0 to € $21.24^{-\mathrm{Mg\ CO_2}}$, with a mean value of € $10.48^{-\mathrm{Mg\ CO_2}}$, and the 2030 price could range from € 3.88 to € $30.55^{-\mathrm{Mg\ CO_2}}$, with a central value of € $15.07^{-\mathrm{Mg\ CO_2}}$. These prices could reach up to € $92.71^{-\mathrm{Mg\ CO_2}}$ when the CFP is included in the estimation. The development of carbon pricing discussed above suggests that the long-term competitiveness of forest biomass is achievable only when society is willing to accept the cost of carbon emissions and prevent a market failure. Otherwise, market-based measures do not seem to be sufficient. Another key issue is the fact that GHG emission prices not only increase the competitiveness of forest biomass against fossil fuels, but also potentially against other products supplied by the forest sector. The long-term impact on other segments of the forest sector cannot be neglected when assessing the use of forest biomass for energy purposes.

14.5 Conclusions

Forest biomass is an important source of renewable energy and thus an alternative to fossil fuels. However, under the current market conditions, the former is not economically competitive with the latter. The high costs of procuring forest biomass and

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[12] The values presented in this chapter were converted from GBP to EUR using the average exchange rate during a one-year period (14.11.2011–14.11.2012). The value was estimated using the European Central Bank (ECB) Internet site (http://goo.gl/VpJOm).

converting it to energy are responsible for this. These costs depend on the source of the biomass (energy crop plantations or forest residues) and how the biomass is produced (thinning or final harvesting), as well as how the cost of bioenergy production is shared in a harvesting operation. Considering all these, logging residues are found to be the cheapest source of forest biomass energy, but are still not competitive with cheaper fossil fuels such as coal.

However, there are a number of potential technology- and policy-based measures to improve the competitiveness of forest biomass energy. Improvement and innovation in technologies regarding forest biomass production and energy conversion could cut the production costs significantly (e.g. Thorsén et al. 2010). Moreover, proving incentives for forest biomass production such as subsidizing energy biomass thinning and harvesting, as well as taxing fossil fuels for carbon emissions, could make energy based on forest biomass economically more attractive. Incentivizing the production and use of bioenergy in substituting fossil fuels, for example through carbon crediting, could also do the same. However, given the trend in the carbon prices in the past years and the predicted future development of carbon price, carbon-based measures do not look very promising as a means to improve the economic competitiveness of forest biomass energy unless additional measures, such as the UK's CFP, are included in the mechanism.

Furthermore, bioenergy policies based on the assumption that such energy is carbon neutral may overestimate the positive impact of using forest biomass for energy production. This is because forest biomass energy is not readily carbon neutral, as carbon emitted during energy conversion could actually take over 100 years to neutralize. Recognizing this fact in policies could further jeopardize the long-term economic prospects of forest biomass energy.

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Part IV
Future Direction/Conclusions

Chapter 15
Conclusions

Seppo Kellomäki, Antti Kilpeläinen and Ashraful Alam

Abstract Mitigation of climate change is among the main priorities of global environmental policy. In forestry, the mitigation includes reducing sources and increasing sinks of greenhouse gases, especially those for carbon dioxide (CO_2). In this context, management of forests e.g., proper tree species choice, soil management, and control of spacing and rotation is among the keys to substantially increase the sink of CO_2 on forest biomass and soils. The use of forest biomass also provides a way to cycle carbon in the biosphere/atmosphere system, with an opportunity to reduce the use of carbon from fossil energy. The integrative management of forest for timber and energy biomass is probably a proper way to increase the supply of energy biomass to meet the future demand. However, large-scale and long-term use of forest biomass for energy are dependent on the sustainability in their production, economic competitiveness with other energy system, and future climate and policy measures, which may lead the decision-making even more complex.

Keywords Carbon neutral · Demand and supply · Energy biomass · Indirect emission · Life cycle assessment · Mitigation · Management · Nordic countries · Timber

The protection of the earth's climate with mitigation of climate change is among the main priorities of global environmental policy. Forests and forest ecosystems play a major role in the global carbon cycle in the atmosphere/biosphere interaction. According to Canadell and Raupach (2008), the terrestrial ecosystems, mainly forests, bind annually 3,000 million Mg of anthropogenic carbon (3 Pg C a^{-1}), which is 30 % of carbon dioxide (CO_2) emissions due to the use of fossil fuels and

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net reforestation. The potential of forest ecosystems in mitigating CO_2 emissions emphasizes, that 4,000 million ha of forest ecosystems (30 % of the global land area) hold an amount of carbon, which is double compared to that in the atmosphere.

Mitigation of climate change refers to any activities to reduce the growth of radiative forcing due to the increase of the concentration of greenhouses gases in the atmosphere. The mitigation includes reducing sources and increasing sinks of greenhouse gases, especially those for CO_2. Regarding forests and forestry, Canadell and Raupach (2008) identify four major strategies to mitigate carbon emissions: "(1) to increase forested land area through reforestation, (2) to increase the carbon density of existing forests at both stand and landscape scales, (3) to expand the use of forest products that sustainably replace fossil-fuel CO_2 emissions, and (4) to reduce emissions from deforestation and degradation." In this perspective, there are close links between the adaptation and mitigation; i.e. the increasing carbon sequestration through proper adaptive management will directly reduce the carbon emissions to atmosphere and thus store carbon in forest ecosystem. Canadell and Raupach (2008) emphasize that the large-scale use of forests to remove carbon from atmosphere should be considered in the context of sustainable development, where carbon mitigation benefits are balanced with environmental (e.g. loss of biodiversity, reduced stream flow) and societal (e.g. reduced food security) impacts, which large-scale reforestation may have.

In the Nordic countries, especially in Finland and Sweden, the use of forest biomass has increased rapidly. This process is much driven by the commitment of the European Union (EU) to reduce the CO_2 emissions in response to the Kyoto Protocol, thus mitigating climate change. Today, the main part of energy biomass is produced in forestry by harvesting logging residues and biomass in precommercial thinnings and tending of seedling stands. In Sweden, in particular, fast-growing willows are cultivated intensively on agricultural lands for energy purposes. Proper tree species choice, soil management, and control of spacing and rotation may substantially increase the potentials to produce biomass on forest lands as well. Sustainable and cost-efficient production of energy biomass on forest lands is possible through integrated management, where the production of timber and energy biomass is balanced in an optimal way.

The use of forest biomass provides a way to cycle carbon in the biosphere/atmosphere system, with an opportunity to reduce the use of carbon from fossil deposits outside the biosphere/atmosphere cycle. However, the energy based on forest biomass is carbon-neutral only, if the indirect emissions are excluded in assessing the climate forcing of forest-based energy. In fact, the indirect carbon emissions from ecosystem itself and the emissions related to the production, harvest and logistics of forest biomass for energy industry may reduce substantially the potentials, which the use of forest biomass may provide in mitigating climate change. However, forest-based bioenergy may be carbon-neutral if the uptake of carbon in forest growth exceeds the emissions (both direct and indirect) in different phases of biomass production chain from forest management to the combustion of biomass to energy. In this respect, the sink/source dynamics of carbon in forest ecosystems are among the main factors affecting the carbon neutrality of forest bioenergy. However, the carbon sink/source dynamics in forest ecosystems have only seldom included in the life cycle assessments and consequent calculations of climatic forcing, which the use of bioenergy may have. Life cycle assessments with integrated ecosystem and technosystem carbon flows and related time dynamics are needed to identify the management strategies for using forest-based biomass in the climate change mitigation.

Currently, the biomass (including forest biomass) accounts for over 10 % of the global primary energy supply. In the European scale, the current potential to produce woody biomass is about 1,000 million m^3 a^{-1}, of which about 700 million m^3 a^{-1} represent forest biomass. In the foreseeable future, the demand of forest biomass will substantially exceed the availability; i.e. in 2030 the demand is estimated to be slightly less than 1,400 million m^3 a^{-1}, but the potential only 1,100 million m^3 a^{-1} (Mantau et al. 2010). To meet the increasing demand, the harvest of existing biomass resources may be intensified, because more than 30 % of available biomass remains in forests. On the other hand, the forest management may be intensified by using fast-growing species and the best available techniques in establishment and managing forest plantations. Even in Northern Europe, the annual mean biomass growth of fast-growing species can be as high as 5–12 Mg ha^{-1} a^{-1} during rotations of less than 20–30 years. Furthermore, the production of energy biomass combined with the timber production can be substantially increased in a cost-efficiency way through intensive management, where the high yield of timber (pulp wood, saw logs) is the main priority in the management. In the large scale, the integrative management of forest for timber and energy biomass is probably a proper way to increase the supply of energy biomass to meet the future demand.

Until now, no major negative environmental impacts of intensive production of energy biomass have been found. This is especially the case, if energy plantations are established on abandoned agricultural lands or otherwise degraded lands. However, biomass harvesting is likely to change the biogeochemical nutrient cycles. Long-term studies show that harvesting of logging residues increases the removal of nitrogen and phosphorus compared to that in stem-only harvesting. The nutrient losses with the reduction in growth may be further enhanced if stumps are extracted for energy biomass, where the soil is disturbed much more than in stem-only harvest. Nevertheless, the stump extraction may improve the quality of energy biomass, but it is likely to reduce the carbon sequestration potentials of soils, which are the main carbon storage in the boreal forest ecosystems affecting the potentials to mitigate climate change in forestry.

In the integrative production of timber and energy biomass, the key questions are how to integrate the management efforts to enhance the production potentials, what are the sustainability issues to be considered in joint production (e.g. carbon and nutrient balances) and what are their overall economic implications for forestry. Under current market conditions, the main economic issue is high production costs, which undermines the competitiveness of forest-based energy systems compared to other energy systems. Long-term prospect of forest biomass energy and necessary policy measures for enhancing the use of bioenergy are still highly uncertain. All this makes the complex decision-making even more complex, including uncertainties and risks. Furthermore, the expected changes in climate increase uncertainties, which are met in assessing the role of forestry in mitigating climate change.

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Annex

Some Concepts Widely Used in Forestry and Forest Management in Producing Timber and Energy Biomass, with Impacts on Carbon Sink/Source Dynamics in Forest Ecosystems and Mitigation of Climate Change in Forestry

- Forest biomass: biomass in trees, including foliage, branches, stem, coarse roots and fine roots.
- Foliage: needles (coniferous) or leaves (deciduous) trees.
- Branches: lateral shoots of several hierarchical levels, to which needles or leaves are attached.
- Stem: the main axis of a tree, with upright position.
- Coarse roots: woody roots anchoring trees in the soil and leading water and nutrients into the vascular system of the stem.
- Fine roots: non-woody roots taking up water and nutrients from the soil.
- Stump: butt of the stem not harvested for timber, including tap and coarse lateral roots.
- Timber: stem wood fulfilling dimensions needed for saw timber and pulp wood used in forest industry.
- Energy biomass: tree biomass, excluding fine roots and stem wood used for timber.
- Carbon uptake: carbon fixed in photosynthesis and converted to different tissues in growth of biomass.
- Carbon emission: carbon released in living processes and the decomposition of organic matter with carbon.
- Carbon source: forest ecosystem taking up less carbon than it emits.
- Carbon sink: forest ecosystem taking up more carbon than it emits.
- Carbon sequestration: removal and storage of carbon from the atmosphere in carbon sinks in forests through photosynthesis.
- Carbon balance: balance between carbon fixation and emissions at a given moment or over a given period.
- Carbon storage (store): amount of carbon in the ecosystem or its compartments (e.g. above- or below-ground, trees, soil etc.) at a given moment or over a given period.

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- Litter: dead organic matter in soil profile with identifiable origin (e.g. needles, branches, stem).
- Humus: dead organic matter in soil profile with non-identifiable origin.
- Carbon cycle: taking up of carbon from the atmosphere, storing it in growth, and emitting it to the atmosphere in the autotrophic respiration of trees/plants and the decomposition of litter and soil organic matter in heterotrophic respiration.
- Substitution of fossil fuels: replacement of fossil fuels (liquid and/or solid ones) by fuels based on biomass.

Units and Conversions

In studies of forest ecology and forest management, the units of measurement may vary even within the same publication. This is the case, especially in older papers dating back to the time before standards provided by the SI system. Therefore, the tables below list a selection of dimensions frequently used in forestry that apply to the production of biomass and energy based on forest biomass. They are used in this book, and they provide help to convert some previously-used dimensions into the SI system not used in older publications.

- **Prefixes and Units**

Prefix in SI system	Explanation	Value
T	Tera	10^{12}
G	Giga	10^{9}
M	Mega	10^{6}
k	kilo	10^{3}
h	hecto	10^{2}
d	deci	10^{-1}
c	centi	10^{-2}
m	milli	10^{-3}
µ	micro	10^{-6}
n	nano	10^{-9}

- **Energy**

Unit	Transformations
J	$1\ \text{J} = 1\ \text{N} \cdot \text{m} = 1\ \text{kg} \cdot \text{m}^2 \cdot \text{s}^{-2} = 1\ \text{W} \cdot \text{s} = 0.239\ \text{cal} = 10^7\ \text{erg}$
W · h	$1\ \text{W} \cdot \text{h} = 3.6\ \text{kW} \cdot \text{s} = 3.6\ \text{kJ} = 0.86\ \text{kcal}$
MJ	$1\ \text{MJ} = 0.278\ \text{kWh}$
cal	$1\ \text{cal} = 4.1868\ \text{J}$
kcal	$1\ \text{kcal} = 1.163\ \text{W} \cdot \text{h}$

- **Pressure**

Unit	Transformations
MPa	$1\ \text{MPa} = 10^6\ \text{Pa} = 10\ \text{bar}$
bar	$1\ \text{bar} = 10^5\ \text{N} \cdot \text{m}^{-2} = 10^5\ \text{Pa} = 100\ \text{J} \cdot \text{kg}^{-1} = 10^6\ \text{erg} \cdot \text{cm}^{-3}$
bar	$1\ \text{bar} = 750\ \text{Torr} = 0.9869\ \text{atm}$
atm	$1\ \text{atm} = 1.0132\ \text{bar} = 760\ \text{Torr}$

- **Amount and Concentration**

Unit	Transformations
Molarity	$mol \cdot kg^{-1}$ of liquid
ppm	1 ppm $= 10^{-6}$ $mol \cdot mol^{-1}$; 1 $\mu g \cdot g^{-1}$; 1 $\mu l \cdot l^{-1}$
ppb	1 ppb $= 10^{-9}$ $mol \cdot mol^{-1}$; 1 $ng \cdot g^{-1}$; 1 $nl \cdot l^{-1}$
ppm	1 ppm $CO_2 = 1.82\ mg \cdot m^{-3} = 41.6\ \mu mol \cdot m^{-3} = 0.101$ Pa (at the temperature of 20 °C and pressure 101.3 kPa)

- **Radiation and Energy**

Transformations
$1\ W \cdot m^{-2} = 1\ J \cdot m^{-2}\ s^{-1} = 31.53\ MJ \cdot m^{-2} \cdot a^{-1}$
1 mol photon $= 1.8 \cdot 10^5$ J (when λ 650 nm) … $2.7 \cdot 105$ J (when λ 450 nm)
$1\ cal \cdot cm^{-2} \cdot min^{-1} = 6.98 \cdot 10^2\ W \cdot m^{-2} = 6.98 \cdot 10^5\ erg \cdot cm^{-2} \cdot s^{-1}$
$1\ erg \cdot cm^{-2} \cdot s^{-1} = 1.43 \cdot 10^{-6}\ cal \cdot cm^{-2} \cdot min^{-1} = 10^{-3}\ W \cdot m^{-2}$

- **Gas Exchange**

Transformations
1 g CO_2 (exchange) $\approx$ 0.73 g O_2 (exchange) ($RQ[CO_2/O_2] = 1$)
1 g O_2 (exchange) $\approx$ 1.38 g CO_2 (exchange)
Diffusion $D_{CO_2} = 0.64\ D_{H_2O}$
Diffusion $D_{H_2O} = 1.56\ D_{CO_2}$
$0.03\ \%_{vol}\ CO_2 = 300\ \mu \cdot l^{-1} = 282\ \mu bar = 28$ Pa CO_2 (partial preasure)
$1\ \mu l \cdot l^{-1} = 1.963\ \mu g\ CO_2 \cdot l^{-1}$ (at preasure of 1013 mbar and temperature 0 °C)
$1\ mg\ CO_2 \cdot dm^{-2} \cdot h^{-1} = 0.028\ mg\ CO_2 \cdot m^{-2} \cdot s^{-1} = 0.63\ \mu mol\ CO_2 \cdot m^{-2}\ s^{-1}$
$1\ mg\ CO_2 \cdot m^{-2} \cdot s^{-1} = 36\ mg\ CO_2 \cdot dm^{-2} \cdot h^{-1} = 22.7\ \mu mol\ CO_2 \cdot m^{-2} \cdot s^{-1}$
$1\ \mu mol\ CO_2 \cdot m^{-2} \cdot s^{-1} = 0.044\ mg\ CO_2 \cdot m^{-2} \cdot s^{-1} = 1.58\ mg\ CO_2 \cdot dm^{-2} \cdot h^{-1}$
$1\ mg\ H_2O \cdot dm^{-2} \cdot h^{-1} = 1.54\ \mu mol\ H_2O \cdot m^{-2} \cdot s^{-1}$
Conductance (at temperature of 20 °C and pressure 101.3 kPa)
$1\ cm \cdot s^{-1} \approx 0.416\ mol \cdot m^{-2} \cdot s^{-1}$
$1\ mol \cdot m^{-2} \cdot s^{-1} \approx 0.024\ mm \cdot s^{-1}$

- **Biomass**

Transformations
$1\ g\ DM \cdot m^{-2} = 10^{-2}\ Mg \cdot ha^{-1}$
1 g DM $\approx$ 0.42–0.51 g C $\approx$ 1.5–1.7 g CO_2
1 g C $\approx$ 2–2.22 g DM $\approx$ 3.1–3.4 g CO_2
1 g CO_2 $\approx$ 0.59–0.66 g DM $\approx$ 0.27–0.30 g C
1 g CO_2 = 3.67 g [= 44/12] C
Biomass = Volume [m^3] × Density of mass [$kg\ m^{-3}$]
Carbon content in stem wood [$kg\ C\ m^{-3}$]:
Scots pine 0.3091; Norway spruce 0.3715; birches 0.4152

DM Dry Mass

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