EFORWOOD
Tools for Sustainability Impact Assessment

Synthesis report on impact of forest management on environmental services

Raulund-Rasmussen, K., Hansen, K., Katzensteiner, K., Loustau, D., de Jong, J., Gundersen, P., Humphrey, J.W., Ravn, H.P. and Klimo, E.

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Preface

This report is a deliverable from the EU FP6 Integrated Project EFORWOOD – Tools for Sustainability Impact Assessment of the Forestry-Wood Chain. The main objective of EFORWOOD was to develop a tool for Sustainability Impact Assessment (SIA) of Forestry-Wood Chains (FWC) at various scales of geographic area and time perspective. A FWC is determined by economic, ecological, technical, political and social factors, and consists of a number of interconnected processes, from forest regeneration to the end-of-life scenarios of wood-based products. EFORWOOD produced, as an output, a tool, which allows for analysis of sustainability impacts of existing and future FWCs.

The European Forest Institute (EFI) kindly offered the EFORWOOD project consortium to publish relevant deliverables from the project in EFI Technical Reports. The reports published here are project deliverables/results produced over time during the fifty-two months (2005–2010) project period. The reports have not always been subject to a thorough review process and many of them are in the process of, or will be reworked into journal articles, etc. for publication elsewhere. Some of them are just published as a "front-page", the reason being that they might contain restricted information. In case you are interested in one of these reports you may contact the corresponding organisation highlighted on the cover page.

Uppsala in November 2010

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EFORWOOD

Tools for Sustainability Impact Assessment

Instrument: IP

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by

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Abstract

This document synthesises the impact of forest management on environmental services from forests. Environmental services including biodiversity, soil quality, carbon stock and sequestration, water quality and water quantity are defined by sets of indicators. Focus is on impacts of the single forest management operations on indicators and of typical management alternatives (combinations of operations) on the services.

Key words

forest management operations; forest management alternatives; biodiversity; carbon sequestration; soil quality; water quality; water quantity.

1. Introduction

by

Raulund-Rasmussen, K., Hansen, K., Katzensteiner, K., Loustau, D., de Jong, J., Gundersen, P. Humphrey, J.W., Ravn, H.P. and Klimo, E.

Mankind has always utilised the forests. In the beginning, the use was concentrated to felling of single trees for firewood and simple constructions. Lateron, more dramatic use and management like burning as part of shifting cultivation systems became frequent. Today, management include several alternatives ranging from simple regulations and low intensity utilisation in the entire forest ecosystem to modern high intensity management including e.g. heavy use of machinery, use of introduced tree species or genetically improved plant material, site preparation like soil tillage, artificial drainage, fertilisation and use of pesticides. The management alternative in use is first of all determined by the purpose of forestry.

Often we demand several goods or services from the forests besides the primary woody products. Focus has been mainly on internal ecosystem services like biodiversity and soil quality and on external protective functions like carbon sequestration and water quality and quantity of the water leaving the forest ecosystem.

Research has documented that management operations cause more or less significant influence on structure, dynamics and functioning of the forest ecosystem and at the end on the services we demand from the forests. The entire and by man untouched forest ecosystem is the most relevant reference for evaluation of forest management operations effect. Today, such nature forest reserves are seldom in Europe and they are often protected, among other things to serve as a reference for research.

Our objective with this report is to synthesise the comprehensive knowledge on effects of forest management operations on environmental services from the forests and present it in the context of the EFORWOOD project (www.eforwood.com/). The report contains five central chapters about each of the services: biodiversity, soil quality, carbon sequestration, quantity of water leaving the system and quality of water leaving the system. A sixth chapter summarises and synthesises the effects of forest management operations across the services and examplifies possible trade-offs depending on the management objective.

The five handled services are defined and effects of forest management operations evaluated by use of the indicator approach (Ferris and Humphrey, 1999, McElhinny et al., 2006). The most central indicators used in our report are listed in Table 1.1 together with so-called MCPFE and EFORWOOD indicators (MCPFE, 2002, Ramensteiner et al., *in preb.*a,b). To a certain degree, the use of indicators offers the possibility to evaluate effects quantitatively. However, an evaluation of management effects also poses several challenges. Some operations have relatively short-term effects and are reversible whereas others have long-lasting legacy effects and are nearly irreversible. Likewise, some operations have local effects whereas others have wider effects outside the treated forested area. Large scale manipulation experiments including at least two operations are the ultimate approach for assessment of operation effects. Although a huge amount of such experiments are conducted the experimental approach is often completed by a generic approach relying on deduced knowledge.

The number of forest management operations in use in European forestry is high and the operations are diverse. In Table 1.2 we have listed and grouped the operations considered in this report. In principle, the operations are isolated and may be combined freely. In practice, however, the objectives of the forestry more or less determine a set of management operations. We call a set of operations for a forest management alternative. We define 5 alternatives in Table 1.3 and based on the examination of the operations effect on the services we synthesize the gross effect of each of the alternatives.

1.1. Methodology

This report contains both elements of a typical review and a synthesis. We do not intend to review the huge amount of literature in detail for all combinations of forest operations and impacts on indicators. Fortunately, more narrow and detailed reviews are available in the literature for many operations. On the other hand, there are also combinations lacking experimental investigations forcing us to use a generic approach. For the water quantity and carbon sequestration services some modelling exercises have been performed in order to quantify impacts of some operations. More precise descriptions will appear in each of the separate chapters.

Table 1.1. Overview and coupling of environmental services, specific indicators for use in this synthesis, MCPFE (2002) criteria and indicators, and Eforwood whole chain (Ramensteiner et al, in prep) and M2 indicators (Ramensteiner et al, in prep).

Environmental service	Specific (sub)indicators for use in this synthesis	MCPFE Criteria	MCPFE Indicators	EFORWOOD Indicators
Biodiversity	- Tree species - Forest continuity - Soil disturbance - Standing and laying coarse and fine woody debris - Presence of key habitats - Authentic water regime	- Maintenance , conservation and appropriate enhancement of biological diversity in forest ecosystems	4.1 Area of natural, seminatural, and protected forests 4.2 Threatened species 4.3 Proportion of land managed for conservation 4.4 Stands of at least 2 species 4.5 Proportion of forest land under regeneration	- Area of regeneration within even aged stands and uneven aged stands, classified by regeneration type - Threatened forest species (red list) - Area of protected forest land - Dead wood - Landscape-level spatial pattern of forest cover - Area of forest and other wooded land classified by number of tree species occurring and by forest type and by protection status
Water quality	-Concentraion of problematic elements and compounds in seepage water leaving the system (nitrate, DOC, aluminium, and heavy metals)	- Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water)	5.1 Protective forests – soil, water and other ecosystem functions	- Area of forest and other wooded land designated to prevent soil erosion, to preserve water resources, or to maintain other forest ecosystem functions, part of MCPFE Class “Protective Functions” - Emissions to water
Water quantity	- Evapotranspiration from the system - Runoff coefficient - Ground water recharge - Snow interception	- Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water)	5.1 Protective forests – soil, water and other ecosystem funcions	- Evapotranspiration and surplus of water (precipitation – evapotranspiration)
Soil quality	- Available nutrient stock - Compaction - Soil acidity - Soil erodability - Forest floor morphology, C/N ratio, and pH	- Maintenance of forest ecosystem health and vitality. - Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water)	2.2 Soil condition 5.1 Protective forests – soil, water and other ecosystem funcions	-Emissions to soil
Carbon sequestration	- Carbon in living biomass (above and below ground) Carbon in forest floor and mineral soil - Carbon loss due to leaching of - - DOC and sediments - Emission of gaseous compounds.	- Maintenance and appropriate enhancement of forest resources and contribution to global carbon cycles	1.4 Carbon Stock	- Green house gas balance - Carbon sequestration

Table 1.2. Forest management operations considered in the report.

Tree species choice	natural, potential natural, introduced
Regeneration	planting, sowing, natural regeneration
Site preparation	• physical manipulation, e.g. soil tillage, ditching • chemical treatments, e.g. fertilisation, liming • prescribed burning
Stand establishment	• weed control • pest control • game (deer or others) protection
Stand management and harvesting	• silvicultural regime, e.g. clear-cut system, continuous cover system • pruning • thinning • rotation length
Other management options (infrastructures?)	• road construction • fencing • other infrastructure • buffer zones
Risk of natural hazards promoted by management	• fire prevention • windthrow • avalanches

Table 1.3. Definition of the forest management alternatives suggested by Duncker et al. (2007) for an environmental services perspective.

Management alternatives		Description and management objective	Tree species Rules	Site management and cultivation rules	Harvest and stand management rules
Increasing degree of manipulation ↓	Forest nature reserve	No management at all, natural disturbances and succession is the driver of development. Reference for authenticity and biodiversity refuge.	Natural	Not applicable	Not applicable
	Close to nature forestry	Only minor management. Could also be called “Low intervention”.	Natural or adapted	Mostly natural regeneration without soil tillage. None or only exceptionally chemical or physical site manipulations.	Thinnings are extensive; final harvesting often according to target diameter
	Multifunctional forestry	An alternative defined by man characterised by inclusion of several considerations and goals, e.g. social, environmental and economic.	Often natural or adapted.	Cultivation might be artificial after site and soil preparations. Chemical manipulation like fertilisation and use of pesticides and other physical manipulation are seldom.	Thinnings and stand regulation are often performed. Rotation length are often lengthened because of environmental and social considerations.
	Intensive even-aged forestry	Focus mostly on saw-logs. Other products or externalities are of second priority.	Optimal according to production or purpose	No restrictions within legislation	No restrictions
	Wood-biomass production	Focus is only on production of lignin typically for energy or pulp. Often short rotation. Could be called lingo-culture or agriculture with trees.	No restrictions. Optimal according to purpose	No restrictions within legislation.	No restrictions. Often short rotation and none or few thinnings. Roots and stumps may be harvested as well.

2. The impact of forest management on biodiversity

by

Johnny de Jong, Jonathan W. Humphrey and Hans Peter Ravn

2.1. Introduction

Forests and other wooded land cover roughly 30 % of the total land area of Europe and deliver a wi range of social, economic and environmental benefits including key components of biodiversity (FAO, 2004; CEC, 2006; EEA, 2006). In recent decades, global and European agreements on sustainable multi-functional forestry have led to the development of policies and targets for the conservation and enhancement of forest biodiversity in Europe (CEC, 2006).

Under the 6th Environment Framework (CEC, 2001), member states in the European Union (EU-25) have agreed to halt the loss of biodiversity in Europe by 2010, and a recent review of progress in achieving this goal has indicated positive trends for forests (EEA, 2006). In particular, forest area is not decreasing, forests are growing older and thus more valuable for conservation, a high percentage of forest area in some countries has now received independent certification indicating that sustainable management is in place, and 25% of the forest area is now protected to retain biodiversity and landscape values. However, there is still a need to address issues such as the impact of habitat fragmentation, harvesting of old-growth forest, climate change and pressure for intensification of forest utilisation leading to simplification of forest biotopes in some countries (EEA, 2006).

One of the most difficult challenges faced by the forestry sector is to deliver improvements in the economic outputs (timber and other materials) from forests whilst not unduly compromising biodiversity (Angelstam et al., 2004). In addition, there is an increasing realisation that biodiversity conservation is unlikely to be achieved by pursuing a strategy that focuses solely on protecting small areas of key biotopes or the needs of a few priority species and targeting economic activity in other places (Andersson et al., 2004; Bruinderink et al., 2003; Watts et al., 2005).

To date there has been no attempt to study the impacts of forestry activities on sustainability across the whole Forest-Wood-Chain (FWC). In this section the impacts of FWC activities on biodiversity is reviewed in a European context. The aim is to present an evaluation and synthesis of the known effects of specific forest management operations (Table 1.2) on biodiversity and to rank the importance of the different effects pointing out the most important and influential management operations.

2.2. Concepts and indicators

The term “biodiversity” or biological diversity has been defined as “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic systems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems [*sic*]” (UN Environmental Programme, 1992). The term is frequently used in conservation discussions. In general terms the most important abiotic factors determining the distribution and abundance of different taxa across Europe are climate, soil type and geomorphologic conditions. Also biotic factors such as competition, predation, parasites, diseases etc are important (Krebs, 1985). In forests other important factors are the structural complexity (e.g. variation in vegetation structure, gaps, edges), occurrence of important substrates (old trees, big trees, dead wood etc.), and tree species composition (Esseen et al., 1992).

The variation of complexity, substrate and tree species composition is a result of different types of disturbances, and all together these factors create a big variation of forest biotopes in which different species are adapted to.

Here we define biotope as the type of environment, often described as a combination of vegetation type, tree species composition, structure (the physical features of the environment, e. g. open, semi-open or edges) and management, e.g. semi-natural pastures, old-growth blue-berry spruce forest, or open meadows. Sometimes “Habitat” is used as synonym to “Biotope”, but here we define “Habitat” as the range of environment in which a species occur (Krebs, 1985). Within the same type of biotope the tree-species composition, structure and management regime may vary, as well as the number of habitats for different species. Over recent years, attempts have been made to develop classification systems for forest biotopes (Larsson, 2001; Barbati et al., 2006) and these systems give a framework and context for understanding the impacts of forest management on biodiversity. However, not only qualities in the forest biotope but also the arrangement of these biotopes in the landscape are important for biodiversity (Angelstam, 1997).

In order to understand how to conserve biodiversity in practical terms it is important to recognise the need for specific goals. Normally, the goal is not to get as many species as possible, but to conserve the species occurring naturally (i.e. not introduced by humans). Total measure of biodiversity is scientifically interesting in order to understand biodiversity pattern, but it is not used for conservation purposes. Instead a number of different indicators of biodiversity have been suggested (Lindenmayer et al., 2000). In reality the conservation discussion mainly focuses on the red-listed species which often have very specific biotope, or substrate requirement. The underlying assumption is that if we focus on the more demanding red-listed species also all other species with more general requirement will be conserved.

Red-listed species are used both for assessing forest qualities and for evaluating management, or conservation methods. Instead of using all red-listed species a subset of species are used as indicators. The idea is to use a nested pattern, in which occurrence of one species indicates occurrence of many other species. There are many suggestions of indicator species, however there are only few examples of scientifically investigated nestedness pattern (Nilsson et al., 2001). Another problem is that occurrence of species, or abundance not always are good indicators e.g. if population viability or source-sink pattern is unknown (Van Horne, 1983). One type of indicator is the umbrella species (Simberloff, 1998), which means that a number of species have similar requirements of substrate or biotope complexity, even though there are no other ecological links between the species. Woodpeckers have been suggested as umbrella species (Martikainen et al., 1998), and as good indicators of naturalness of forests (Angelstam and Mikusinski, 1994).

Species identification is often a problem, and because of that many other types of indirect indicators are used, such as abundance of dead wood, tree-species composition, occurrence of specific substrates etc (Nilsson et al., 2001). One problem with these indirect measurements is that you also have to know how much quantity is needed of the specific substrate or biotope for species survival. Some threshold values have been suggested, e.g.20-40m^3/ha for abundance of dead wood in temperate conifer forest (Humphrey et al., 2004, de Jong et al., 2004) and area of suitable biotopes, but for most species we have no data on the limiting factors and threshold values. Further, the threshold value might vary within regions and during the seasons (Wiktander et al., 2001).

Another method for biodiversity assessment without making species surveys is to use Habitat Suitability Index (HSI). Instead of detailed knowledge of species occurrence this is based on the composition of habitats in the landscape (Angelstam et al., 2004). However, when the HSI is created detailed knowledge about habitat selection, habitat use, dispersal pattern and other factors for some indicator species occurring in the landscape must be known. In addition landscape structure is also of key importance for species survival (Andrén, 1994; Fahrig and Merriam, 1994; Villard et al., 1999). During the 1980s landscape ecology became a scientific discipline of its own with big influence on conservation biology. Habitat fragmentation has been pointed out as one of the most negative factors behind species extinction (Fahrig, 1997).

Habitat fragmentation contains two components affecting biodiversity: Habitat loss and isolation. Habitat loss means that the suitable area is decreasing and fewer individuals can use the resources. The population size decreases, and finally if the habitat loss continues the population will not be viable due to genetic or demographic factors. The habitat loss might also result in a patchy landscape. If these patches are small and isolated the population on each patch will go extinct, even though the total number of individuals in the landscape is big. Species survival depends on habitat area, habitat isolation, occurrence of migration routs through the matrix (corridors), and quality of the matrix. In the forest landscape it is obvious that clear-cuts creates a fragmented forest. However, it is important to remember that also other types of management create fragmented forests. Forests with high quality for biodiversity are islands in a well managed forest. In Sweden, for example, the forest area and the forest volume have increased considerably during the 20th century, but meanwhile the fragmentation has increased due to more intensive management. That fragmentation is a real problem has been demonstrated in many empirical studies (Saari et al., 1998; Komonen et al., 2000). On the other hand there are a number of species, including some species specialised on specific substrates of dead wood, which are able to disperse long distances or survive on clear-cuts with high abundance of dead wood and which not experience forest patches as real islands (Ås, 1993).

The dispersal capability varies a lot among different species. Species with low dispersal capability will be the first one affected by fragmentation. If low dispersal capability is combined with low persistence (i.e. low possibility to survive during critical periods) the extinction risk increases. However, also species with low dispersal capability are moving around, and by creating a good infrastructure species might survive and explore new areas with suitable habitat. Therefore it is relevant to talk about continuity of suitable habitats on the landscape level (Angelstam, 1997; Sverdrup-Thygeson and Lindenmayer, 2002).

Many studies, both empirical and theoretical, have demonstrated that extinction due to habitat loss and fragmentation often is a slow process (Hanski 2000). Until a certain limit populations of species will survive even though their habitats disappear. However, when this limit is passed the extinction might be rapid. To identify this threshold value of remaining habitat for species survival has been an important area of study in conservation biology (Fahrig, 2001). Several studies indicate that the probability of extinction increases dramatically when less than 10-30 % of the original habitat area remains (Andrén, 1994).

2.3. Forest Management – methods, approaches and effects on biodiversity

Forest management includes for example: clear-cutting, drainage, soil scarification, plantations, pre-commercial thinning and thinning. Often it results in even-aged monocultures. Long-term consequences of forest management in the landscape include decreasing areas of old-growth forest, decreasing number old trees, dead wood and other for biodiversity important structures (Linder and Östlund, 1998; Andersson and Östlund, 2004). However, the consequences of forest management on biodiversity can vary considerably depending on which methods are used, and in many cases forest management and species conservation can be combined.

2.3.1. Tree species choice and methods of regeneration

When the forest regenerates naturally, the next generation of trees is a result of the available seed sources and natural competition within and between species. When seeds or seedlings are planted man is involved in the selection. The traditions in European forestry on tree species selection are highly variable between the regions. In intensively driven forests where yield in cubic meters has priority, exotic genetic varieties or species is often the rule. Some tree species – e.g. Norway spruce, lodge pole pine and Sitka spruce – have been turned into the main tree species outside their natural vegetation zones. Even where broadleaved trees such as beech are the natural vegetation the seeds used for planting may have been selected from an exotic origin. In some areas of Europe where the former natural tree vegetation has been removed by man, there have been attempts to re-establish this vegetation sometimes using exotic species. For example, tree planting experiments in sub-arctic parts of Europe the tree species may have been collected on the southern hemisphere. E.g. *Nothophagus spp* on the Faroe Islands (Ødum, 1979). The change of tree species will affect biodiversity as well as the homogenous structure of the plantation. In general dense coniferous plantings will allow no vascular plants or other vegetation to survive on the forest floor and very few insect species will survive in these areas. Further on acidification of the soil will influence the microarthropod fauna. The number of insect species associated with various tree species has been analysed in several studies. The number of species of the major plant feeding orders of insects (Lepidoptera, Coleoptera and most groups of Hemiptera) associated with British trees is closely correlate with the number of records of their Quaternary remains (Southwood, 1961; Kennedy and Southwood, 1984). In Britain the highest number of insect species is found on oak, willow, birch and hawthorn, whereas in Russia the highest number is found on Pine (Southwood, 1961).

Diversifying the tree species composition of plantations can be extremely beneficial to biodiversity. For example in the UK, naturally created gaps in upland spruce forests are often colonised by broadleaved trees and mixed conifer/stands are becoming increasingly common (Humphrey et al., 1998; Mason, 2006). Increasing the broadleaved area and number of native broadleaves species in conifer plantations is generally beneficial to biodiversity (Patterson, 1993; Humphrey et al., 1998). The diversity of fungal (Humphrey et al., 2000) lichen and invertebrate communities (Humphrey et al., 1998) has been shown to increase in response to increasing broadleaves.

Intra-specific variation in different trees species may also be of importance for dependant diversity. For example genetically modifying trees for resistance to pests and diseases can impact on the value of that tree species as host for a variety of organisms (Carnus et al., 2006). Therefore when dealing with stands of site native species in particular, the conservation of biodiversity is often best served by using natural regeneration which helps to retain autochthonous genetic variability (Peterken, 1993).

2.3.2. Site preparation

2.3.2.1. Physical manipulations

The main site preparation methods used prior to both afforestation and reforestation are tillage, ploughing and scarification. Site preparation is important for several reasons, e.g. it has a negative effect on weeds competing with the planted seeds or seedlings, and exposed mineral soil around the new plant has a negative effect on the pine-weevil, *Hylobius abietis*. Scarification is beneficial for some vascular plants adapted to disturbances (Pykälä, 2004; Haeussler et al., 2002). The species composition, species richness and abundance of vascular plants are all affected. Haeussler et al. (2002) demonstrated that species richness of vascular plants peaked after moderately severe site treatment, and that the removal of soil organic layers resulted in a higher abundance of species regenerating from seeds. However, some other organisms are negatively affected. Bellocq et al. (2001) demonstrated that arthropod diversity declined with increasing post-harvest site disturbance especially collembolans and mites – which is important for keeping the soil fertile by making adventitious pore structure. Drainage of wet habitats such as peatland, fens and swamps has in the past led to loss of wetland biodiversity, e.g. in northern Scotland, planting on deep peat led to erosion and loss of habitat for wading birds (Lavers and Haines-Young, 1997). In the forest of Grib skov in Denmark, Rune (1997) documented extensive reduction in wet areas over the last 100 years with a dramatic change in the flora as a consequence.

2.3.2.2. Chemical treatments

The use of chemical control methods in forestry is in general limited in comparison to other growing systems (agriculture and horticulture). However where the rotation is intensive in time (energy forest, Christmas tree production) or space (nurseries) the pesticide usage is also intensive. Control methods always have side effects on non-target organisms. In forestry at large these side-effects are considered limited. The closer production and management methods resemble intensive agriculture the more we could expect the same negative consequences on biodiversity as known from e.g. agriculture. For example, the collembolan species *Folsomia quadrioculata* has in the Boxworth growing system experiment shown to be negatively correlated to more intensive pesticide usage (Greig-Smith, 1992). This species is abundant in forests soils and is essential for good soil structure.

2.3.2.3. Prescribed burning

In some regions, prescribed burning is used to reduce competition from vegetation on tree establishment. However, it can also have benefits for biodiversity. In former times wild fires were the most important factor affecting the abundance of dead wood in the northern boreal forests (Ehnström, 1997). Through prescribed burning it is possible to create more favourable conditions for the organisms especially adapted to the post-burning situation. This occurs where trees are left on clear-cuts before burning. Many rare and threatened insect species benefit from prescribed burning and burnt trees that it creates (Wikars, 1992). Also several bird species are favoured by the variation in the landscape created by fires (Dale, 1997). Mychorrhiza fungus has been shown to respond to fire by fructification (Vrålstad et al., 1998). Some species regarded as pests are also attracted to fire, e.g. the longhorn beetle *Monochamus sutor* and the wood wasp *Urocerus gigas* may cause economical damage on the wood. Also *Hylobius abietis* is attracted to burned areas. The fungal pathogen *Rhizina undulata* gets virulent when exposed to temperatures 35-45°C (Petersen, 1971).

2.3.3. Stand management and harvesting

2.3.3.1. Forest management alternatives

The five management alternatives described in Table 1.3 all have different consequences for biodiversity as described below:

1. *Forest nature reserve.* Conservation of biodiversity is the key objectives for this type, which means leaving the stand without doing any management. Stands in which the organisms are adapted to natural disturbances (e.g. large areas in the boreal forests of northern Europe) that are left without any management very soon (within 30-40 years) develop high nature values, mainly due to increased amount of dead wood.
2. *Close to nature forestry*. In order to conserve biodiversity it has been suggested that forestry must mimic the consequences of natural disturbances (Bengtsson et al., 2000). This idea has partly been adopted by forestry and includes leaving some dead wood, big trees, small biotopes with high value for biodiversity (Larsson and Danell, 2001; Ferguson and Elkie, 2003) and the result have been evaluated in a number of studies (Väisänen et al., 1993; Esseen et al., 1997; Kaila et al., 1997; Hazell and Gustafsson; 1999; Vanha-Majamaa and Jalonen, 2001; Hautala et al., 2004; Ekbom et al., 2006). It is however obvious that it seems to be impossible to combine conservation of some species with economic sustainable forestry and this is an important reason for including a network of forest reserves within the managed landscape (Ökland, 1994, 1996; Niemelä, 1997). However there are also some stands in which the organisms are adapted to cultural disturbances (e.g. deciduous forest in the hemi-boreal or nemoral zones) the biodiversity might decrease without management intervention.
3. *Multi-purpose forestry***.** The purpose here is both to increase forest production and conserve biodiversity. This includes leaving all tree-stumps, leaving some old or big trees, leaving dead wood, in some regions it also mean to conserve a high proportion of deciduous trees. If the goal is to optimise forest production it is always negative to leave areas for free development except where thinning increases the risk of windthrow e.g. in north west Europe (Quine, 2000).
4. *Intensive even-aged forestry.* In this management alternative, the purpose is to create a monoculture with high forest production. Rotation lengths can be quite short (20-40 years) and harvesting takes place by clear-cutting. The consequences of this type of management for biodiversity is invariably negative at the stand level, except where the intensive plantation has replaced a land use of even lower biodiversity value such intensive agricultural fields (Humphrey, 2003).
5. *Wood-biomass production***.** This management altrenative is an extreme version of (4) where rotation lengths are even shorter. In most cases the cut trees are collected and used for bioenergy. This alternative is always negative for biodiversity compared to the other four options.

2.3.3.2. Silvicultural regime

In the management alternatives 2, 3, and 4, there are a range of different silvicultural regimes available to managers which have differential impacts on biodiversity. These are reviewed in general below.

2.3.3.3. Clear-cut system

Clear-cutting is the most dramatic change in the forest succession. The consequences of clear-cutting on biodiversity might be positive or negative. The result depends on which species or species group that are considered, and how the cutting has been carried out in relation to the natural disturbances in the area. For species adapted to old forest with small scale disturbances and long continuity of tree cover (e.g. many cryptogam species), clear-cutting results in habitat loss and fragmentation of remaining suitable habitat. However, some of these species are able to persist during the regeneration phase, and species with good dispersal ability are less affected. Species adapted to large-scale disturbances might benefit from clear-cutting provided that suitable habitat and substrate are created. This means that in some types of forest managed under the Close to nature management alternative, some clear-felling may be appropriate to conserve biodiversity (Quine et al., 1999). In some natural forest the fire is the main disturbance creating large areas of open forests. Some of the species, but not all, adapted to post-fire biotopes are able to survive on clear-cuts. In some cases, clear-cutting is combined with bio-fuel harvest. This will decrease the structural diversity at the site which decreases the possibilities for some ground living species to survive the open biotope succession phase (Åström et al., 2005).

Vascular plants is one example of a species group which is less affected by clear-cutting, or which even might benefit from clear-cutting. Early succession stages of forests are important for many plant species, and the abundance might increase considerably (Lindholm and Vasander, 1987; Humphrey et al., 2003). In a study of plant communities in Canada Haeussler et al. (2002) demonstrated that species richness was 30-35% higher 5-8 years after logging compared to the old forest. The result was confirmed in Finland by Pykälä (2004) who concluded that the number of species was almost double in clear cuts compared to mature herb-rich forests. As a consequence of increasing abundance of some herb species on clear-cuts several mammals benefit, such as rodents and cervids. Also some generalist predators such as red fox, wolves and lynx benefit from increasing abundance of rodents and cervids. Some of the most negatively affected species are the pine marten (Brainerd et al., 1995), squirrels and some species of bats (de Jong, 1995, Ekman and de Jong, 1996). However, most species of bats benefit from increasing edge-area. The response on bird species varies considerably. In short predators feeding on rodents or generalist predators are favoured by a more open landscape benefit by clear-cuttings (Petty, 1998). Also many other species common in the agricultural landscape associated with open or semi-open grassland and bushes are favoured by clear-cutting (Humphrey, et al., 2003), while species adapted to permanent tree cover or natural wildfire or water disturbances decrease. Bird species in the latter group are often non-migratory, e.g. wood-peckers (Mikusinski et al., 2001).

Amphibians are severely affected by clear-cutting. During some parts of the year amphibians are connected to water, but many species spend a lot of time in terrestrial biotopes. Several studies have demonstrated a total elimination of salamanders due to clear-cutting (Petranka et al., 1993; Petranka, 1994). In general many species of amphibians requires humid condition and occurrence of dead wood. However, by using adapted management near aquatic biotopes it might be possible to combine forestry with clear-cuts and conservation of amphibians. Invertebrates and cryptogams adapted to old-growth forest with natural disturbances, with high degree of specialisation, low dispersal ability and low persistence belongs to the most negatively affected species in the managed forest. Most of the red-listed species in forest belongs to this group and in general clear-cut is the main threat. Because of low dispersal ability fragmentation also affects some generalist arthropods such as spiders and ground living beetles (Miyashita et al., 1998; Abildsnes and Tømmerås, 2000).

One well studied consequence of clear-cutting is the edge effect. A new edge means new climatic conditions and interactions with new species for the species living in the forest. The result depends on the composition of the edge (structure and species composition) and sun and wind exposure. The increased wind exposure often results in higher abundance of dead wood near the edge which is positive for many species adapted to disturbances and using dead wood. One example is beetles of the family Scolytidae of which several species plays important roles in the boreal forest ecosystem (Weslien, 1992, 1994). Several other insect families are very abundant in the edge biotopes (Helle and Muona, 1985; Ferris and Carter, 2000), which also favour birds eating insects. In the boreal region many species of birds and mammals are attracted to edge biotopes (Hansson, 1994). However, for many other species edge-effects are mainly negative due to the climatic changes and increased competition (Spence et al., 1996; Esseen and Renhorn, 1998). For bryophytes it has been found that the climatic consequences are more dramatic for south-facing edges compared to north-facing edges (Hylander, 2005).

In the boreal region clear-cutting has been compared with fire disturbances. By mimicking post-fire biotopes as much as possible it might be possible to increase the species number on the clear cuts (Similä et al.; 2001). In North-America Reich et al. (2001) compared clear-cuts and wildfire areas of different succession phase and found no difference in plant species diversity. Even though there is a structural similarity, there are also some important differences (Delong and Tanner, 1996; Bergeron et al., 2002). A fire disturbance creates a lot of dead wood and a more varied structure and the ground cover is burned off (Bergeron, 2004; Harper et al., 2002, 2004). To leave some trees and biotopes on the clear cut does not completely compensate for fire disturbances, also restoration of biotopes by using fires is important (Niemelä, 1997).

2.3.3.4. Continuous cover systems

Continuous cover silvicultural systems encompass varying types of non-clear fell management (Mason et al., 1999), and include shelterwood, group, and selection systems (Mathews, 1999).

Small-scale cutting (e,g, small groups <0.25 ha or individual trees) is an alternative harvesting method to clear-cutting. At some sites it is difficult to regenerate the forest when using clear-cutting and thus selective cutting might be a solution. Selective cutting is also used for conservation reasons. Small gaps created by thinning and small group felling allows the development of larger trees and the provision of habitat conditions for a range of species group such as fungi, red squirrels, bryophytes and hole-nesting birds (Humphrey, 2005). This will benefit species requiring continuous tree-cover such as ectomycorrhizal fungi. These species are naturally adapted to low intensity fires of which most tree individuals survive, or to refuge biotopes (Dahlberg, 2002). However, in contrast Kropp and Albee (1996) demonstrated that thinning affects mycorrhizal fungi. The total number of species was reduced with some species negatively affected while others were positively affected.

Not only is continuous tree cover important, but so also is the continuity of other resources such as large trees and deadwood. Selective cutting regimes can be particularly valuable if some trees are left in perpetuity to create large diameter standing and fallen deadwood (Peterken et al., 1992; Humphrey et al., 2002; Humphrey, 2005).

In a study in Sweden, Bader et al. (1995) demonstrated that the consequences for wood-inhabiting fungi differ a lot depending on how the selective cutting is carried out. For these species the abundance and quality of dead wood in the new stand is most important. Among saproxylic beetles especially species living in hollow trees requires continuity of substrates (Nilsson and Baranowski, 1997). There are a number of other studies demonstrating the importance of continuity of different substrates which in general is rare in managed forests (e.g. Mycetolophidae, Ökland, 1994).

2.3.3.5. Thinning and pruning

All silvicultural regimes rely on some form of pre-commercial thinning and commercial thinning if the goal is to maintain or increase forest production. The methods adopted are also important for biodiversity. There are however, very few experimental studies on the consequences for biodiversity of thinning or cleaning. However there are many studies comparing biodiversity in old-growth forests and managed forests (Söderström, 1988; Gustafsson and Hallingbäck, 1988; Andersson and Hytteborn, 1991; Siitonen and Martikainen, 1994; Martikainen et al., 2000). One important difference between old-growth forests and managed forests is thinning. Old-growth forests have long continuity and natural disturbances which are prevented in managed forests). Pre-commercial thinning (or cleaning) and thinning change the composition and structure of the stand, and it has been shown in several studies that increased complexity and variation due to retention harvesting increase the conservation value (Work et al., 2003). In conservation mainly big trees have been in focus, however, for some invertebrates and fungi fine woody debris is very important (Kruys and Jonsson, 1999; Nordén et al., 2004).

The reason why big trees are such an important structure for many red-listed species is that this resource is rare. Concerning fine woody debris the situation is totally different. Even though some species are specialised on fine woody debris few of these are red-listed. Until now fine woody debris has not been a limiting resource. However, with increasing interest for bio-fuel there is a risk that also fine woody debris will be a limiting resource and that the number of species falling into the red-listed category will increase.

2.3.3.6. Rotation length and crop rotation

Often when a tree is harvested it is less than halfway through its natural life cycle. The senescence, dying and decay of a dead tree is omitted in rational forestry. The tree is cut when the annual increase in growth starts to decrease. However, most species living on wood use old trees, dying or dead trees (Samuelsson et al., 1994) and by removing the tree before it is biological mature the habitat for a number of organisms will be removed.

In many growing systems crop rotation is considered good management practise since negative influences by pests and diseases are diminished in this way. However this method is also negative for organisms of no economic importance. For example by interrupting a long continuity of beech forest by introducing Norway spruce the fungal flora will change dramatically (Flemming Rune, pers. comm.). Peterken et al. (1992) and Humphrey (2005) have recommended extending rotations in temperate plantations to benefit species associated with late successional forests such as lichens. In contrast there is also evidence that species can adapt to plantation forestry and ecologically short rotations. For example Humphrey et al. (2000) found that Sitka spruce plantations supported a range of red data book fungi.

2.3.4. Other management options

2.3.4.1. Road construction, fencing, other infrastructure

Transporting attacked and diseased wood out of the forest is an important countermeasure against pests. Consequently road building in forests is one of the most important actions to handle problems with pests and diseases. During the “battle against bark beetles” in Norway 1978-1980, most resources were dedicated to road construction (NOU, 1979). However, for other species with limited dispersal abilities roads, tracks and other infrastructures may act as barriers eliminating or limiting migration.

Jepson (1994) presented information on boundary permeabilities to certain ground beetles. He found that 3 m dirt road reduces permeability to 50%, 1 m gravel track reduces permeability to 15%, 0.5 m paved road reduces permeability to 23% and 5.7 m railway embankment reduces permeability 10-17% compared to the original biotope. Fry and Robson (1994) showed that even a 1m high hedge significantly reduced landscape permeability to butterflies. Wildlife bridges have been made to facilitate high-way crossing of deer, toads/frogs and dormouse, *Muscardinus avellanarius*. Wildlife fences are made to keep out deer from new plantations. Not only culture plants are influenced but also herb diversity will be different on the inside of the fence compared to the outside.

2.3.4.2. Management of edges, buffer zones and open space for biodiversity

Open space and edge habitats are key features for biodiversity in managed forests and can include areas of unimproved scrub, tree-line/montane scrub, grasslands, crags, bogs, heaths, and limestone pavements (Ferris and Carter, 2000). Wet areas, such as lakes, ponds, streams, fens, bogs and marshes provide valuable aquatic habitats (Lindemayer et al., 2006). Species suited to open ground and forest edges abound in some parts of Europe because of a long history of cultural management of ‘open’ forest with glades and rides, such as wood-pastures (Vera, 2000). Way-leaves, roads, rides, and thinning racks can form a network of open ground and edges to allow species movement inside and outside the forest (Ferris and Carter, 2000). Management of edges and open areas is usually required to maintain diversity; vegetation succession if left unchecked may lead to loss of valuable species. For example, many birds depend on the maintenance of a diverse edge structure (Fuller and Browne, 2003). Butterflies require nectar sources and food plant associated with edges and open areas (Tudor et al., 2004). Management actions can include: thinning ride edge trees and encouraging the natural regeneration of native shrubs/plants to increase nectar sources and produce a graded edge structure; maintaining the diversity of successional habitats such as scrub woodland (e.g. by flailing), grassland (e.g. by grazing/mowing) and heathland (e.g. by period burning); encourage mosaics or gradations of open ground and woodland for species such as black grouse (Cayford, 1993).

Buffer zones are important features in managed forests as they offer protection to sensitive biotopes such as aquatic ecosystems. Riparian areas form some of the most important buffer zones in managed forest (Potvin and Bertrand, 2004) as they form the interface between aquatic habitats and woodland. Riparian zones can have high biodiversity value as they contain a diversity of habitats and act as important corridors for the movement of wildlife (Petersen et al., 2004). Riparian woodland also has an important role in improving floodwater storage and providing resources for aquatic communities. The most important management consideration in the riparian zone is the density and distribution of trees, and therefore shading, and how this relates to natural bank features. Maintenance of bank processes and habitats supports a wide variety of wildlife. Some riverbanks may be relatively species-poor as a result of heavy shading by trees, for example densely planted non-native conifers. Riparian buffer zones are also prone to invasion by non-native invasive species such as Himalayan balsam (Hejda and Pyšek, 2004). These are often pioneer species able to thrive in dynamic habitats within riparian zones.

2.4. Synthesis

Forest biodiversity is a result of different types of natural and anthropogenic disturbances creating a high variability of habitat, structures and substrates. This variability is also related to soils, climate and biogeographic zone. The impact of FWC chain activities on biodiversity has to be evaluated in relation to biophysical context and the different forest types found in different parts of Europe. Thus it is difficult to generalise across Europe as a whole. Furthermore, the consequences of different management methods are difficult to generalise because it depends a lot on how the management is carried out, and different species respond in different ways to management.

However our review has highlighted some broad management issues which are important for biodiversity regardless of context. These issues relate quite closely to the five guiding principles of Lindenmayer et al. (2006) for biodiversity conservation in multi-purpose forests.

1. the maintenance of connectivity (i.e. addressing the effects of habitat fragmentation);
2. the maintenance of landscape heterogeneity;
3. the use of natural disturbance regimes to guide human disturbance regimes (i.e thinning and cutting regimes)
4. the maintenance of stand structural complexity (including taking account of the impact of associated specific stand management operations such as drainage; cultivation etc);
5. the maintenance of aquatic ecosystem integrity;

With regard to the sustainability of the Forestry Wood Chain it is important also to consider the specific role of the intensive forest management and lingo-culture production alternatives (4 and 5 in

Table 1.3), which are somewhat out with the envelope of multi-purpose forestry since there is an overriding focus on timber or biomass production.

2.4.1. Intensive forest management

As far as we know there are few examples of rare or threatened species which are able to survive in very intensively managed forest plantation systems. In general the number of species is very low and is dominated by a few species with general habitat requirements. Normally, at the stand level the conservation value of these areas is very low. On the landscape level the significance of intensively managed forest plantation depends on how large and area they cover and what kind of habitats have been removed. If they cover large areas and natural forest has been removed it might be a threat against many species. On the other hand if small areas are covered and mainly intensive agricultural fields are used for the plantation there are no threats against biodiversity. Some people even argue that intensively managed forestry plantation benefit biodiversity because by concentrating timber production to these areas other more natural forests can be less intensively managed.

2.4.2. Habitat loss, fragmentation and restoring habitat networks and connectivity

Some of the biggest threats to biodiversity are habitat loss, leading to fragmentation, leading to dispersal- and genetic problems. Examples of important habitat types in the natural forest which are lacking or fragmented in the present forestry landscape are uneven-aged stands, old-growth forests and forests with continuous tree cover. In general, species at most threat from habitat loss and fragmentation have very special habitat requirements and poor dispersal ability. These requirements can be difficult or even impossible to combine with forestry. Some examples are species demanding very high abundance of dead wood, old trees, big trees or specific natural disturbances. The only possibility to conserve these species is to conserve their habitat and the processes creating the right conditions. Normally this is only possible by strict protection through nature reserve or national parks. In order to conserve these types of species a network of such protected areas in the landscape covering different types of habitats is important. The area needed for protection depends on a number of factors: type of land-use in other parts of the landscape, the spatial distribution of high-quality forest patches, and the life-history of the species in the protected area.

The dispersal capabilities of threatened species can also be enhanced by management to increase landscape heterogeneity and improve the “permeability” of the matrix between habitat patches. Invariably the matrix forms the bulk of the managed forest and there is scope here for modifying management to improve hospitability of the matrix to red data species. For example, leaving legacies on clear-fells or doing variable density thinning. In theory, catering for red list species should ensure that wider diversity is also catered for.

Habitat restoration plays a key role in reversing fragmentation, but is only useful if some habitat qualities still exist and there are source of species in the landscape. Therefore restoration is most efficient near hot-spots and for connecting different hot-spots to each other. However, if the continuity of qualities is lost and the species is extinct, restoration is ineffective and it will take very long time until the species return (if ever). In order to find hot-spots many different types of indicators have been suggested. However, very few have been scientifically evaluated. To find suitable restoration level threshold values of the abundance of critical resources have been suggested, but so far too few examples exist to make this operational, and most examples are only valid in a specific group of species in a specific geographical situation. Further development and scientific evaluation is needed.

2.4.3. Landscape planning to improve heterogeneity and connectivity

There is no single prescription for a stand that will maximise biodiversity, different species groups have different requirements therefore landscape level solutions are required. Nor is it always possible to combine forestry for timber production and conservation on the stand level. In addition, many species are wide ranging and rely on threshold amounts, and/or specific spatial configurations of habitat at the landscape scale to ensure persistence over time.

In essence, to conserve biodiversity there must be a focus on creating and managing a landscape in which species can find suitable habitats and dispersal possibilities over a long-time frame. Forestry and other forms of land use alter the distribution and condition of habitats, but on the landscape level it is possible to conserve habitat and key structures whilst retaining economic productivity. Different tools can be combined such as: strict reserves, voluntarily reserves, conservation agreements, adapted management, and forestry with conservation considerations. Often some kind of economic compensation is paid to the landowner. In these examples also other values, such as cultural history and social values have been added. Basically there are two main approaches:

1. Establishment of large areas of national parks or nature reserves (often 20-30% of the land area) in which there is no forestry, combined with intensive management outside the reserves.
2. A combination of relatively small nature reserves and different types of extensive or intensive forestry. In order to create an ecological infrastructure the area of conserved habitats and general conservation considerations are related. The larger area that is conserved the smaller is the need for general conservation considerations and vice versa.

There is also some evidence to suggest that management regimes should be varied at the landscape scale simply to improve general structural heterogeneity. However, careful planning is required to ensure a balance between this aspiration and the need to ensure conservation of specific priority species and spatio-temporal connectivity of their habitats. One of the key considerations is the spatial and temporal arrangement of harvest units and the different types of silvicultural regimes (e.g. clear cutting versus low impact regimes; short rotations versus long rotations etc.). Natural disturbance regimes can offer a template for informing the spatial and temporal arrangement of different types of management.

2.4.4. Mimicking natural disturbance regimes

Both natural and anthropogenic disturbances have varied during history, and the result of this is a dynamic ecosystem, which can be very difficult to restore or conserve. Lack of management at the same time as natural disturbances are prevented (e.g. fire suppression in nature reserves) might result in a loss of biodiversity. However, as knowledge of the importance of natural disturbance for conserving biodiversity has increased there are an ever increasing range of examples where natural disturbance is being used to guide the spatial and temporal distribution of silvicultural regimes across the landscape as well as informing the creation of structural diversity within stands. In the former the general approach is to reflect a gradient of increasing severity and scale of disturbance, and to mimic different types of disturbance. In large landscapes this can lead to the use of prescribed burning of varying intensity in different parts of the landscape and a mix of clear- cutting and low intensity silviculture to reflect variability in the spatial distribution of stand-replacement disturbance regimes and gap-phase disturbance regimes.

2.4.5. The maintenance of stand structural complexity

By using some adapted forestry methods it seems to be possible to combine conservation with forestry for the majority of species including several of the red-listed species. Most important is to mimic the structures and habitat that naturally are created by natural disturbances, which means retaining key elements of stand structural complexity (Lindenmayer et al., 1996). These elements include structures such as dead wood, old trees, big trees, under storey vegetation etc. Structural complexity can be catered for throughout the “normal” crop rotation or by extending rotations. There is good evidence that clear-cutting combined with structural retention can cater for a wide range of different taxi. Similarly, variable density thinning can be used in young and middle-aged stands to improve structural diversity and spatial heterogeneity, and retain dead and dying trees. Smaller biotopes such as wet flushes, boggy areas etc can also be left undisturbed in larger stands whilst also carrying out stand management operations.

Measures to improve structural diversity within stands can affect economic yield, in that the most efficient harvesting method may not used and some trees are left etc. However, there are many examples from different areas in Europe showing that this kind of forestry can still be profitable. Often mechanised operations such as harvesting can be directed to avoid key biotopes and this can save time and money. For example, tree growth can be reduced in wet areas and hence the crop may be of reduced value and not worth harvesting.

Whilst the soil and vegetation disturbance associated with harvesting operations and cultivation prior to establishment can have a negative impact on both above and below ground biodiversity, these impacts can be relatively short-lived compared to more permanently damaging activities such as drainage or road building. Roads can be planned to avoid damaging key biotopes, and current forestry and nature conservation legislation offers protection for aquatic habitats in most countries.

2.4.6. The maintenance of aquatic ecosystem integrity

Lindenmayer et al. (2006) emphasize the importance of maintaining aquatic ecosystem integrity, and in our review we highlighted the importance of buffer strips for protecting aquatic ecosystems from the impact of forest operations. As mentioned in the previous section, aquatic ecosystems should be protected by legislation due to their importance in maintaining and delivering ecosystem services such as flood control, drinking water, fish production as well as biodiversity. However, careful inventory and planning is needed to ensure that the conservation of aquatic ecosystems is maintained alongside forestry operations.

2.4.7. Biodiversity indicators

The indicators selected by the EFORWOOD project for assessing the impacts of different forest management scenarios on biodiversity are listed in Table 1.1. Here we focus on the indicators in the second column in Table 1.1. The rationale for selection has been given in the document PD2.2.1 and can be summarized here. Tree species composition is measured in terms of the number of tree and shrub species. In general, as this number goes up, there is an increase in the number of different niches for dependent flora and fauna. This indicator should be of particular value in assessing changes to the Multifunctional forest management alternative (Table 1.3), where diversifying tree species is often a key objective of management. Forest continuity is an indicator of long-term provision of habitat required by a range of rare forest species. Continuity can be measured by the area and historical continuity of woodland key habitats (e.g. in the Close to nature forest management alternative) and also at the large scale by the overall area of forest nature reserve. Deadwood is a key indicator in all of the forest management alternatives except the wood-biomass alternative, where by definition there can be no deadwood. Presence of key habitat such as remnants of old-growth and wet forest is an important indicator in the Close to nature and Multi-functional forest types. Due to the statutory framework in most countries, authentic water regime (measured by changes in ditches and irrigation) is a key indicator in all the forest types in Table 1.3.

3. The impact of forest management on soil quality

by

Karin Hansen, Karsten Raulund-Rasmussen and Inge Stupak

3.1. Introduction

Forest soils serve multiple production and environmental functions. Maintaining and enhancing forest soil function is crucial if forestry should be practiced in a sustainable way in accordance to the Montreal and Helsinki processes (Schoenholtz et al., 2000; Moffat, 2003). Therefore, a growing interest in determining consequences of different forest management practices on plant productivity as well as on the soil quality (SQ) has emerged.

An increased demand for forest products, mainly timber and pulp wood, has led to a more mechanised forestry with shorter rotations, higher volume increment, site and soil manipulation, and species that are genetically improved for faster growth (Moffat, 1996; Johnston and Crossley Jr., 2002). These rapid advances in management practices have made modern silviculture and forestry operations more intensive but also more diverse than earlier in the 1900s.

Some forests are managed at low intensities, but a large part of European forest soils are highly manipulated by management practices. These practices comprise for example choice of tree species, site preparation, ditching and draining, fertilisation, liming, wood ash recycling, prescribed burning, use of herbicides, mulching, cleaning and harvesting. All such silvicultural choices and management operations have an impact on forest ecosystems. Acceleration of soil erosion, nutrient removal, compaction and changes in organic matter and soil water status are identified as the most important affected features (Haines et al., 1975; Vitousek and Matson, 1984; Worrell and Hampson, 1997; Johnston and Crossley Jr., 2002).

Negative impacts are often balanced by natural processes such as soil formation and nutrient input from weathering. A fundamental requirement for sustainable management is that the impacts of forestry operations should not exceed the natural long-term capacity of sites to restore themselves (Worrell and Hampson, 1997). Therefore, also SQ must be retained in the long-term (Power and Myers, 1989; Page-Dumroese et al., 2000). For example erosion losses should not exceed soil formation rates and nutrient removals should not exceed nutrient inputs by weathering and by deposition (Worrell and Hampson, 1997). Impacts that continue over a long time are more likely to exceed the self restoration capacity of the site than short-term impacts. Impacts therefore need to be aggregated over at least one tree rotation before significant statements regarding for example nutrient sustainability can be made (Worrell and Hampson, 1997).

Soils differ in vulnerability, resilience and recovery time. However, if the natural long-term self restoration capacity is exceeded, the impacts will cause soil degradation and lead to a reduction in SQ (Worrell and Hampson, 1997). The degree to which forest soils can be restored following degradation depends on the reversibility of individual impacts. Several impacts are reversible and recovery of the soil resource to a certain degree is possible either by introducing less intensive management and letting natural recovery processes act over long periods or by improved treatments (Worrell and Hampson, 1997).

3.2. Methods and approaches

In the EFORWOOD project, the aim is to predict a range of responses to impacts imposed by changes in forestry. This also includes quantification of SQ responses to such impacts. This chapter focuses on the impact of a range of individual forest management operations (listed in Table 1.2) on SQ in order to identify and illustrate which management options may influence SQ mostly and under which conditions. The main processes by which forest soils are influenced by management operations are identified and management-induced changes in SQ are quantified when possible. The chapter furthermore describes the nature and scale of changes to forest soils triggered by forest management practices. It is a position statement rather than an exhaustive review of these issues, and our analyses often build on existing data syntheses and reviews supplied with newer scientific research.

A second step after this analysis will aggregate the effect of the individual operations into evaluations of the impact of management alternatives (preliminarily defined in Table 1.3). This process will run in parallel with the further detailing of the management alternatives in EFORWOOD. However, a preliminary evaluation is presented in Chapter 7.

3.3. Concepts and indicators of soil quality

The concept of SQ has been described through a variety of definitions in the literature (Andrews et al., 2004). The simplest definition of SQ is ‘the capacity (of soils) to function’ (Karlen et al., 1997) or it may be defined as ‘the capacity to accept, store, and recycle water, nutrients, and energy, sustain biological productivity, maintain environmental quality, and promote plant and animal health’ (Doran and Parkin, 1994; Périé and Munson, 2000). A committee within the Soil Science Society of America presented an expanded version of this definition that defines SQ as ‘the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation’ (Karlen et al., 1997).

Hence, SQ is management- and ecosystem-dependent. It has been described as a vessel that contains what is assigned to it (Patzel et al., 2000). The evaluation of SQ can be based upon indicators of the soil function, which should be defined in terms of physical, chemical and biological properties and processes and assessed against a certain standard to determine whether a soil is being improved or degraded. The overall purpose is to develop a sensitive and dynamic way to document a soil’s condition and how it responds to management (Schoenholtz et al, 2000). Hence, the concept of SQ is relative, e.g. when soil measurements are considered of good quality for one purpose, they may be of poor quality when judged for another purpose. Goals for SQ therefore need to be established (Patzel et al., 2000).

3.3.1. Indicators

The need for quantitative tools and methods to assess and monitor SQ are apparent (Doran and Jones, 1996; Andrews et al., 2004). The important role of soil productivity to sustained forestry is clear, but which soil properties could be used as indicators for sustainable forest management? (Burger and Kelting, 1998; Staddon et al., 1999; Page-Dumroese et al., 2000). Karlen et al. (1997) along with them pose two questions that must be answered: i) how does the soil function and ii) what indicators are appropriate for making the evaluation of impacts to its function?

Indicators of SQ can be defined as quantifiable soil properties and processes that have the utmost sensitivity to changes in soil function. Thus, indicators are only useful if they can be linked to the concept of soil function, and if change in trends is likely to be detected in an appropriate time-scale (Moffat, 2003). Doran and Parkin (1996) emphasise that SQ indicators should correlate well with ecosystem processes, integrate soil properties and processes, be accessible to many users, responsive to management and climate, and, whenever possible, be parts of existing databases (Andrews et al., 2004). Especially, soil indicators responsive to variations in management are needed to compare the effects of management practice on soil through time. For example soil texture and soil depth change little through time, and are not useful for assessing management effects (Schoenholtz et al, 2000).

In literature, different indicators have been proposed through the last 15 years. For example, Karlen et al. (1997) as well as Schoenholtz et al. (2000) proposed a list of selected indicators of SQ. Among all proposed indicators, we choose to work with the following: available nutrient stocks, pH, acidification, the ratio between carbon and nitrogen in the soil (C/N ratio), bulk density and soil porosity. These all indicate the maintenance of the production capacity and nutritional sustainability (Table 1.1). Also, we will look at soil erosion and sediment transport, which point to the protective functions of the soil.

3.3.1.1. Available nutrient stocks – input output nutrient budgets

The nutrient balance is a crucial factor for the long-term productivity and change in nutrient stock can be an important indicator. Several forest management operations, e.g. intensive harvesting, soil preparation, and choice of tree species, have documented effects on the nutrient stocks (Adams et al., 2000; Watmough and Dillon, 2003). Changes in nutrient stock can be determined historically or retrospectively, e.g. by experimental assessment before and after a forest management operation. Another way to determine changes in nutrient stock is relying on estimates of all inputs to the system and all outputs from the system where after a balance is calculated, i.e. the nutrient balance approach or input-output nutrient budgets. The use of input-output budgets as a diagnostic tool for sustainable forest management is well described and used, e.g. by Ranger and Turpault (1999) and Raulund-Rasmussen et al. (2007). Inputs are elements entering to the system from outside sources, e.g. dissolved in precipitation, attached to dust or through fixation and fertilisation, whereas output are elements leaving the system through leaching, harvesting or evaporation. For all elements except N, important inputs also derive from chemical weathering of the soil minerals. If the input-output budget is balanced, the elements are neither accumulated nor depleted from the system. A positive budget means that the considered element is accumulated in the system while a negative budget suggests that export exceeds import and the system is unsustainable over time. Input-output nutrient budgets may give valuable quantitative information for recommendations applicable to forest management. Total soil N and phosphorous (P) concentrations are often cited elemental indicators of productivity.

The long rotation of forests causes a gradual shift in the distribution of nutrient pools and fluxes as the stand develops. This results in an increasing accumulation of nutrients in living biomass and detrital material (the return of nutrients from the plant to the mineral soil), and a decreasing dependence of trees on the mineral soil to meet annual requirements in favour of internal retranslocation and nutrient release through decomposition of the forest floor (Cole and Rapp, 1981; Miller, 1981; Johnson, 1985). Actual distribution and cycling patterns vary with nutrient, vegetation type and tree species (Cole and Rapp, 1981). Nutrient demands by trees are dynamic and change with time. Generally, nutrient requirements and soil uptake by trees are greatest before canopy closure and decrease in the later stages of stand development when nutrient uptake is primarily driven by wood increment. The shift in nutrient distribution away from the mineral soil as forest matures, and the greater dependence on organic matter decomposition for nutrient supply to trees can be reflected in an upward shift in fine root distribution with stand age and a decreasing dependency on the mineral soil with stand age (e.g. Grier et al., 1981; Schoenholtz et al., 2000). The relative role of SQ in growth and production thus varies with time and stand development (Miller, 1981; Nambiar, 1996).

A long-term prediction of the development in available nutrient stocks can, however, be difficult to perform. The crucial point is often the weathering rate which is notoriously difficult to determine although the input from weathering differs much between soils and often is the factor determining the vulnerability of the soil to management operations with nutritional impacts (Raulund-Rasmussen et al., 2007). The use of computer simulation models is a logical step to predict future changes and make a conceptualised representation of reality (Schoenholtz et al., 2000).

3.3.1.2. Soil acidity

Soils become acid when the acid production rate exceeds the acid neutralisation rate. The most important acid producing processes in most forest soils in humid climates are i) biomass growth and subsequent harvesting, ii) nitrification and subsequent leaching of nitrate and neutral base cations, iii) net production of organic acids during decomposition of litter, and iv) deposition of N and S due to air pollution and subsequent leaching of nitrate and sulphate together with neutral base cations, whereas soil mineral weathering is the only in principle source of acid neutralisation (van Bremen et al., 1983, Raulund-Rasmussen et al., 2007). Acid production related to decomposition of organic material may mainly show up as soil acidity in the organic matter rich soil horizons whereas acid production originating from growth and harvesting or acid deposition also may cause a decrease in base saturation or increase in soil acidity in mineral horizons. Leaching of DOC will also affect mineral horizons (Froberg et al., 2006).

At low pH, acidity produced by biological activity or deposited from the atmosphere is accumulated in the soil indicating an insufficient acid neutralisation by chemical weathering whereas a neutral pH indicate a sufficiently high weathering rate to compensate ongoing production of acidity in the system. Increase in soil acidity may indicate insufficient soil mineral weathering or too high acid production.

Forest management operations like harvesting and choice of tree species significantly accelerate natural increases in soil acidity due to growth which is why soil acidity might be a valuable indicator for SQ.

3.3.1.3. Bulk density and soil porosity

Compaction takes place as a consequence of harvesting traffic in the forest and on forest roads and skid trails. An estimate of the compaction degree can be made from measurement of bulk density and soil porosity.

3.3.1.4. Soil erosion and sediment transport

Site preparation like ploughing and draining, harvesting and construction of skid trails and forest roads influences rates of erosion and sedimentation of water bodies (Worrel and Hampson, 1997). Erosion is often a local phenomenon and it can be estimated as the difference in sediment yield before and after management operations. Damage to soils will occur when the soil formation rate is slower than the erosion rate.

3.4. Tree species choice

Individual tree species vary in their soil-forming impact. As Dokuchaev (1900), founder of soil science, put it: "… different vegetation gives rise to different soils". Plant-mediated characteristics such as litter quality and root structure contribute to the chemical composition and physical characteristics of the soil. Differences in effects among different tree species should therefore be expected due to differences in litter quality, the activity of earthworms, differences in canopy architecture and its interception of atmospheric deposition, differences in root structure and rates of nutrient uptake and growth (Miles, 1985). Early insights claimed that SQ can be preserved by correct selection of tree species and that species which lead to deterioration of the soil can be mixed in plantations with species that improve SQ. Augusto et al. (2002) made a relevant updated review on the impact of different tree species on SQ.

Tree species differ in their influence on nutrient flows and nutrient balances. Trees filter the atmosphere and capture gases and air-borne particles, mainly N and S compounds in industrialised regions, salt near oceans and dust particles near arid land. The deposition depends strongly on the canopy architecture, where height, leaf area index (LAI) and aerodynamic roughness length play important roles (Parker, 1983; Draaijers, 1993; Balsberg-Pahlsson and Bergkvist, 1995) as well as topographic position and the distance to the forest edge (Draaijers, 1993). Several studies have shown a much larger throughfall deposition to coniferous tree species than to deciduous tree species (Ulrich, 1983; Parker, 1983; Draaijers, 1993; Rothe et al., 2000; Kristensen et al., 2004; Rosenqvist et al., 2007).

Nitrogen fixation is the reduction of atmospheric N_2 to amino N, a process performed by certain soil microbes. Some tree species have developed symbiotic relationships with bacteria (e.g. *Rhizobium*) in root nodules or symbiosis with cyano bacterias (bluegreen algae) as in lichens. These microbes receive a supply of carbohydrates under anaerobic conditions and produce amino N for use by the tree. Nitrogen fixing trees like black and red alder as well as mountain pine have been used in silviculture as a tool to improve the soil fertility (Tarrant and Trappe, 1971). The annual symbiotic N-fixation can be substantial and it was estimated to be between 50 and 200 kg N ha^{-1} (Binkley et al., 1992, 1994; Bormann and DeBell, 1981) adding to the internal N pool. However, N-fixation is not considered a major issue in Europe where N-fixing species do not play a strong economic role in forestry.

Nutrients that are not taken up can be leached from the soil (Binkley, 1994). Studies comparing the output via water seepage in different tree species observed 2-4 times higher output of nutrients from Norway spruce than from beech (Lelong et al., 1990; Bergkvist and Folkeson, 1995; Fichter et al., 1998; Rothe et al., 2000).

The mineral weathering rate has an important impact on SQ. However, the mineral weathering flux is difficult to estimate and only few studies have compared the effect of different tree species on weathering and have shown that Norway spruce promote weathering of soil minerals and had a weathering rate which was 2-3 times higher than under species like beech, oak and birch (Lelong et al., 1990; Bergkvist and Folkeson, 1995; Fichter et al., 1998; Augusto et al., 2002). The mineral weathering rate is mainly influenced by soil pH and soil concentrations of dissolved organic carbon (DOC) (Drever, 1994; Raulund-Rasmussen et al., 1998) and such studies showed even 2-3 times more DOC under Norway spruce than under beech and oak (Raulund-Rasmussen et al., 1998; Augusto and Ranger, 2001).

pH has an effect on the mineral weathering rate, and different tree species have different effect on pH in soil. The difference in pH between different tree species could be as much as 1 pH unit but most often it was between 0.2-0.4 units (Augusto et al., 2002). A specific study by Pedersen and Bille-Hansen (1995) showed that deciduous tree species (oak and beech) had 0.2-0.4 pH units higher than coniferous tree species (Norway spruce and Sitka spruce) in the upper 15 cm of the soil after 30 years of growth on similar soils. The input of strong acids from air pollution accelerates the natural soil processes, lowers pH and increases the concentration of Aluminium (Al) and heavy metals in the soil solution. Whether or not a change in tree species will recapture these processes and cause a restoration of soils is uncertain. Based on their review of different tree species on SQ, Augusto et al. (2002) ranked tree species in order of acidifying ability as follows: (Norway spruce; Sitka spruce; Scots pine) ≥ (White spruce; Douglas-fir) ≥ (Birch, beech; oak) ≥ (ash; lime; maple). On this basis, they recommend that tree species with low acidifying impact will be planted on soils with low buffering capacity in areas of high atmospheric deposition and that acidifying tree species might be mixed with less acidifying tree species.

Early studies have shown that trees shed variable quantities of organic matter of different chemical composition (Bonnevie-Svendsen and Gjems, 1956; Bray and Gorham, 1964). Differences in litterfall quantity are generally smaller (between 3.5 and 4.0 t ha^{-1} yr^{-1}, reviewed by Augusto et al., 2002) than differences in litter quality (nutrient concentrations, ratios among nutrients and specific components of varying recalcitrance) (Binkley, 1994). Foliage of deciduous tree species has generally higher concentrations of N, potassium (K), calcium (Ca) and magnesium (Mg) than coniferous tree species and thus the chemical composition of litterfall can be richer in nutrients under deciduous tree species (Augusto et al., 2002). For N and P, the nutrient input via litterfall was 10-50% higher in deciduous than in coniferous tree species, while a difference of 100-400% was apparent for K, Ca and Mg in litterfall (Augusto et al., 2002).

Nutrients stored in the soil organic matter become available for the trees again through decomposition (or mineralisation), exchange reactions and dissolution. Soil communities exert strong influence on the processing of organic matter and nutrients. Earthworms have for example been shown to have a greater effect on carbon dioxide (CO_2) loss than for instance daily mixing or N addition (Wolters and Schaefer, 1993). Tree species have a strong effect on the composition of soil microbial communities, but little is known for comparisons. One study by Mardulyn et al. (1993) suggested less than half the N mineralisation and nitrification rate in the upper mineral soil of a Norway spruce forest compared to a beech forest. More studies point to deciduous forest stands having more bacteria and fungal biomass in the mineral soil than in the coniferous forests (reviewed by Binkley, 1994). Accumulation of nutrients in the forest floor also significantly differs among tree species indicating differences in the biodegradability of the litter (Klimo, 1992; Vesterdal and Raulund-Rasmussen, 1998). The decomposition activity of the micro flora and ultimately the turnover of nutrient are connected to the initial litter quality, light transmittance, air temperature and moisture. Light transmittance is for example negatively correlated with canopy cover and LAI and forest management through particularly the initial stand density and thinning intensity might lead to higher decomposition.

A classic belief is that conifers degrade soils while hardwoods improve them. Norway spruce is considered to be deteriorating the site while beech is characterised as the “mother of the forest” producing plenty of raw humus (Bonnevie-Svendsen and Gjems, 1956). When the input-output nutrient budget is considered for different tree species, it appears that deciduous tree species often have a balanced budget whereas coniferous tree species like Norway spruce have a negative balance (Bergkvist and Folkeson, 1995; Fichter et al., 1998; Augusto et al., 2002). The possibly larger loss of nutrients for the coniferous species, especially in regions with high N and sulphur (S) atmospheric deposition, is the background for recommending limited plantation of these species in regions with low nutrient stocks (Augusto et al., 2002).

3.5. Site preparation

Site preparation includes physical manipulation like soil preparation carried out before stand regeneration and ditching to improve the aeration of the soil as well as chemical manipulation like fertilisation, liming and ash recycling.

3.5.1. Physical manipulations

3.5.1.1. Soil preparation

Under natural circumstances forest soils are disturbed physically by e.g. wind throw (Peterken, 1977; Worrell and Hampson, 1997). During heavy storms windthrow can cause large disturbances of soil profiles. However, most disturbances are manmade and different types of mechanical methods have been invented mainly to improve survival of seedlings during forest regeneration. Patch scarification, trenching, mounding, and ploughing represent methods with increased intensity of disturbance of the soil. Patch scarification involves the inversion of patches of humus upside down to create spots to plant in, whereas the use of either mounding, trenching or ploughing always inverts, and in addition covers or mixes, humus with mineral soil. Deep ploughing is the preferred preparation for afforestation of former arable land since it represses weeds more efficiently than the other mechanical removal methods. Ploughing causes considerable disturbance to the soil profile since horizons are mixed and turned up and down in this way disrupting the pedogenic processes (Worrell and Hampson, 1997).

Soils mostly affected by erosion include sandy and loamy soils on steep slopes in upland sites and in high rainfall areas (Carling et al., 1993). Erosion is often a local phenomenon. Damage to soils will occur when the soil formation rate is slower than the erosion rate. Ploughing may cause erosion to increase (Worrel and Hampson, 1997) and it is reasonable to recommend the use of lower impact physical manipulations on susceptible sites in order to reduce erosion.

The amount of vegetation cover after soil preparation indicates the degree of soil disturbance. Matthesen and Kudahl (2001) compared the effect of different mechanical preparation methods on the vegetation cover of competing weeds and grasses in the first growing season on 3 afforestation sites in Denmark. Deep ploughing (down to 60 cm) reduced the cover percentage to approximately 50%, whereas trenching in between rows and agricultural ploughing (down to 20 cm) only reduced the cover 2-8%. Munson et al (1993) studied the growth of White pine and spruce after scarification and found that there was no significant effect although the nutrient reservoir in forest humus was removed in the treatment resulting in decreased nutrient availability in the mineral soil. Worrell (1996) ranked different soil preparation techniques according to the volume of soil disturbed: ploughing (350-850 m^3 ha^{-1}) > mounding (170-340 m^3 ha^{-1}) > scarification using a disc trencher (110-280 m^3 ha^{-1}).

Although soil preparation in the regeneration phase significantly improves growth and survival of seedlings a significant increase in leaching of nutrients may take place (Chapter 6). In a study by Burgess et al. (1995) scarification caused significant losses in nutrient capital which was thought to impact negatively on future forest productivity. Brand (1991) also noted a compaction of the mineral soil as a consequence of scarification after two years.

3.5.1.2. Ditching and drainage

Reclamation of peatland by artificial drainage has taken place all over Europe, however most intensively in Scandinavia and north-eastern Europe. Growth rates of natural trees on bogs or fens, or introduced forest cultures following the artificial drainage have often been improved substantially due to improved conditions for root growth and increased mineralisation of decomposing peat (Westman and Laiho, 2003). On peatland, artificial drainage therefore improves the available stock of nutrients and SQ in the short-term. However, often the increase in mineralisation rates may cause a surplus of available nutrients with significant leaching to groundwater or water streams as a consequence (Callesen et al., 1999: Gundersen et al., 2006; Prevost et al., 1999; Westman and Laiho, 2003). This may be problematic for water quality (Chapter 6) and biodiversity (Chapter 2). The positive effect on growth may be temporary and a continuous effect may depend on ditch cleaning or secondary ditching (Westman and Laiho, 2003; Åstrom, 2002). In the long-term, severe ditching may reduce the nutrient capital especially on bogs relying on nutrients from precipitation. In a review of a series of Finnish experiments, Westman and Laiho (2003) could, however, not find any signs of nutrient deficiency up to 75 year after the first ditching.

Ditching of insufficiently drained mineral soils e.g. due to tight argillic horizons are also practised but to a much lesser extent. If ditching is effectfull and rooting depth increase permanently, the available nutrient stock will increase. However, we have found no evidence in scientific literature for this statement.

Drainage may furthermore cause erosion to increase (Worrel and Hampson, 1997). Forest management operations that cause exposure of soil and provide obvious drainage channels will contribute most to erosion (Worrell and Hampson, 1997).

3.5.2. Chemical treatments

3.5.2.1. Fertilisation

Fertilisation will normally lead to an increase in the soil nutrient stock and in this way improve the capacity of the soil to produce and better the SQ. Nitrogen is the fertiliser element most commonly applied. There are two aims of fertilisation: i) to gain a short-term positive growth response especially ameliorating harvesting-related reductions in growth and ii) to secure SQ in the long-term especially securing the nutrient balance after organic matter removal through intensive harvesting.

Growth improvement as a result of fertilisation with N, P and K has been observed (e.g. Smith et al., 2000; Simcock et al., 2006). However, Smith et al. (2000) furthermore observed that fertiliser additions caused a significant decrease in foliar concentrations of all nutrients except for N. After fertilisation, Ballard (2000) observed a short-term rise in soil pH, which later turned to a long-term drop in soil pH. Both fertilisation and the presence of N-fixers caused a marked positive change in the concentration of soil N in the A horizon (Johnson and Curtis, 2001). However, Munson et al. (1993) and Périé and Munson (2000) observed no consistent effect of fertilisation on the soil nutrient availability in White spruce and pine, except for Ca.

Mostly fertilisation improves the nutrient status of the soil. However, if N is added alone or in too high concentrations, a negative impact on the balance of the other elements, mainly P and the base cations, might emerge, either due to a positive growth response or due to increased leaching (Gundersen et al., 2006). This effect may be relevant e.g when using biosolids derived from municipal sewage sludge or from mill residues as organic matter amendments, especially to less fertile soils (Stone and Powers, 1989; Johnston and Crossley, 2002). The content of N and P in these amendments is normally rather high, which in relation to N saturation and following base cation leaching may have serious negative effects on e.g. the soil Ca and Mg stock. The use of sewage sludge and mill wastes are furthermore controversial because of potential contaminants such as trace metallic or organic elements and will continue to be until all risks for accumulation in the ecosystem, transport to adjacent waters and transfer to humans have been attended to.

3.5.2.2. Liming

Positive growth effects of liming in agriculture have for more than 100 years inspired forest managers and researchers to look for the same gain in forestry. Many experiments has thus been performed in Northern- and Central Europe and elsewhere. Often no positive growth response has been observed. To our knowledge a thorough international review has not been done. However, a comprehensive review of Finnish liming experiments in Norway spruce and Scots pine showed negative growth response of liming when not followed by other fertilisation (Derome et al., 1986). In Sweden, growth effects of liming included both positive and negative short-term response (Popovic and Andersson, 1984). An immobilisation of N in high C/N-ratio raw humus was suggested to explain the negative effects.

Accelerated leaching of Ca and Mg following deposition of acidifying N and S compounds reinitiated many liming experiments in the 1980s and 1990s. Especially, Mg shortage initiated by acid deposition seems to be compensated by dolomitic liming (Kreutzer, 1995).

In principle, liming should improve important SQ indicators like soil pH, the Ca storage, and the Mg storage if dolomitic lime is used (Kreutzer, 1995). Absence of positive growth response could be caused by other growth factors like N availability or drought or by negative effects of liming neutralising or hiding a possible effect. Such negative effects might be leaching of N, negative microbial effects etc.

3.5.2.3. Wood ash recycling

With current practices, the increased use of forest fuels results in an intensified export of nutrients from the forest. A large part of the forest fuel consists of branches, tops and needles that were earlier left to decay in the forest. Although these fractions only amount to a small proportion of the total weight of the tree, they have a much higher nutrient concentration per unit weight than stems. Thus, the increase in nutrient export might be significant. Another undesired effect of the nutrient export is enhanced soil acidity. Returning of wood ash after incineration of wood has therefore become relevant. The principle aims of recycling of wood ash to the forest are to i) avoid depletion of essential soil nutrients and to ii) reduce the harmful effects of acidification of forest soils and adjacent waters (Aronsson and Ekelund, 2004).

The major components of wood ash are Ca, K, Mg, silicon (Si), Al, iron (Fe) and P (Nilsson & Timm, 1983; Steenari et al., 1999; Holmroos, 1993; Eriksson & Börjesson, 1991; Kofman, 1987). Ash is generally low in N and S because it is vaporised during combustion. Trace elements found in ash include arsenic (As), barium (Ba), boron (B), cadmium (Cd), copper (Cu), chromium (Cr), silver (Ag), molybdenum (Mo), mercury (Hg), nickel (Ni), vanadium (V) and zinc (Zn) (Booth et al., 1990). The amounts of toxic compounds in wood ash have been observed to variate greatly and due to different mobility of elements like Cd and caesium (Cs), caution must be made when wood ash is applied to forests. The nature of the raw material strongly influences the composition of the wood ash since ash derived from branches, foliage and bark is richer than stemwood in many elements (Pitman, 2006). Also, the quality of the wood ash is dependent on the tree species. In general, ash from deciduous tree species contains more K and P as well as higher proportions of macronutrients than ash from coniferous tree species but less Ca and Si and are therefore likely to be a more effective fertiliser (e.g. Werkelin et al., 2005; Pitman, 2006). Other factors that cause variation in the concentrations of elements in wood ash are the type of burner and incineration conditions as well as contamination of the fuel and storage conditions.

Wood ash neutralises acidity when it is dissolved in water. When applied to a soil it will raise the pH. The effect in the organic O horizon depends significantly on both the type of ash and the dose applied (Bramryd & Fransman, 1995; Saarsalmi et al., 2001; Levula et al., 2000; Moilanen & Issakainen, 2000). Raw ash gives the largest and most rapid pH increases and the higher the dose the higher the increase in pH. The effects of wood ash on the acidity of soils seem to last over a long period of time. Ash doses around 3-5 t ha^{-1} have been shown to elevate pH 1 to 2 pH units 10-19 years after application (Mälkönen, 1996; Moilanen & Issakainen, 2000; Bramryd & Fransman, 1995; Saarsalmi et al., 2001). The transport of alkalinity down through the profile is slow and the effects deeper in the profile are found to be small and usually only occurring a considerable time (>10 yrs) after the application of the ash (Bramryd & Fransman, 1995; Saarsalmi et al., 2001). Hence, an increase in the pH of the soil solution in mineral soils is not usually found (Ring et al., 1999; Arvidsson, 2001; Fransman & Nihlgård, 1995) except when high doses (>10 t ha^{-1}) have been applied (Kahl et al., 1996).

Naylor and Schmidt (1986) found that the availability of K was directly related to the applied amount whereas Callesen et al. (2007) found that the availability can be significantly reduced by hardening the ash. Also, the supply of both K and P from wood ash seemed to be lower than from commercial fertilisers (K: 65-70%; P: 28-70%) (Naylor and Schmidt, 1986; Pitman, 2006). Some elements in ash are quickly leached with the percolating soil solution. Elevated concentrations of K can be found in the soil solution at deeper levels shortly after the ash application while the leaching of Ca and Mg is slower (Rumpf et al., 2001; Arvidsson, 2001).

In a recent experiment where 8 t ha^{-1} wood ash was applied to a Norway spruce forest, the effect on soil and fine roots were followed (Brunner et al., 2004). An increase in soil exchangeable Ca and Mg and these elements in fine roots along with a decrease in Fe, Zn and Al in the soil exchangeable fraction were observed. Furthermore, pH increased from 3.2 to 4.8, base saturation increased from 30% to 86% and BC/Al ratio increased from 1.5 to 5.5. At the same time, no signs of increased concentrations of heavy metals in the soils were detected.

The addition of wood ash does not result in a significant tree growth increase or decrease in forest stands on mineral soils. The limiting factor for tree growth in stands on mineral soils is in most cases the availability of N (Tamm, 1991). As long as N remains the growth limiting nutrient, the addition of other nutrients will not increase growth. However, wood ash addition in forest stands on nutrient rich peat soils has shown a significant positive effect on tree growth (Ferm et al., 1992) and improved conditions for natural stand regeneration (Huikari, 1951; Lukkala, 1951; Lukkala, 1955). Peat soils deficient in K and P but with a good N status show the highest increase in tree growth (Silfverberg and Moilanen, 2000) while tree growth on peat soils low in N (<1%) remains low (Silfverberg and Huikari, 1985a; Silfverberg and Issakainen, 1987). An increase in productivity of 3-4 m^3 ha^{-1} over 55 years was found after wood ash applications of 5 t ha^{-1} to drained peat (Silfverberg, 1996; Korpilhati et al., 1999).

In order to decide for an optimum wood ash application, existing guidelines for liming could be looked at (Pitman, 2006) where base saturation target values are supplied. Andersson et al. (1995) recommended an application of wood ash to achieve a humus layer base saturation of 50% and a mineral soil base saturation of 20%. At the same time, it is necessary to apply different doses to different tree species since tree species such as beech, oak, Norway spruce, Scots pine and Douglas fir typically grow best with base saturation rates of 30% while hornbeam and sycamore prefer levels in excess of 50% (Pitman, 2006). An efficient form of ash has been shown to be in granulated form since it is easily spread and creates a slow release of chemical elements and thus reduces the risk of alkaline flushes through the forest soil (Pitman, 2006).

3.6. Forest stand establishment

Forest treatments that are performed in order to ensure successful establishment of the new forest include, e.g. prescribed burning and weed control in order to create a favourable environment that promotes fast and efficient establishment and good survival of seedlings. This is obtained by reducing the competing ground vegetation (e.g. grass and herbs).

3.6.1. Prescribed burning

Fire has long been recognized as a natural ecological driver. Wildfire is a natural disturbance in some systems, but fire can also be an effective management tool throughout much of the boreal forests in order to remove undesirable vegetation and slash (Murphy et al., 2006). At times, forestry depends on it for renewal (Scheuner et al., 2004). This is particularly true in cool temperate areas where decomposition of forest residue is slow, and needles, leaves, and logs accumulate on the forest floor. Fire reduces much of this organic material to mineral-rich ash, releasing and recycling nutrients, as well as creating openings where new forests become established (Weber and Flannigan, 1997; Cade-Menye et al., 2000; Scheuner et al., 2004). As a silvicultural tool, burning of clear-cut sites is widely used for site preparation and fuel hazard reduction (cf residue management). Further functions include controlling species composition and competing vegetation, as well as reducing insect and disease infestations (Wikars and Schimmel, 2001).

Prescribed burning are generally low intensity and low severity fires performed under controlled conditions when soil moisture content is moderate to high. This kind of fire is used for site preparation prior to seeding or planting and during the rotation to reduce woody competition and lower the risk of wildfire (Van Lear and Waldrop, 1991; Carter and Foster, 2004). They are intentionally ignited for the purpose of achieving a clearly defined management goal (Carter and Foster, 2004). Approximately 50% of the available forest floor and understory is consumed in a typical prescribed burning. Usually, prescribed burning begins after crown closure and is repeated every 2-5 years until rotation age. On the other hand, wildfire is never intentional and they often occur when it is warm and soil moisture is low. Often wildfires are therefore more severe.

The largest change of forest soils caused by fire is a removal of organic matter (Certini, 2005). Bulk density increases as a result of the collapse of the organo-mineral aggregates (Giovannini et al., 1988; Certini, 2005) and the clogging of soil pores by ash or freed clay minerals (Durgin and Vogelsang, 1984), which causes a decrease in the water holding capacity of soil (Boyer and Miller, 1994; Boix Fayos, 1997). The pH of the soil tends to increase after a fire due to hydrolysis of the base cation oxides, which are abundant in ash. In a clear-cutted black spruce stand, acidity of the humus layer was reduced by up to one pH unit after prescribed burning. The first wetting fronts after a fire are of extremely high pH (Grier and Cole, 1971). The magnitude and duration of pH-rise may be quite large for poorly buffered soils, but the response curve tends to be short and broad for soils rich in clay and/or organic matter (Ballard, 2000). In a prescribed burning in Sierra Nevada on fine-loamy soil, however, fire had no effect on soil solution pH and only a small effect on soil pH (Murphy et al., 2006).

Overall burning released nutrients that had been immobilised in the slash, surface vegetation and the organic matter, resulting in a substantial increase in some topsoil fertility parameters. The immediate response of soil organic N to fire is a decrease because of loss through volatilisation (Fischer and Binkley, 2000; Certini, 2005; Murphy et al, 2006). Fire can cause substantial elemental losses of N, C and S through volatilisation even at low temperatures (Ahlberg and Ahlgren, 1960; Kelsall et al., 1977; Woodmansee and Wallach, 1981; Caldwell et al., 2002; Murphy et al., 2006). Estimates of N volatilisation loss during combustion of forest floor and other fuels range from ca. 50 to 100% of the N content (DeBell and Ralston, 1970; Feller, 1982, 1983; Little and Ohlmann, 1988). A slightly lower decrease in N contents in forest floor (31 to 51%) was observed after a prescribed burning (Murphy et al., 2006). Estimates of S loss by volatilisation range from ca. 20 to 90% of the S contained in the fuel - higher with higher temperatures. More prolonged burns tended to result in greater losses (Allen, 1964; Sanborn and Ballard, 1991; Tiedemann, 1987). The long-term effects of fire on soil N are dependent upon fire intensity and time since the fire (Johnson and Curtis, 2001). Low-temperature fires (e.g. prescribed burning) may cause little initial loss in mineral soil N (Wells, 1971). Other elements, such as P, Ca and Mg require higher temperatures to volatilise. Murphy et al. (2006) found no significant change in Ca and Mg contents in forest floor after a prescribed burning in Sierra Nevada. They hypothesised that the lack of change in these elements after fire depends on an existing large base cation pool in the mineral soil before burning which makes it difficult to detect fire effects. Fire may increase soil pH and stimulate nitrification with the potential for a temporary increase in nitrate leaching after the fire (Chapter 6.9).

Tree seedlings were observed to grow better on burned sites (Ballard, 2000). Wildfire commonly generates a pulse of plant-available nutrients in the soil that can be taken up by regenerating vegetation, and reduces forest floor depth, resulting in a seedbed appropriate for the establishment of early successional species.

3.6.2. Weed control

Weeds exert a strong competition in establishment of a new culture and understory control in this period greatly improves tree growth in a number of species (Chang and Peston, 2000; Munson and Timmer, 1995; Sutton, 1995; Nambiar and Sands, 1993). It is the increased competition for water and nutrients between weeds and trees that might cause decreased tree growth. A significant effect of weed control (both mechanical and chemical) on tree growth was observed on Loblolly pine in a range of sites in the United States (Sanchez et al., 2006). Weed control includes a number of removal methods:

- Mechanical removal by patch scarification, trenching, mounding and ploughing (increase mineralisation)
- Chemical removal by the use of herbicides (decrease biological uptake)
- Mulching
- Competitive weed control

Mechanical methods were described above. Use of herbicides before and after planting of seedlings is the most common way of controlling weed and by far the most cost-effective method as well. This may when performed efficiently, leave the soil bare after treatment. Périe and Munson (2000) observed an effect of both scarification and herbicide treatment on the organic layer quality. Herbicide application reduced organic C mass by 46%, total N mass by 15% and acid phosphatase activity by 64%. However, the use of several herbicides is restricted in many European countries. Mulching involves covering the soil around trees with a cover material, which will prevent weeds from germination. Wood chips have been widely used, but also degradable plastic and cardboard have been tested. Removal of weeds in cultures by competitive weed control involves the use of other vegetation to take over and suppress the weeds, yet allow the seedlings to get enough light and water to grow. Especially, rye is used as competitive vegetation to weeds when planting on former agricultural soils. Nursery trees, growing faster than the proper tree species, may also help to create a faster forest climate earlier and hereby repress several weeds and prevent frost.

All removal methods disturb the soil to some extent. The most intensive mechanical disturbances increase net N mineralisation, nitrification, and nitrate losses to seepage water (Vitousek and Matson, 1985; Ogner, 1987a, b; Attiwell and Adams, 1993). In minimally treated plots nitrification declined from nearly 100% to 30% over a 5 years period, whereas herbicide and other intensive removal methods caused a consistent increase in nitrification (Vitousek et al., 1992). The patterns of disruption and recovery were the same: disturbance increased mineralisation rates for a relatively short period followed by a longer period in which net mineralisation decreased.

3.7. Stand management and harvesting

Changes to the soil caused by cleaning and harvest, especially after clear-cutting, are sudden and at times also drastic. The effects are more pronounced when the share of removed biomass is increased as for example in whole tree harvesting (WTH). Canopy removal by thinning and clear-cut harvesting temporarily increases the amount of precipitation and sunlight reaching the forest floor, reduces transpiration rates and causes soil moisture to increase leading to more favourable conditions for decay microorganisms (decomposition and mineralisation) (Piene and van Cleve, 1978; Binkley, 1984; Johnston and Crossley Jr., 2002). Without canopy as after clear-cutting, soil temperature increases, and the net effect is an increase in nitrification and N mineralisation at a time when plant uptake is disrupted and N demands are low. Furthermore, the outflow of run-off and seepage water is larger due to lower evapotranspiration (Knights et al., 1991; Qualls et al., 2000). As a result, ammonium is available as a substrate for nitrifying bacteria, and nitrate is produced and subsequently leached below the rooting zone (Vitousek et al., 1982) leading to soil acidification and loss of nutrients (Hüttl and Schaaf, 1995).

A significant amount of nutrients are exported from the ecosystem in harvesting (Glatzel, 1990; Augusto et al., 2002; Raulund-Rasmussen et al., 2007). Many studies have shown a significant negative effect of intensive biomass harvesting on productivity (e.g. Smith et al., 2000). Mostly, the negative effect is explained as an induced nitrogen deficiency. If the effect continues in the long-term, such forest management does not satisfy criteria and indicators for sustainable forest management (Raulund-Rasmussen et al., 2007). Fertilisation may neutralise the possible negative production effect, at least in the short-term (Jacobson et al., 2000). However, other studies have shown no negative production effect of intensive biomass harvesting (e.g. Powers et al., 2005) indicating that the soil including atmospheric deposition has been able to supply the stand with sufficient amounts of nutrients, at least in the short-term.

There are two major types of forest management systems, intensive and continuous cover forestry (CCF) as described by Gadow (2001) and Mason et al. (1999). The close-to-nature-forestry management alternative (Duncker et al., 2007) is a type of CCF forestry. Intensive alternatives are characterised by a regeneration period followed by thinning and a final harvest after reaching a final rotation age. CCF on the contrary is characterised by selective cuttings in different cycles not following a certain harvest-regeneration pattern and the stand age is undefined. Intensive harvesting of biomass from the forest ecosystem is likely to influence some relevant SQ parameters which are the nutrient stock and availability, soil acidity, the content of C, soil structure and turn over of organic matter, mineralisation rates and soil biology. Clear-cutting and WTH are parts of the wood biomass production forest management alternative and partly also of the intensive even-aged alternative (Duncker et al., 2007). CCF is expected to reduce the negative impacts of the intensive alternative, and is therefore attractive within a multi-purpose forestry framework. The transformation from even-aged stands in the intensive alternatives to uneven-aged stands in CCF will imply a change from a simple, homogeneous structure to a variable, complex structure (O'Hara, 2001). This change is, however, still not well studied.

3.7.1. Nutrient pool

The amount of nutrients in the ecosystem is reduced in harvesting operations. Compared to natural systems without harvesting the long-term inputs of organic matter and nutrients to the soil are reduced. Significant amounts may already be removed in stem harvesting, while intensified forest biomass utilisation for energy purposes causes further export of nutrients from the forest ecosystem.

For single operations, far most nutrients are removed in a clear-cut compared to thinnings, both for stem harvesting and when WTH is practiced. The size of flux depends on site productivity, harvesting intensity, soil type, the species of trees harvested, and size of the trees at the time of harvesting (Stupak et al., 2007a). Some studies estimated removals in single operations corresponding to clear-cutting (Table 3.1).

Table 3.1. A selection of case study and model estimates of removal in single operations corresponding to clear-cutting. The latter are based on biomass functions of Marklund (1988) combined with Danish and Swedish growth models (Leary et al. 2007a,b; Ekö, 1985), and Finnish, Swedish and Norwegian data for nutrient concentrations in biomass fractions. Note that estimates are theoretical figures of removal without taking into account whether it is technically possible to for example remove all above-ground biomass (Stupak et al. 2007a)

Tree species	Diameter	Age	Harvest	N	P	K	Ca	Mg	Reference
	cm	years				kg ha^{-1}			
Case studies									
Black spruce		65	Stem only	43	12	25	98	8	Ballard, 2000
Black spruce		65	WTH	167	42	84	277	27	Ballard, 2000
Jack pine		65	WTH	185	14	93	132	20	Ballard, 2000
Norway spruce		65	WTH	534	70	167	521	-	Nykvist, 1974
Norway spruce		65	Stem only	370	43	241	451	60	Nihlgård, 1972
Norway spruce		65	WTH	770	89	437	459	70	Nihlgård, 1972
Norway spruce		77	WTH	413	75	370	789	95	Klimo, 2002
Pinus taeda		60	WTH	250	25	155	185	65	Switzer et al., 1968
Model estimates									
Norway spruce	20-40		Stem only	60-500	5-73	30-240	70-530	9-62	Stupak et al. 2007a
Norway spruce	20-40		WTH	240-1050	23-146	80-470	170-1010	24-41	Stupak et al. 2007a
Scots pine	20-35		Stem only	40-320	4-37	20-160	40-250	9-57	Stupak et al. 2007a
Scots pine	20-35		WTH	130-500	12-58	50-250	70-330	17-76	Stupak et al. 2007a
Birch	17-30		Stem only	30-480	3-48	20-250	40-890	6-55	Stupak et al. 2007a
Birch	17-30		WTH	50-690	5-72	20-320	60-560	8-78	Stupak et al. 2007a

There is a substantial difference in the size of removal, depending on the size, age (Ranger et al, 1995) tree species and density of the trees at the time of cutting (Cole and Rapp, 1980; Perala and Alban, 1982; Glatzel, 1990; Augusto et al., 2000), site productivity, harvesting intensity, and nutrient concentration level in the biomass (Stupak et al., 2007a). Initial planting density and applied thinning regime seem to be of less importance for the size of average biomass and nutrient removal, but there are studies showing a potential effect of initial spacing (Gamborg 1997), and thinning regime (Soalleiro et al. 2007, Skovgaard et al., 2006).

Compared to harvesting of stems only, very intensive biomass removal, including also harvesting of stump and roots, theoretically may cause the average removal of nutrients per year over one rotation to increase more than 6 times for N and P (Stupak et al., 2007a). The corresponding increase in removal of biomass was only calculated to be about 2 times. A typical intensive harvesting scenario in practice today, especially in spruce, is removal of logging residues at clear-cut in addition to removal of stems in all thinnings and clear-cut. This corresponds to the “logging residues” scenario in Table 3.2, where results from selected model calculations are shown. For the “logging residues” scenario, the removal of biomass increased about 20-30% for Norway spruce, and about 10% for Scots pine and birch, while the removal of nutrients more than doubled for spruce, and increased about 20-70% for pine, and 20-50% for birch. For comparison, Goulding and Stevens (1988) found losses of K twice as high for WTH logging as for conventional stem only logging of Sitka spruce.

Table 3.2. Increase in average removal of biomass and nutrients per year over a rotation for different harvesting alternatives relative to harvesting of stems only. The harvesting scenarios are defined: “Low intervention”: removal of stems only in the final felling,, “Logging residues”: branches, twigs and needles are additionally removed in the final felling, “Intensive”; branches, twigs and needles are additionally removed in all thinnings, and “Very intensive”: root and stumps are additionally removed after final felling. The intervals are based on results from 6 spruce, 4 pine and 3 birch scenarios on sites of varying productivity rates (production class 5-20 $m^3\ ha^{-1}\ yr^{-1}$) . All scenarios for birch are without leaves. In the scenarios, it has not been considered that in practice, 20-40% of the logging residuals are left on the site, and some needles might have been shed due to pre-drying.

Harvesting alternative	**Biomass**	**N**	**P**	**K**	**Ca**	**Mg**
Norway spruce						
Low intervention	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7
Logging residues	1.2-1.3	1.8-2.4	1.8-2.2	1.6-2.2	1.6-2.1	1.6-2.1
Intensive	1.4-1.6	2.8-3.5	3.0-3.6	2.5-3.0	2.4-2.9	2.4-2.9
Very intensive	1.8-2.1	3.2-3.9	3.4-4.1	2.9-3.5	2.8-3.4	2.8-3.4
Scots pine						
Low intervention	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7	0.5-0.7
Logging residues	1.1	1.5-1.7	1.5-1.7	1.5-1.6	1.2-1.3	1.2-1.3
Intensive	1.2-1.3	2.1-2.4	2.1-2.5	2.0-2.3	1.5-1.7	1.6-1.7
Very intensive	1.6-1.7	2.4-2.8	2.4-2.9	2.3-2.7	1.8-2.1	1.9-2.1
Birch						
Low intervention	0.4-0.5	0.4-0.5	0.4-0.5	0.4-0.5	0.4-0.5	0.4-0.5
Logging residues	1.1	1.2-1.4	1.3-1.5	1.2-1.3	1.3-1.4	1.2-1.3
Intensive	1.2	1.6-1.7	1.8-1.9	1.4-1.5	1.6-1.7	1.4-1.5
Very intensive	1.5	1.9-2.0	2.1-2.2	1.7-1.8	1.9-2.0	1.7-2.0

A consequence of nutrient removal in harvesting is that reserves may become depleted unless mineral weathering is rapid, atmospheric deposition is high, or fertilisation is performed. Atmospheric deposition of N due to pollution is high in some areas, while atmospheric deposition of sea salt might provide a substantial nutrient input of especially Mg, but also K and Ca to the system on sites located in the vicinity of the sea (Kreutzer et al., 1998).

The nutrient balance approach (section 3.3.1.1) has been used in several studies to quantify the effect of intensified biomass utilisation on the ecosystem nutrient status (Ek and Boyle, 1972; Boyle et al., 1973; White, 1974; Boyle, 1976; Krapfenbauer, 1985; Olsson, 1996; Møller, 2000; Akselsson and

Westling, 2005). Likens et al. (1998) and Huntington and Ryan (1990) have shown a biomass export related decrease in soil stores of nutrients, often in Ca. A decrease in soil stores may not *per se* be critical as long as the level is far above a critical low level. But the negative nutrient balance may be interpreted as a warning for a long-term negative effect. Nitrogen has often been in focus in nutrient balances since most forest soils are in shortage of available N. N related effects have been demonstrated, but also shortage of P, Ca and K may become a problem in areas with significant N deposition (Raulund-Rasmussen et al., 2007). Magnesium deficiency may occur in some areas just like it was induced by acid rain in Central Europe in the 1980s (Schaff, 1995)

In studies by Watt et al. (2005) and Powers et al. (2005), concentrations of total soil C, N and P were reduced after harvest of all aboveground living vegetation. Other researchers have likewise found that WTH significantly reduces soil concentrations of N (Merino and Edeso, 1999) and P (Mroz et al., 1985; Tuttle et al., 1985; Sanchez et al., 2006). Reductions in concentrations of these elements may have been caused through removal of the humus layer and topsoil in disturbed plots during harvesting, thus reducing the potential availability of nutrients. In other studies, harvesting had no significant effect on soil N (and C) (Johnson and Curtis, 2001; Sanchez et al., 2006).

3.7.2. Soil acidity

Forest management operations like harvesting significantly influence all the acid producing processes (Section 3.3.1.2). Several estimates are available trying to quantify the contribution of harvesting versus acid deposition. In high deposition areas air pollution is considered the most important source of acidity whereas biomass harvesting may be more important when performed at a very intensive level (Adams et al., 2000; Watmough and Dillon, 2003). Clear-cutting may increase soil acidity and the loss of base cations as shown by Likens et al. (1970), Mann et al. (1988), Keenan and Kimmins (1993) and Simard et al. (2001).

3.7.3. Compaction and soil structure

Forest harvesting equipment has evolved considerably during the past decade from man-held chain saws towards mechanised heavy machinery such as crawler tractors, rubber-tired feller-bunchers and grapple skidders. Soil disturbance from harvesting may induce substantial alterations to soil physical properties since the heavy harvest machinery compresses the soil which may lead to increased bulk density, loss of soil porosity, increased erosion, decreased availability of water and as a consequence of these, reduction in seedling survival and potential reduction in diameter growth (Cullen et al., 1991; Wang et al., 2005; Eisenbies et al., 2005).

Soil disturbance by harvesting machinery is influenced by factors such as the amount of litter and slash on the forest floor, soil texture, drainage and soil moisture, weight of harvest machines, wheel size, speed, operator skill, terrain and weather (Sirois et al, 1985; Wang et al, 2005). Especially, soil moisture and weather play important roles since the machinery may compact soils and interfere with normal soil drainage when run on moist and saturated soils (Moehring and Rawls, 1970; Adams and Froehlich, 1981; Greacen and Sands, 1980; Miwa et al., 2004; Wang et al, 2005; Kabzems and Haeussler, 2006). Ares et al. (2005) observed that approximately half of the observed forest area was affected by trafficking. Cullen et al. (1991) observed no difference in compaction between moderate and severely trafficked areas and suggested that most compaction will occur during the first few passes of the machinery over the area. This was even observed by Shetron et al. (1988). Forest practices and site conditions are, however, extremely variable and soil disturbance do not always occur. When it does do occur the impacts are not always equal (Adams and Froehlich, 1981).

Results indicate that variations between disturbed and undisturbed treatments were mostly attributed to a reduction in macro-pore volume towards a higher percentage of small pores and to a lesser extent to increased bulk density (Gent et al., 1983; Watt et al., 2005). Typically, an applied force pushes together the mineral and organic particles effectively filling or compressing many of the existing soil pores (Adams and Froehlich, 1981), through which air and water normally flows unrestrictedly. Simcock et al. (2006) found that macro-pore volumes decreased from 13.5% to 5.7% in the topsoil after compaction. In a study by Tan et al. (2005), soil porosity was reduced by 50%. A macropore volume <10% has been observed to restrict root growth (Greacen and Sands, 1980; Rab, 2004; Simcock et al., 2006). This critical value for macro-pore volume seems to be a valuable indicator across a wide range of soils, whereas the use of bulk density as an indicator is more difficult since the bulk density may vary widely (Simcock et al., 2006).

Despite this, most studies of compaction have concentrated on the effect on bulk density. Wang et al (2005) compared bulk density in a forest area before and after harvest in West Virginia and observed no significant difference in pre-harvest and post-harvest soil bulk density at the site. On the other hand, Kabzems and Haeussler (2006) observed an increase of 16%, Cullen et al. (1991) found an increase of 21-76%, Tan et al. (2005) an increase of 24%, Powers et al. (2005) an increase of 18% and Ares et al. (2005) an increase of 27% in bulk density as a consequence of trafficking and compaction on different soil types down to a depth of 15-20 cm. Several more such studies can be quoted showing increase in bulk density after compaction (Greacen and Sands, 1980; Breland and Hansen, 1993; Simcock et al., 2006). Furthermore, Klimo (2002) observed reductions in porosity immediately after felling both in the stand and at the skid trails. Powers et al. (2005) observed that little compaction occurred in soils where the initial bulk density was larger than 1.4 Mg m^{-3} since these soils could not be compacted further. Above bulk densities of 1.40-1.55 Mg m^{-3} plant roots has been found incapable of penetrating soils (Kozlowski, 1999).

Compaction furthermore increased the available water holding capacity (Ares et al., 2005) and reduced soil moisture content by 11% after forest floor removal (Tan et al., 2005). Other studies have shown that the water holding capacity might increase with compaction on coarse-textured soils; however, on fine-textured soils the water holding capacity is generally reduced (Cullen et al., 1991).

Large machinery sometimes remove organic matter from the forest floor and along with the extra compaction, soil nutrient availability will change (Briars et al., 1995). Also, compaction and removal of organic matter from the forest floor might influence weathering rates and N mineralisation (Zabowski et al, 2000; Wang et al, 2005), the latter since organic material might be physically protected against microbial attacks in small pores after compaction (Breland and Hansen, 1996). The changed macro-porosity after compaction caused the available volume for nematodes to reduce which led to a smaller grazing of microorganisms and thus a smaller turnover of N. These authors observed that compaction had a negative effect on N mineralisation, which was reduced by 18%, mostly at higher soil moisture contents. However, Tan et al. (2005) found no change in net N mineralisation rate caused by compaction in neither forest floor nor mineral soil, but along with Hendrickson et al. (1985) they observed an increase in the net N mineralisation rate after the removal of the forest floor since the environmental conditions were improved for microbial activities. E.g. soil temperature in the forest floor and the mineral soil was reduced by 2.2 and 0.7 °C, respectively. The compaction studied by Tan et al. (2005) did not affect pH, total C and N contents or C/N ratio in the forest floor whereas the two latter were increased in the mineral soil. Removal of the forest floor otherwise represents a substantial loss of habitat for most soil fauna.

Compaction will affect growth through a deteriorated plant root environment where roots will have difficulties of extending during dry summers and during wet winters because of lack of oxygen. Simcock et al. (2006) found that seedlings planted in compacted soils had greater mortality and slower growth than seedlings planted in uncompacted soil. Also, Greacen and Sands 1980) reviewed 142 studies dealing with compaction and found that compaction led to reductions in tree growth in 82% of the cases. On the other hand, Sanchez et al. (2006) observed no effect on tree growth after compaction on a range of 62 sites in the United States.

3.7.3.1. Forest roads and skid trails

Ground-based skidding results in soil compaction as observed by Wang et al. (2005) and other soil structural changes, influencing soil water retention, and reducing soil aeration, drainage, and root penetration as discussed. Moist, fine-textured soils are particularly susceptible, whereas frozen soils tend to be quite resistant to structural degradation induced by traffic (Ballard, 2000). Reductions in tree height, diameter and volume growth are often observed where soils have been affected by skidding activity (Froehlich et al., 1986; Wert and Thomas, 1981). The greatest increases in soil bulk density (more than 50%) on skid trails are associated with the first trips over the ground (Adams and Froehlich, 1981; Froehlich et al, 1981; Greene and Stuart, 1985; Shetron et al, 1988; Wang et al, 2005), and even quite low ground pressures (e.g. 35-65 kPa) can result in substantial compaction (Froehlich, 1978). Even where additional trips do not result in significant increases in bulk density, changes in pore size distribution may continue to occur, with large pores collapsing to form smaller ones (Lenhard, 1986). Localised nutrient removals can be significant. The most severe compaction occurs within the upper 5-10 cm of the soil surface which is also where most of the fine-roots are found (Adams and Froehlich, 1981).

As for harvesting, construction of skid trails and forest roads cause rates of erosion and sedimentation of water bodies to increase (Worrel and Hampson, 1997). Also, Hartanto et al. (2003) found soil losses to be highest from skid trail plots in harvested forests.

Partial harvest methods, such as group selection, thinning or shelterwood harvest have been suggested as more sustainable methods of bole removal than clear-cutting (Boyce and Martin, 1993; Johnston and Crossley Jr., 2002). Tradeoffs between N losses during harvest must be balanced against the increased heavy machinery traffic for partial harvesting. More frequent entries into the forest are associated with group selection methods, resulting in an increased amount of roads and skid trails.

It is recommended to focus skidding to a few selected trails and minimize trafficking in other areas to limit the area of compacted soil. Also, the construction of skidding trails and roads in the forest as well as forest ground-based equipment operations should preferably be limited to times when soil moisture levels are low (Adams and Froehlich, 1981; Wang et al., 2005).

3.7.3.2. Recovery after compaction

Soils might recover from compaction; however, such recovery might be slow and take more than 20 years (Rab, 2004). Recovery depends on soil type and the degree of compaction. Coarse-textured soils tend to recover faster than fine-textured soils (Adams and Froehlich, 1981). Recovery has been estimated to take everything from 5-15 years after logging to 40 years or more under skid trails and forest roads (Greacen and Sands, 1980; Worrel and Hampson, 1997; Klimo, 2002). Powers et al. (2005) resampled soil 10 years after actual compaction and measured bulk density and found that soils rarely recover from severe compaction, regardless of their opening bulk density. If compaction persists for longer periods beyond one rotation period the impacts will increase cumulatively and a new more compacted steady state of soils will most likely be the result of today’s use of heavy machinery within forestry.

3.7.4. Residue management

Treatment of harvest residues is an important issue for management of forests performed in order to prepare for a new stand. Some countries have rather detailed legislation for management of harvesting residues which sets the framework (Stupak et al. 2007b). Harvest residues might be retained at the forest area, removed when harvesting or burnt directly on the area. The *in-situ* retention of residues includes leaving them in rows, planting in-between rows, or distributing the residues randomly over the forest area. The appropriate management of these residues is critical for maintaining tree growth and maintaining the long-term site productivity. It has been reported that retention of harvest residues on the area has improved tree growth in the short-term (Chen and Xu, 2005; Mendham et al., 2003). Long-term effects may be comparable to the effect of intensive harvesting.

As for prescribed burning, burning of residues will result in a loss of organic material as C, N and S by volatilisation (e.g. Mendham et al., 2003). Residue burning will thus result in nutrient export through volatilisation and leaching (e.g. Mendham et al., 2003; O’Connell et al., 2004).

Maintenance of site productivity will partly depend on management operations that aim to maintain an adequate supply of soil N and other nutrients that influence the growth rate. Residue management after harvesting can have a large effect on mineral soil N in coniferous forests (Debyle, 1980; Burger and Pritchett, 1984; Stone and Elioff, 1998; Smethurst and Nambiar, 1990). On average, residue removal (i.e. WTH) caused a 6% reduction in A horizon N whereas leaving residues on site caused an 18% increase in soil mineral N content. The positive effect on soil N of leaving residues on site seems to be restricted to coniferous species. Several studies have clearly shown that residues had little or no effect on soil N in hardwood or mixed forests (Hendrickson et al., 1989; Mattson and Swank, 1989; Knoepp and Swank, 1997; Johnson and Todd, 1998; Johnson and Curtis, 2001) as well as in Eucalypts (Mendham et al., 2003). Soil total C and N contents in the topsoil were considerably higher when residues are retained in the area compared to when they are removed (Chen and Xu, 2005). On the contrary, Olsson et al. (1996) found no such impact. O'Connell et al. (2004) found an annual N mineralisation rate in the topsoil (0-10 cm depth) which was significantly higher in treatments with retained residues as compared to treatments where residues were removed. Retention of residues after harvest might also result in higher quantities of soil exchangeable K, Ca and Mg as seen by Mendham et al. (2003).

Other manipulations such as sub-soiling, harrowing, disking, and bedding have been employed for reasons such as competition control, micro-site improvement and to increase nutrient availability. However, these techniques have also been found to disrupt the soil structure and mix the remaining stumps, logging slash, and root networks after harvest (Terry and Hughes, 1975; Haines et al, 1975; Johnston and Crossley Jr., 2002). See also site preparation in chapter 3.5.

A growing trend in residue management is the removal of stumps. Stumps are removed to decrease the root rot infection in the new stand, and to harvest biomass for energy. The extra removal of nutrients after stump harvesting has been estimated in table 3.1 in the very intensive harvesting alternative, but there are also other effects than the nutrient removal itself. In order to remove stumps they are drawn up from the soil disturbing the soil rather vigorously and the effect will probably be comparable to heavy soil preparation methods, mixing the soil around. An environmental assessment review on multiple effects of stump harvesting was lately compiled in Sweden (Egnell et al., 2007). They conclude that the amount of inorganic N probably increases after stump harvesting and probably there is an increased downward transport of sediments with seepage. However, the effects of such removal to SQ are not well documented, and research is needed in this area.

3.7.5. Erosion

Harvesting cause erosion to increase, and as a result, sedimentation of water bodies also increases (Worrel and Hampson, 1997). Soil erosion from forest harvesting is a major environmental concern, especially when harvesting is performed on steep slopes (Kochenderfer et al., 1997; Hood et al., 2002). There is little literature on erosion after logging as well as research comparing the effects of silvicultural treatments on soil erosion. Powers et al. (1990) found the erosion after logging in the US to be within the limits of the rate of soil formation while McColl and Grigal (1979) estimated a rather high erosion after logging. Hood et al. (2002) estimated soil loss for forest land in the Appalachians on five harvesting treatments which included a clear-cutted system, a leave-tree harvest, a sheltergroup, a group selection and an uncut control. They concluded that the group selection would have more soil loss over a 100-yr rotation than the other harvested treatments, primarily caused by repeated improvement cuttings in the openings every 20 years and the hereby multiple entries on the skid trails (see Chapter 3.8). The magnitude of soil loss was observed to depend largely on canopy cover, plant density, depth of the forest floor, and presence of woody debris (Hartanto et al., 2003). The size of the tree canopy determines the erosive power of precipitation while the other factors protect the soil surface and prevents soil loss.

3.8. Synthesis

In this chapter we reviewed the impact of forest management operations on SQ with the main focus on available nutrient stocks, soil acidity, and soil erosion and compaction.

Forest growth is a complicated issue that involves many complex processes between the forest soil and the trees. Forest managers need to know how forest operations and other site disturbances may be performed to sustain site productivity and tree growth. They need to know if decisions like organic matter removal incl. harvest residue removal, tree species changes, fertilisation, compaction etc. will reduce SQ and if they can ameliorate SQ reductions profitably. Therefore, recommended best management practices and possibly other decision support tools should be developed. Examples of recommendations are given in Johnston and Crossley Jr. (2002) which would help forest managers judge the positive and negative effects of their practices on sustainability. Other examples are recommendations for extraction of forest fuels (e.g. Koistinen & Ajala 2005, National Board of Forestry 2002).

Within the agricultural community, SQ concepts and methods have been developed and are suggested to form the basis for soil-based evaluation of the impact of management operations on the long-term productivity (Kelting et al., 1999). These authors demonstrate a model – the Soil Quality Index Model, which integrates the components of SQ into one measure of management impacts on soils. Here, they look at five key growth-determining attributes of forest soils, which are: 1) root growth promotion, 2) storage, supply and cycling of nutrients, 3) accept, hold and supply of water, 4) promoting gas exchange, and 5) promoting biological activity. Such models could help in the development of codes of practice within forestry as well.

In this chapter, we have identified harvesting of biomass to cause a significant decrease in the soil content of almost all nutrients and an increase in soil acidification depending on the weathering capacity of the soil minerals and the kind and intensity of biomass removal. Change in tree species might also accelerate the negative nutrient balance and acidification both due to increase in biomass harvesting and increased deposition of air pollution compounds. Modern intensive forestry includes heavy machine trafficking with negative influence on bulk density, macro-porosity and erosion and large reductions in growth potential are observed hereafter.

4. The impact of forest management on the carbon cycle

by

Denis Loustau and Emil Klimo

4.1. Introduction, concepts and indicators

4.1.1. Temporal and Spatial domain

We considered in this section the forest ecosystem including the soil and vegetation. Forestry practices, including fertilisation, site preparation, water management (drainage), vegetation management, logging and harvest are accounted for in this review. The spatial scale is typically the homogenous site and the temporal scale is the entire life cycle of the forest ecosystem, from trees regeneration to the final clear-cut or disturbance (fire, windstorm) causing the trees to be felled. Issues regarding the downstream carbon flow in the wood products life cycle and the fossil fuel consumption for wood processing transport and use. Acknowledging that it may represent the major potential impact of forests on the future climate (Cannell, 2003), we shall however not discuss in details the effect of fossil fuel substitution by wood products since it is addressed in other EFORWOOD modules.

4.1.2. Carbon and greenhouse gases biogeochemical cycle

The carbon element is the major constituent of anthropogenic greenhouse gases molecules such as the carbon dioxide, CO_2, methane, CH_4, and halocarbon (CFC). The uptake, release and storage of carbon in soil, biomass and wood products are the main process by which forests and forest-wood chains affect the biosphere-atmosphere interactions and the atmospheric greenhouse effect. In addition, forests and forest-wood chains may potentially interfere with climate through other mechanisms such as the surface energy balance, a biophysical effect including the albedo and the Bowen ratio, and additional greenhouse gases such as N_2O and ozone. This biophysical effect must be taken into account as far as a complete assessment of the environmental impacts of forest is the objective (Betts, 2000; Betts et al., 2007; Gibbard et al., 2005).

The potential for enhancing carbon sequestration by forestry, including land use changes, is estimated to 11-15% of the actual fossil fuel emissions at the global level and 5-11% in the EU15 (Brown et al., 1996; Cannell, 2003). In the case of EU15, the potential including fossil fuel substitution by energy would be raised to 25% of the fossil fuel emissions.

The analysis of the management impact on the forest carbon cycle must assess the entire life cycle from tree regeneration to final harvest and wood products use. The life cycle assessment has become a standardised protocol to examine the environmental impacts of product or process (Sonne, 2006). Unfortunately, most of the available studies available so far do consider only a part of the forest life cycle or a fraction of the ecosystem which renders their interpretation difficult in the context of the global carbon cycle. For instance, the fossil fuel consumption associated with management operations, the energy use for seedling production and transportation or the energy use for fertiliser production and application are rarely considered.

4.1.3. The carbon cycle in forests, terms and definitions

The terrestrial carbon cycle includes five major pools, the atmosphere, soil, biomass, harvested products and fossil fuel (Figure 4.1). Their turnover-rate is decreasing from the atmosphere to the biomass, soil, fossil carbohydrates and sediments. The residence time of carbon, is typically one to two orders of magnitude lesser in the atmosphere (5-7 years) than in biomass (1-250 years) and soil (5-10 000 years). The soil C pool is characterised by a wide range of turnover rates, from 1 to 10^{-4} yr^{-1}

which are determined by the chemical quality of the organic matter and the biophysical environment which makes it available for decomposition rates (temperature, oxygen, water, pH etc.). Carbon is withdrawn by vegetation from the atmosphere through the photosynthesis (gross primary production) which fixes carbon atoms in carbohydrate molecules such as sugars. When the energy encapsulated in this process as covalent bonds is being used, carbon is oxidised as CO_2 which returns into the atmosphere. This mineralisation occurs in plants as ”autotrophic” respiration, and animals and decomposers as ”heterotrophic” respiration and during other mineralisation processes (combustion in industrial processes, fires). The balance of CO_2 exchanged by a given forest ecosystem with the atmosphere is called the net ecosystem exchange (NEE). The net carbon balance of the only vegetation compartment is the net primary production (NPP). Other molecules such as methane, CH_4, volatile organic compounds (isoprene etc.) contribute marginally to the exchange between the ecosystem and the atmosphere. Methane is produced under anaerobic conditions, during soil organic matter decomposition and this emission may be taken into account for managing wetlands and peatlands. Methane has been demonstrated to be also produced by green leaves under normal conditions but the cause and the relevance of this emission in the carbon cycle is still under debate (Keppler et al., 2006). The relative importance of the dissolved carbon flow in the forest carbon cycle is not crucial in the context of this chapter even if recent studies show that it may account for several percents of the total carbon exchange between ecosystem and the environment.

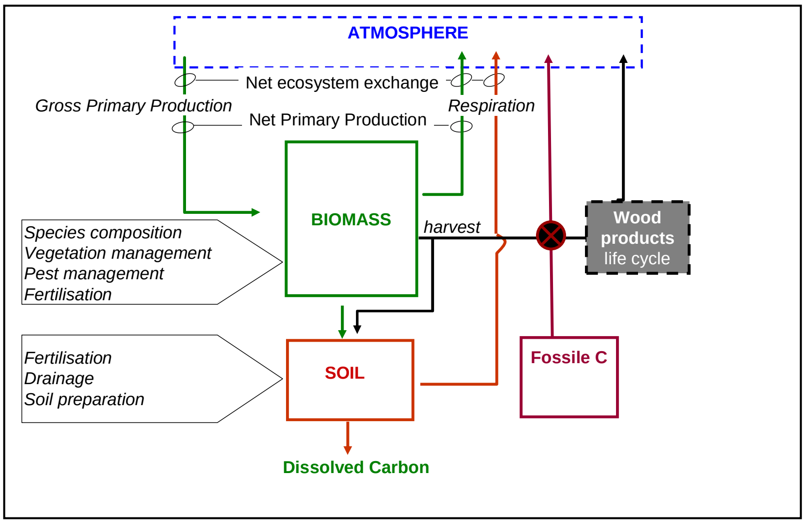

Figure 4.1. Carbon cycle in a managed forest ecosystem and main effects of management practices.

4.1.4. Indicators

The global carbon cycle includes five pools, the atmosphere, biomass, harvested products, soil and fossil carbon pools which must be included in the assessment of the greenhouse gas balances of forestry scenarios. Two methods can be used for calculating the greenhouse gases balance and evaluate the impacts of forest and land management, namely the “flux” and “stock” methods. For convenience, practicability and consistency with IPCC guidelines (Naaburs et al., 2004), we recommend to adopt the stock change approach for instance for the TOSIA model. The stock change approach quantifies the net change in stock of each of the five pools between two dates. It may also be used for a comparison to a reference value or scenario. In the context of the EFORWOOD project and for the TOSIA, it was proposed to compare the steady state value of stocks at equilibrium for each FWC scenario with the value observed in a given reference scenario (PD. 221).

Not all the greenhouse gases considered have the same global warming potential (Houghton et al., 1990). Releasing in the atmosphere 1 kg of carbon as CO_2 or CH_4 affects differentially the climate (Table 4.1). It is therefore recommended to distinguish each greenhouse gases as far as possible. Table 4.2 lists the main operational indicators derived from this approach and proposed within the framework of the EFORWOOD project.

Table 4.1. Global warming potentials (unitless) relative to the carbon dioxide (from Houghton et al. 1990).

Trace gas	**Estimated lifetime in the atmosphere (years)**	**Global Warming potential**		
		integration time (years)		
		20	**100**	**500**
CO_2		**1**	**1**	**1**
CH_4	**10**	**63**	**21**	**9**
N_2O	**150**	**270**	**290**	**190**

Table 4.2. (from PD 221). Indicators list proposed for assessing the forestry scenarios impact of the greenhouse gas balance.

Carbon sequestration in the forest ecosystem		
carbon in the above ground living tree compartment biomass	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon in the root biomass	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon in above ground living herb and bush compartment biomass	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon in the forest floor	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon in dead woody biomass	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon in the mineral soil	tons C pr ha	MCPFE (C.1.4) Eforwood WCI (16)
carbon loss due to leaching of DOC	tons C pr ha pr year	
carbon loss due to loss of sediments	tons C pr ha per year	
Greenhouse gasses emissions		
emission of methane	kg C pr ha pr year	Eforwood WCI (22)
emission of nitrous oxide	kg N_2O pr ha pr year	Eforwood WCI (22)

4.1.5. Direct and indirect impacts of forest management alternatives

Management affects the carbon cycle in forest ecosystems directly. For instance, site preparation techniques such as ploughing may deplete the soil carbon stock through fastened mineralisation of the soil organic matter. There are also indirect effects such as the faster recovery of photosynthesis due to enhanced tree growth. Both sorts of effects will be addressed here. The assessment of the overall impact of a FWC scenario will be evaluated comparatively and will include the soil, biomass, fossil carbon and atmosphere. The changes in carbonated rocks and sediments will not be considered here.

4.2. Approach and methods

Literature can be used for documenting and quantifying the direct and short-term impacts of management practices on the forest carbon cycle: the carbon stocks in biomass and soil, the net primary production, wood biomass, soil respiration. Conversely, there is few data allowing a direct comparison of the effects on carbon cycle between management alternatives. Models of forest carbon cycle models are therefore necessary for providing an overall assessment of the FWC scenarios on the carbon cycle.

4.3. Effect of management operations

4.3.1. Site preparation – Physical treatments

4.3.1.1. Clearing operations (prescribed burning), soil tillage, ditching, local ploughing, mounding, etc.

The soil preparation aims at improving tree regeneration and growth through enhanced nutrient availability, a better soil structure and aeration, drainage and controlling competing weeds or pathogens. It allows a faster growth of trees at the juvenile stage and is therefore required for ensuring the success of tree regeneration. Soil preparation affects positively the photosynthetic carbon uptake by trees, the gross primary production. However, most soil preparation techniques speed up the mineralisation of carbon and nutrients from soil organic matter and forest floor, i.e. the heterotrophic respiration, depleting to some extent the amount of carbon stored in the soil carbon pool (Johnson, 1992). Intensity of soil preparation practices varies from local (mounding, ditching) to plain ploughing and the net loss of soil C generally increases with intensity (Johansson, 1994; Orländer, 1996). Soil preparation is the second most expensive forest operation in fossil carbon after harvesting operations.

4.3.1.2. Drainage

Long-term repeated measurements of carbon stock have shown that drainage has no effect on the carbon stock of Scots pine peatland in Finland (Minkinnen et al., 1998). Conversely, in accordance with the common understanding of the effect of drainage, flux measurements operated on forest stand evidenced much higher rate of soil respiration in drained areas than the average measured in other undrained sites. However, most peatland is emitting methane and it is not known whether the drainage and afforestation of peatland may lead to a positive or negative effect on the greenhouse gas balance (Cannell et al., 1993). This must be depending closely on site conditions, peat thickness, etc. so that the impact of drainage on the carbon balance of lowland sites and peatland is uncertain (Minkinnen et al., 2002 Laiho et al., 2003). Hargreaves et al. (2003) estimated the mineralisation rate of peat following afforestation in Scotland was relatively slow, ca. 1t $C\text{-}CO_2 \, . \, ha^{-1}.yr^{-1}$ or less so that the overall forest-wood products carbon balance was a net sink for 90-190 years. Laine et al. (1997) came to the same conclusion in Finland accounting for the global warming potential. Minkkinen et al. (2002) estimated accordingly that the net effect of afforestation of drained peatland from 1990 to 2100 in Finland was beneficial due to enhanced C store in tree stands, decreased methane emission whilst N_2O emission was marginally increased. However, Hargreaves et al. (2003) stressed that the total amount of carbon accumulated in peatland may exceed the potential sink capacity of the forest biomass making it detrimental to afforest peatland with high amount of carbon stored. Frokling et al. (2006) also showed that ancient peatland have a net cooling effect which suggests that drainage and afforestation should mainly target recent peatland with low carbon stocks.

4.3.2. Site preparation – Chemical treatments

4.3.2.1. Fertilisation

Considering that production in most northern temperate and boreal forests is chronically restricted by lack of N (Tamm, 1991; Vitousek and Howarth, 1991), a clear relationship between N fertilisation or deposition and annual NEE could be expected, as suggested recently by Magnani et al. (2007). Fertilisation studies show that the likely response to increasing N inputs is increased production of leaves and wood, including coarse roots (Oren et al., 2001; Iivonen et al., 2006) at the exception of ecosystems approaching N saturation where the effect of N may be less pronounced or even reversed (Magill et al., 2004).

N fertilisation is reported to decrease the mineralisation rate of the organic matter (Butnor et al. 2003) and to increase the C flux into the detritic pool, i.e. litterfall, which results in a net increase of the amount of C stored in humus layers and mineral soil (Fog, 1988; Persson et al., 2000; Johnson et al., 2001; Franklin et al. 2003; Jandl et al., 2002; but see Jandl et al., 2003 for an opposite result). The N-induced increase of the forest floor pool in a Scots pine forest may reach 100% over a century, 70% of this increase resulting from the decreased decomposition rate and 20% to increased litter production (Franklin et al., 2003).

4.3.2.2. Liming

A literature review showed that liming causes a net loss of C from forest soils in temperate and boreal forests owing to increased microbial activity and dissolved carbon leaching (Brumme and Besse, 1992; Jandl et al., 2003; Lundström et al., 2003). Lime application is routinely used in many European forests to ameliorate soil acidity. In a mature European beech (*Fagus sylvatica* L.) forest on acid soil with a moder humus and eight years following silvicultural treatments, there was no change in C and N over the entire forest soil profile including forest floor. Reductions in the F and H layers in limed gaps were compensated for by increases in soil C and N in the surface (0-10 cm) mineral soil (Bauhus et al., 2004). Although it increased soil pH and the mineralisation rate of carbon from forest floor, liming had similarly little effect on soil carbon storage in spruce plantation ten years after application (Nilsson et al., 2001). Unfortunately, an assessment of the complete carbon balance of the ecosystem was not achieved in either of the studies, making it impossible to realise how far the enhanced biomass growth could compensate the carbon loss by faster organic matter decomposition, lime decarbonatation and associated fossil carbon loss due to lime extraction, transport and application.

4.3.3. Tree species

4.3.3.1. Species selection.

Species differ in their capacity to build up carbon stocks in biomass and soil. Picea stands have comparatively higher amount of C in the forest floor than the other species (Giardina, 1988). Differences in the potential of carbon sequestration, defined as the rotation-average net primary production, have also been established in comparative trials among Pinus species (Balboa-Murias et al., 2006), faster growing species such as Pinus radiata sequestering more carbon. Coniferous species tend to accumulate carbon in upper layers of he soil, due to their shallow root system, whereas broadleaved trees input carbon deeper in the soil profile on average (Jandl et al., 2007). No clear difference for the total amount of soil carbon between these group of species emerge even if forest inventory data suggest that some species may accumulate more carbon in soil and biomass (Table 4.3).

Table 4.3. Wood density of European tree species and median of C pools in European forests (de Vries et al., 2003).

Species	**Wood density [kg/m^3]**	**Tree C [t/ha]**	**Soil C [t/ha]**	**∑C [t/ha]**
Pinus sylvestris (Scots pine)	490	60	62	122
Picea abies (Norway spruce)	430	74	140	214
Abies alba (Silver fir)	410	100	128	228
Fagus sylvatica (beech)	680	119	147	266
Quercus sp. (oak)	660	83	102	185

4.3.3.2. Understorey vegetation

The ground vegetation is a target of forest management especially because it competes with trees especially during the regeneration phase and may also represent a risk for fire initiation and propagation. On the other hand, the ground vegetation may also play a role of shelter for young trees, retain nutrients in forest environments prone to leaching and stabilise the upper soil horizon and enhance mechanical impedance of soil. Understorey removal decreased the net primary production of the ecosystem and fine root production which, in turn, decrease the soil carbon. The removal of vegetation beneath trees is profitable for wood production and storage in the tree biomass pool as far as windthrown risks is not increased. However, no effect on soil carbon has been found in slash pine plantations (Shan et al., 2001). Balance between the positive and negative imapcts of the understorey vegetation on the carbon storage is therefore very much depending onthe local site and management conditions.

4.3.4. Stand management and harvesting

Carbon tends to accumulate in soil, forest floor and biomass during the life cycle of the ecosystem and reach maximal values in old-growth stands (Sogn et al. 1999, Böttcher and Springob 2001, Cerli et al., 2006, Guillet et al., 2007). Forest floor C increased during the first decades after a disturbance while mineral soil C seems age-independent, at least at the time scale of a century (Agren et al. 2007). As expected, aboveground ecosystem C increased also with age due to an increase in aboveground tree biomass whereas belowground ecosystem C remained similar in the early decades after establishment and increased later (Peichl and Arain, 2006). From a comparison of the average rate of carbon net uptake between different *Pinus* species , thinning regimes and rotation length Balboa-Murias et al. (2006) demonstrated that thinning intensity and rotation duration exert opposite effects on carbon stocks. For radiata pine, C sequestration in total aboveground tree biomass for the whole rotation (thinnings and clear felling at 30 years) ranged from 3.4 t ha^{-1} year^{-1} (the lowest initial stocking density, the worst site quality and 35% of thinning intensity) to 5.9 t ha^{-1} year^{-1} (the highest initial stocking density, the best site quality and 15% of thinning intensity). For maritime pine, the amount of tree biomass at stand level was significantly lower than in radiata pine stands, and mean annual carbon storage ranged from 2.3 to 4.6 t ha^{-1} year^{-1} for the same rotation length. Reduction of the thinning intensity and perhaps extension of the rotation length were proposed as possible strategies when the main objective is biomass production and C sequestration.

Eriksson and Johansson (2006) have compared different broadleaved species among 93 stands growing in abandoned farmlands in Northern Europe showed that optimal rotation length was depending on the age at which growth and production are maximal. They concluded that long rotations are beneficial if the objective is to maximize the average carbon stock in biomass. If, on the other hand, the intention is to optimize reductions in atmospheric CO_2 emissions, rotations should be short for aspen, silver birch and grey alder stands. Similar result was established for maritime pine by Loustau and Dupouey who compared the climate effect of management alternatives accounting for both biogeochemical and biophysical components (Table 4.4 a and b). These authors concluded that the substitution of fossil carbon by wood products predominate on the long-term and render the intensive forestry with a high productivity more cooling than the less intensive scenario. However, they also stress that the parameterisation of the model used, with a replacement factor 1:1 and a large productivity range was clearly conditioning the conclusion ot their modelling experiment. A conservative conclusion is that fertile sites may be devoted to higher production forestry whereas unfertile sites may better value low intensity management alternatives.

Table 4.4. Impact of three management scenarios on carbon stocks (a), and relative global climate impacts (b) partitionned among biogeochmical, fossil fuel substitution and radiative (albedo) effects for complete rotations of maritime Pine (Pinus pinaster, Ait) in southwestern France (from Loustau et Dupouey, 2005). Global climate impact are expressed as a unitless index relative to the reference scenario. A cooling effect is counted negatively.

(a)	**Mean carbon stock**			**Mean annual increment**	**Fossil C replaced**
	Biomass		Soil		
	(tC.ha^{-1})			(m^3.ha^{-1}.an^{-1})	(tC.ha^{-1}.an^{-1})
Management intensity	trees	other			
Low (5 thinnings, 75 yr rotation)	57.5	7.5	80	6	1.3
Reference (5 thinnings 45 yr rotation)	27.7	5	60	12	2.6
High (2 thinnings, 25 yr rotation)	14.2	0	40	18	3.9

(b)		Relative climate impact (°C)			
integration time		Carbon storage	Fossil carbon	Albedo	**Total**
20 y					
	High	0.16	-0.11	-0.01	**0.04**
	Low	-0.22	0.11	0.01	**-0.10**
50 y					
	High	0.16	-0.27	-0.01	**-0.12**
	Low	-0.22	0.27	0.01	**0.06**

4.3.4.1. Thinning and harvest operations

There is no evidence of long-term effects of thinning and harvests on soil carbon, as far as whole tree harvesting is not concerned (Johnson and Curtis, 2001; Misson et al., 2005; Vesala et al., 2005; Jandl et al., 2007) see also section 3.6. Thinning modifies the microclimate at the soil surface generally improving temperature and soil moisture and decreases temporarily the litterfall. It may therefore deplete the forest floor carbon pool (Piene and Van Cleeve, 1978, Aussenac, 1987) unless thinning residues are left on site (De Wit and Kvindelsaland, 1999). For Picea abies in Denmark, data from a long-term thinning experiment and an adjacent spacing experiment at stand ages of 58 and 41 years, respectively, showed that C stocks of all biomass compartments decreased with increasing thinning grade, while the distribution between compartments was hardly influenced. Thinning influenced the C stock of the forest floor and mineral soil oppositely, resulting in no effect of thinning on total soil C (Skovsgaard et al., 2006).

The effects of harvest management have been compared in old growth beech forests in Europe and this survey showed that the impacts of the investigated moderate silvicultural practices on the carbon budget of European beech forests were lower than those reported for clear cuttings in temperate forests (Mund and Schulze, 2006). A review of harvesting techniques made by Johnson and Curtis (2001) suggests that the effect on soil C is small and depends most on the residues management: left on site, burning, none (whole tree harvesting). However, carbon loss by 5 to 20tC/ha have been measured following harvest by Pennock and Van Kessel (1997). Chronosequence studies support the hypothesis that carbon tends to accumulate in soil, forest floor and biomass during the life cycle of the ecosystem and reach maximal values in old-growth stands (Sogn et al. 1999, Böttcher and Springob 2001, Cerli et al., 2006, Guillet et al., 2007). This implies that shortening rotation may diminish the steady state value of carbon stock in forest ecosystems.

4.3.5. Risk and natural hazards

The carbon sequestration is more efficient as the turn-over time of the accumulating pools is longer and reciprocally, less efficient when it is shortened. Biotic and abiotic hazards may reduce substantially the residence time of carbon in the biomass and forest floor pools. Minimising risk exposure and ecosystem vunerability is therefore desirable in the context of sustainability of the forestry-wood chain.

In geographic regions with a low human density, natural wildfire is one of the major regulators of the carbon stock in forests, together with windstorms. Impacts of fire depend upon their intensity. While severe fires mineralise most of the biomass stock, a variable part of the ground floor and soil stocks and lower the photosynthetic carbon uptake to zero, light fires do mainly transfer some carbon from the biomass to the soil without subsequent depletion in productivity (Wang et al., 2001).

The effect of windstorms on the carbon and greenhouse gas balance of managed forest ecosystems is poorly documented. Depending on the severity and damages, the totality or part of the trees felled may be harvested, this at a higher cost in fossil fuel than for a normal harvest operation. Restoration of the site may also induce additional costs linked to stump and slash removals and soil preparation. Conversely, a high amount of slash and boles is usually left on site and increases the soil and forest floor carbon stocks (Thürig et al., 2005).

Next decades will be characterised by a rapidly changing climate. While this may not be so important for short-term silvicultural scenarios, wood-biomass and intensive forestry, this non-stationarity must be accounted for in management for longer rotation scenarios because important shift in species area and yield class are expected (Loustau et al., 2005). Tree species will be more vulnerable and affected at their southern margin and substitution by southern vicariants may be considered there.

4.4 Forest management alternatives

Thornley and Cannell (2000) used a mechanistic forest ecosystem simulator, which couples carbon, nitrogen and water (Edinburgh Forest Model) to mimic the growth of a pine plantation in a Scottish climate according to thinning and harvesting regimes as follows. The model was run to equilibrium (1) as an undisturbed forest, (2) removing 2.5, 10, 20 or 40% of the woody biomass each year (3) removing 50% of the woody biomass every 20 years, and (4) clear-felling and replanting every 60 years as in conventional plantations in this climate. More carbon was stored in the undisturbed forest (35.2 kg C m^{-2}) than in any regime in which wood was harvested. Plantation management gave moderate carbon storage (14.3 kg C m^{-2}) and timber yield (15.6 m^3 ha^{-1} $year^{-1}$). Notably, annual removal of 10 or 20% of woody biomass per year gave both a high timber yield (25 m^3 ha^{-1} $year^{-1}$) and high carbon storage (20 to 24 kg C m^{-2}). The efficiency of the latter regimes could be attributed (in the model) to high light interception and net primary productivity, but less evapotranspiration and summer water stress than in the undisturbed forest, high litter input to the soil giving high soil carbon and N_2 fixation, low maintenance respiration and low N leaching owing to soil mineral pool depletion. They concluded that there is no simple inverse relationship between the amount of timber harvested from a forest and the amount of carbon stored. Management regimes that maintain a continuous canopy cover and mimic, to some extent, regular natural forest disturbance are likely to achieve the best combination of high wood yield and carbon storage in the context of pure even-aged coniferous plantations but vey few data is available so far for checking this hypothesis (O'Hara et al., 2007). Management shifts carbon allocation toward commercial timber production. A comparison of the ratio of wood biomass to NPP among a database assembled recently by Janssens and Luyssaert (pers. com.) exemplifies this result for 229 forest sites (Luyssaert et al., 2007). The carbon allocation among biomass compartment is shifted toward the commercial timber at the expense of other compartments like roots, branches and foliage.

The question regarding the balance between fossil carbon emissions associated with silvicultural intensification (herbicide, fertilisation, thinning, harvesting, etc.) and the net gain in carbon sequestration in biomass and soils due to productivity enhancement was addressed recently (Liski, 2001; White et al., 2005; Markewitz, 2006; Sonne, 2006). The fossil carbon emissions could be estimated to 3 Mg C ha^{-1} over a 25-year rotation in an intensively managed pine plantation in the southeast USA indicating that fossil C emissions from silviculture would largely counter-balance 75% of the expected gains in soil C (16 Mg C.ha^{-1} over 100 years) or in pulp products due to added productivity (Markewitz, 2006). In contrast, the growth, harvest, and utilisation of saw logs as timber appear to provide a clear benefit for C sequestration, 35 Mg C.ha^{-1} over 100 years, relative to the C emissions incurred from intensive silvicultural activities, 12 Mg C.ha^{-1} over 100 years. The major implication of this analysis is that the fossil C emissions from intensified silvicultural activities can impact the net amount of C sequestered in managed forests but a net C gain should still be realized, particularly if trees are allowed to grow to a saw log category. A comprehensive assessment of the greenhouse gas balance of 408 management regimes of intensive forestry has been recently proposed by Sonne (2006) for Douglas fir planted forests in the Pacific Northwest (USA). This study is among the first assessments based on a life cycle assessment approach and accounting for upstream as well as on site carbon emissions for such a range of management regime (Johnson et al., 2005). It concludes that carbon emissions associated with management practices are significant, accounting from 6 to

12.5% of the wood carbon storage according to scenarios, longest scenarios being less carbon-expensive. Upstream emissions, associated with e.g. seedlings, fertiliser and phytocide production and transportation, were 16% and on site emissions 84% of the total. The biggest contribution was log transportation followed by harvesting, site preparation and fertilisers. The author concludes there is an opportunity to enhance carbon sequestration in forests through minimising management emissions.

Forest management orientated towards production tends to intensify the stand productivity, e.g. through fertilisation, efficient species, vegetation management, site preparation. It keeps the forest stand in an actively growing state through thinning and shortened rotation duration. As compared with a more ”close from nature” management, the intensification of silviculture shifts the carbon cycle towards a maximisation of harvested products at the expense of soil and biomass which generally leads to net carbon loss in the atmosphere[1] because the turn over time of wood products is shorter than biomass and soil. The carbon stocks *in situ*, in soil and biomass, are relatively low compared to a low intensity scenario.

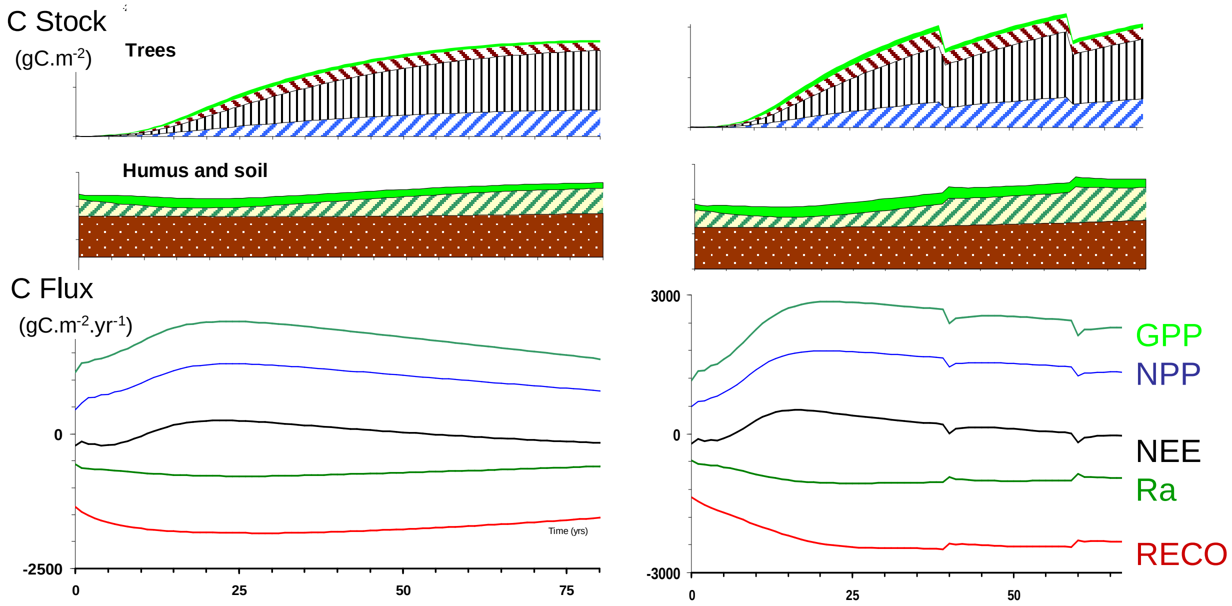

Figure 4.2. Course of carbon stock (upper diagrams) and flux (Lower diagram) in a forest ecosystem according to two management class, unmanaged (left) and managed (right).

4.5 Synthesis

Forest ecosystems represent the largest terrestrial storage of carbon and there is an increasing evidence that human activities are controlling the carbon cycle in forests at the global scale through direct and indirect effects (Wang et al., 2001; Magnani et al., 2007). From this review, we conclude that the management effects on the carbon cycle in continental forest are considerable. The impacts of forest management on atmosphere and climate are therefore a key issue of the sustainability of the forestry wood chain.

Direct and short-term effects are easier to observe and quantify and therefore generally well understood (Table 4.5). The carbon storage in the biomass of managed forests has been managed for long by foresters and is relatively well understood, even if interactive effects of management, climate, disturbances and pollution are not that clear. A key issue here is the question of the balance between on site biomass storage and fossil carbon replacement by wood products which is tightly depending on the integration time used.

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[1] Not considering the fossil fuel substitution by wood products use.

Table 4.5. Qualitative effects on average carbon stocks of management operation in managed forests over a rotation compared with the rotation period prior to management (modified from Freeman et al. 2005). From Hyvönen et al. (2007).

	Soil C stock	**Biomass C stock**	**Ecosystem C stock**
Stand initiation phase			
Prescribed burning [1]	Decreasing	Decreasing, neutral or increasing	Decreasing, neutral or increasing
Drainage of peatlands [2]	Decreasing	Increasing	Decreasing, neutral or increasing
Site preparation method [3]			
Low-intensive	Neutral	Increasing	Increasing
Intensive	Decreasing	Increasing	Decreasing, neutral or increasing
Tree species change [4]			
To conifers from broadleaves	Increasing	Increasing	Increasing
To broadleaves from conifers	Decreasing	Decreasing	Decreasing
To mixed conifers and broadleaves from mono-specific coniferous	Neutral or decreasing	Neutral or decreasing	Neutral or decreasing
Stem exclusion phase			
Thinning method [5]	Neutral or decreasing	Decreasing	Decreasing
Fertilisation [6]	Increasing	Increasing	Increasing
Increased rotation length [7]	Decreasing, neutral or increasing	Increasing	Increasing
Harvesting method [8]	Decreasing, neutral or increasing	Decreasing, neutral or increasing	Decreasing, neutral or increasing

[1] Biomass and ecosystem C stocks depend on regeneration success and nutrient loss.

[2] Ecosystem C stock depends on loss of from soil and gain in biomass.

[3] Ecosystem C stock after intensive preparation depends on loss from soil and gain in biomass.

[4] Soil C depends on the decomposition rate which is generally lower for conifer litter than for broadleaf litter. Biomass and ecosystem C depend on the growth rate which is higher over longer periods for many conifer species than for broadleaf species.

[5] Removal of thinning residues can give large transient reductions in soil C.

[6] Higher production increases all C stocks. N-fertilisation increases litter production and may reduce decomposition in soil in the long-term.

[7] All C stocks depend on developmental stage of the forest when the rotation period is prolonged and on the thinning method.

[8] Soil C depends on the removal of residues. Biomass and ecosystem C for the following rotation period depend on how regeneration and growth conditions are affected by the harvesting operation.

Comparatively the soil part of the carbon cycle became only recently a matter of interest in forestry but relatively few experiments are operational so far. Apart from the nitrogen fertilisation, the soil carbon stock is either depleted or left unchanged by management operation depending on the case studied. Inhibition of mineralisation and enhancement of litter production by nitrogen may lead in some cases to an increase in soil carbon stock.

- Logging and site preparation deplete the soil and humus carbon stocks. Management intensification may decrease the soil carbon stock by as much as 50% through more frequent logging, drainage, and soil preparation operations.
- The fossil carbon emissions associated with management operations is higher for more intensive scenarios and has the critical drawback of extracting carbon from a reservoir with an extremely long residence time, as compared with the other forestry wood chain pools. At most, it may offset 6 to 12% of the carbon stored in the wood compartment.

Indirect and longer term effects are less documented and therefore difficult to integrate in management. They are however critical for managing the carbon cycle in forest ecosystems because the main consequence of management is a change in carbon allocation among compartments differing in residence time by one to two order of magnitude.

Carbon is a widely used indicator because stocks in biomass, wood products and even soil are measurable at low cost. However, as far as the global climate is concerned, the information provided by inventorying and comparing carbon stocks among management alternatives is only partial. Indeed, the consumption of fossil carbon associated with forest management and downstream transformations must be accounted for as well as the other greenhouse gases, methane, nitrous oxide, ozone. As far as land use change is an issue, the biophysical role of forests must also be taken into account. The approach of life cycle assessment, allowing completeness and comparability, must also be favoured in the future to provide a complete picture of the climate impacts of management scenarios.

5. The impact of forest management on the water cycle of forest stands and watersheds

by

Klaus Katzensteiner, Emil Klimo and Ute Szukics

5.1. Introduction

One of the services of forested watersheds is the provision of pristine water in sufficient quantity. The effect of forest cover on runoff processes has also a tremendous influence on the protective function of forest ecosystems. Thus a thorough understanding of management impacts on the water cycle is of crucial importance for sustainability assessments.

The influence of forests on hydrological processes at different spatial and temporal scales has been a matter of debate for centuries (Chang, 2006). Scientific watershed research on the effects of forests on the water cycle started in the late 19th century by pioneers in forest hydrology like Engler in Europe (Keller, 1988) or Bates and Henry in the US (McCulloch and Robinson, 1993; Douglass and Hoover, 1988). The famous paired catchment study in the Swiss Emmental already showed increased annual runoff in a sparsely forested compared to a fully forested catchment. Furthermore the regulating effect of forests on flow rates: a decrease of peak flow in rainy periods and an increase of base flow during dry periods as well as effects on snowmelt were documented (Engler, 1919). Since then a large number of catchment studies have been established. Measurements of runoff at weirs give integral results on water yield for whole catchments.

The first review by Hibbert (1967) of 39 experimental catchments has been updated subsequently by Bosch and Hewlett (1982) with additional 55 experiments. More recent reviews by Stednick (1996), Sahin and Hall (1996), Best et al. (2003) or MacDonald and Stednick (2003) confirm more or less the general patterns gained already in the first investigations: a reduction of water yield by increasing forest cover and a decrease if forest cover is reduced. They also could show differences in the response between conifer-type forests and deciduous hardwoods and make general quantitative predictions of management effects on water yield. Climatic variation, different patterns and dynamics of regrowth and site conditions in general cause a high variation of the response of water yield to treatments. The response of the size of peak flows is even more complex, though a general extenuating effect of forests can be concluded.

Besides impacts on species composition and forest cover and structure, forestry also alters the hydrological regime of watersheds by construction of roads, skid trails and landings. In catchment scale studies a clear separation of the effect of road construction from the effects of stand treatment is usually impossible. Direct surface runoff from forest roads is mainly directed to ditches draining into stream channels. In sloping terrain in addition subsurface components of runoff may be intercepted. Both components are delivered rapidly to the stream network and can increase size of peak flows. The discussion on significance of these effects is however still controversial (MacDonald and Stednick, 2003).

To uncover effects of forest management on sub-processes of the water cycle, measurements at stand scale, combined with micro-meteorological measurements and models like Bowen energy balance ratio or eddy co-variance modelling are appropriate methods. A number of studies treat the effects of tree species on canopy interception and intercepted precipitation evaporation, showing the clear differences between coniferous and deciduous hardwood species (Brechtel and Pavlov, 1977). For montane forests even a considerable net gain of precipitation due to interception of cloud and fog-water in forest canopies could be observed (Grunow, 1955). Measurements of xylem sap flux give information on transpiration rates of individual trees and allow the scaling up for whole stands (Cermák and Nadezhdina, 1998). Changes in soil water storage can be measured by several methods, Time and Frequency Domain Reflectometry (TDR and FDR) becoming a convenient tool to gain measurements in high spatial and temporal resolution. TDR has been successfully applied to describe the drying and refilling patterns of forest soils in high resolution and revealed distinct differences between spruce and beech (Schume et al., 2003).

Regression based models derived from statistical analysis of results gained from direct measurements are one way to predict possible management impacts on hydrological processes. A more promising way is the use of mechanistic hydrological models for the prediction of evapotranspiration and runoff. Accuracy of runoff predictions at a watershed scale for areas where long-term time series of meteorological data and runoff values can be used for calibration of models is usually high. Hydrological models have been used to make predictions on effects of afforestation on water yield (Feger and Wang, 2007), to estimate the impact of a change of tree species on runoff (Armbruster et al., 2004) or as a support tool for silvicultural decision making in face of changing climatic conditions (Katzensteiner and Schume, 2007).

Complex meteorological situations, in particular in mountainous terrain, spatial variability of site conditions, and missing homogeneity of forest stands make precise forecasts on management effects by the use of models still rather uncertain.

The current report should give an overview on the present state of knowledge on impact of forestry on water yield and modification of peak runoff. A statistical analysis of literature data serves as a basis for the general quantification of management impact upon water yield. For generic forest ecosystems a modelling approach has been applied to allow for a ranking of the impact of different management options on evapotranspiration.

5.2. Concepts and indicators

The impact of forest management upon water budget of forest stands/watersheds will be evaluated based on the terms of the water balance equation.

$$Po = Es + I + T + \Delta SW + \Delta GW + Q$$

Po…Precipitation, Es…Soil evaporation, I…Intercepted rain evaporation, T…Transpiration, ΔSW…delta Soil water storage, ΔGW…delta Ground water storage; Q…runoff; all units in [$mm.a^{-1}$].

With respect to the service ‘water yield’, water use by the ecosystem, namely EIT (E_s+I+T = EIT) is a suitable indicator. Numerous process models of different complexity and hydrological field experiments are available to estimate this term. A second indicator will be runoff, as this value is frequently given as an output of paired catchment studies. This term will be meaningful under the assumption that over longer periods, ΔSW + ΔGW will equal zero.

With respect to the protective function of forested systems also the impact upon surface runoff and peak discharge is of importance. As peak discharge is dependent on a number of factors – timing and intensity of precipitation, snow accumulation and snowmelt patterns, site factors like watershed size and topography, surface roughness and soil properties (porosity, pore size distribution and continuity), water content of soils prior to a precipitation event and vegetation cover, a generalization of quantitative results gained from watershed investigations is problematic.

Different approaches will be used to express the hydrologic response to the impact of harvesting. The analysis of hydrographs from paired catchment studies (control versus treated watershed) gives information on *response patterns*.

Rainfall-runoff coefficients for the simple *rational equation* are an option to compare forest to other land use types.

$$Q_p = k\ C\ I\ A$$

where Q_p [$m^3.s^{-1}$] is a peak discharge for a given rainfall intensity, I is the rainfall intensity, A is the catchment area, C is the runoff coefficient (ratio of runoff/rainfall) and k is a factor for unit conversion (in the example 0.00278 if I [$mm.h^{-1}$] and A [ha]). A uniform rainfall intensity is assumed until time of concentration (time required for water to travel from the furthest part of a watershed to the outlet). Still C will be a function of topography, soil properties and vegetation.

5.3. Methods and approaches

5.3.1. Review

The current study includes a literature review of publications about the effects of forests and forest management upon hydrological processes. The review includes results on management effects on runoff from single and paired catchment experiments, stand level data on sub-components of the water cycle and information on physiological response of trees (leaf area and stomatal conductance) to fertilisation. With respect to the service ‘reduction of peak runoff’ a crude review of published experiments and runoff coefficients is included.

5.3.2. Meta analysis of literature values

Paired basin experiments provide a continuous, and continuously changing record of vegetation structure, composition, climate, and their effects on streamflow (Jones and Post 2004). Single or paired catchment approaches have been applied for more than a century and a considerable literature has been generated. A literature research on paired catchment studies provided a broad range of data, comprising reviews (Hibbert, 1967; Bosch and Hewlett, 1982; Stednick, 1996) which were completed by more recent studies (Robinson, 1998; Robinson and Dupeyrat, 2005; Troendle et al., 2001; Jones and Grant, 1996; Iroumé and Huber, 2002; Pike and Scherer, 2003).

In course of the construction of a meta-database, empiric data for water yield, evapotranspiration and interception before and after forest management were collected. This approach was based on two different scales, either the whole catchment or just a managed stand was considered. Water yield data were exclusively coming from watershed studies while evapotranspiration and interception data referred to the stand scale.

The available data were classified hierarchically regarding the affiliation to the Köppen-climate classification, the size of the investigated drainage area, the vegetation type and the management practices. Classification according to (Köppen, 1931) distinguishes 5 climate zones: tropical (A), arid and semiarid (B), temperate (C), continental (D) and polar (E) climates. The rather rough classification according to Köppen was chosen, as a finer categorisation was hardly possible due to the lack of detailed information in many studies and reviews. The focus was laid on the climate zones C, temperate and D, continental climates.

The response of runoff to change of forest cover from paired catchment experiments was analysed separately for deciduous hardwoods and for conifers by using multiple linear regression analysis with mean annual precipitation and percent area cut as independent variables. A simple multiple regression analysis provided sufficient results (after outliers from earlier databases used by other authors could be identified as erroneous values not conforming to original data). Residuals were high, but normally distributed. Thus it was not necessary to use fuzzy regression techniques like has been done by Sahin and Hall (1996).

5.3.3. Modelling approach

To visualize the effect of tree species, rotation length and thinning regime, the hydrological model BROOK90 (Federer, 1995) was used on generic forest systems. The growth and yield scenarios were provided by Eckmüllner (personal comm.).

Three scenarios for the rotation of Norway spruce according to long-term experiments of the Austrian Federal Forest Research Center (http://bfw.ac.at/rz/bfwcms.web?dok=3554) were selected: afforestation with 2500 stems per ha and heavy pre commercial thinning and early thinning (scenario intensive forestry), leading to a rotation of 80 years to reach a target diameter of the central tree of 40 cm, afforestation with 3500 stems per ha, precommercial thinning and selection thinning leading to a rotation of 100 years and a third scenario with afforestation of 2500 stems per ha and moderate thinning from below starting rather late leading to a rotation length of 120 years (scenario multi-purpose forestry). Leaf areas have been calculated for the different development stages according to Eckmüllner (2006). Maximum canopy conductance per unit leaf area was set to 0.25 and albedo to 0.14 from an age of ten years onwards. To simulate the effect of clear-cut and herbaceous vegetation in the clear-cut phase, a decreasing conductance from 0.4 to 0.25 in the first 10 years and an increasing albedo from 0.15 to 0.23 in the first four years, decreasing then to 0.14 over the next six years were assumed according to Katzensteiner (2000).

One scenario for European beech – a rotation length of 125 years, thinning from below with different intensities and natural regeneration by applying shelterwood cuts in year 110 has been calculated (scenario low intervention). The scenario of stand development was applied after Kennel (1972), leaf area was estimated by allometric functions from Hietz et al. (2007). A maximum canopy conductance per unit leaf area of 0.53 and an albedo of 0.18 was kept constant throughout the rotation.

Soil data for mesic site conditions: a cambisol with sandy loam and a species specific rooting pattern (shallow rooted spruce and deep rooted beech) have been used according to Katzensteiner and Schume (2007).

To allow for climatic variation a 25 years time series of climate data from Eastern Austria, including both wet and dry years has been applied to each stage of development (Figure 5.1). Thus it was possible to get the response of the terms of the water balance in both wet years and extremely dry years.

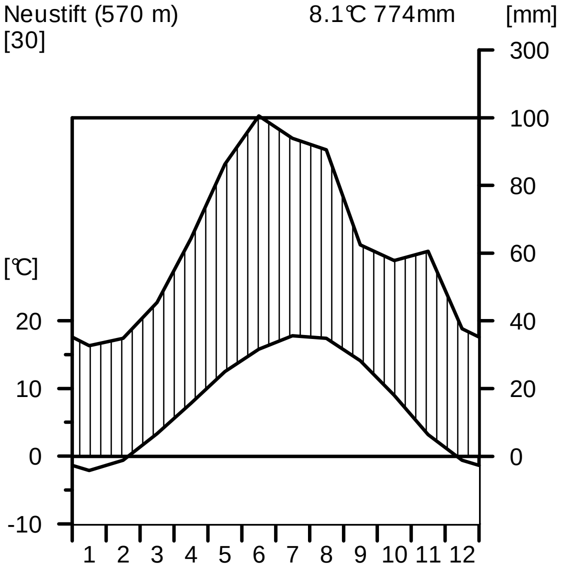

Figure 5.1. Climate diagram of the meteorological station used for scenario modeling.

5.4. Results and discussion

5.4.1. Choice of tree species

5.4.1.1. Intercepted rain evaporation

The evaporation of precipitation intercepted by forest canopy is closely related to vegetation cover and vegetation type. The amount of intercepted and transpired water highly depends on tree species and their leaf-area index (LAI).

Differences in intercepted rain evaporation between tree species are well studied, in particular for spruce and beech (Benecke, 1984; Brechtel, 1970) and their relevance concerning interception, water balance and microclimate could be prooved in many studies (Augusto et al., 2002). The famous Solling study showed a comparatively high intercepted rain evaporation of spruce (29%) compared to beech (17%) (Benecke, 1984; Ellenberg et al., 1986). The higher interception rate of Norway spruce can be attributed to higher leaf area, both in summer time and particular in winter. The study of Kantor (1995, transcription of Klimo, 2007 personal communication) also showed higher interception losses in mature coniferous stands (16%) compared to broadleaved forests (7 %) but these values are much lower than the Solling results. Of course there is a pronounced year to year variation depending on climatic conditions (Table 5.1).

Species-referred differences are modified by stand age (Molchanov, 1960), stand structure and forest management (Mitscherlich, 1981; Hager, 1988).

Table 5.1. Water balance of a spruce and a beech stands in the Orlicke Mountains (Cz.) in the hydrological years 1976/1977 – 1980/1981 (Kantor, 1995).

Hydrological year	Precipitation of an open area	Interception	Transpiration	Evaporation	Surface runoff	Horizontal soil runoff	Seepage	± soil moisture							
	mm	mm	%	mm	%	mm	%	mm	%	mm	%	mm	%	mm	%
Spruce stand															
1976/1977	1263.6	190.4	15.1	234.2	18.5	84.8	6.7	2.4	0.2	11.4	0.9	743.3	58.8	-2.9	-
1977/1978	1187.0	192.3	16.2	199.9	16.8	74.5	6.3	11.5	1.0	22.7	1.9	670.2	56.5	15.9	0.2
1978/1979	1071.0	226.3	21.1	165.1	15.4	97.5	9.1	19.3	1.8	21.2	2.0	546.0	51.0	-4.4	1.3
1979/1980	1500.0	264.3	17.6	184.2	12.3	62.3	4.2	13.9	0.9	24.1	1.6	944.8	63.0	6.4	-
1980/1981	1460.5	187.4	12.8	192.3	13.2	82.1	5.6	17.0	1.2	15.4	1.1	961.3	65.8	5.0	0.4 0.4 0.3
Mean	1296.4	212.1	16.3	192.2	15.1	80.2	6.2	12.8	1.0	19.0	1.5	773.1	59.6	4.0	0.3
Beech stand															
1976/1977	1263.6	73.0	5.8	202.2	16.0	82.4	6.5	20.3	1.6	14.8	1.2	872.8	69.1	-1.9	-
1977/1978	1187.0	54.7	4.6	175.0	14.7	73.4	6.2	18.0	1.5	18.2	1.5	843.4	71.1	4.3	0.2
1978/1979	1071.0	92.4	8.6	160.7	15.0	90.6	8.5	14.1	1.3	13.5	1.3	698.8	65.2	0.9	0.4
1979/1980	1500.0	102.9	6.9	173.7	11.6	54.6	3.6	29.3	2.0	27.6	1.8	1108.9	73.9	3.0	0.1
1980/1981	1460.5	110.3	7.6	192.4	13.2	82.8	5.7	25.4	1.7	26.8	1.8	1019.1	69.8	3.7	0.2 0.2
Mean	1296.4	86.6	6.7	180.8	13.9	76.8	5.9	21.4	1.6	20.2	1.6	908.6	70.1	2.0	0.2

Besides the impact upon total interception losses tree species also modify spatial patterns of canopy throughfall and stemflow. While trees with a funnel like crown and smooth bark like beech guides a high share of precipitation as stemflow close to the trunk (up to 20% of precipitation), conifers like spruce rather guide precipitation to the periphery of the crown (Peck, 2004 Beier, 1998). For spruce rarely more than 2% of precipitation reaches the forest floor as stemflow, except in very young stands. Pine shows stemflow values around 3% and Douglas fir even up to 8%.

The generalisation: higher interception losses in coniferous stands is not valid for every situation, no clear effect of tree species was found for example by Huber and Iroumé (2001) in temperate Chilean areas with high precipitation rates. In this case for coniferous plantations interception ranged from 11 to 39% of precipitation and in comparable broadleaved forests interception varied between 10 and 37% of precipitation.

Crockford and Richardson (2000) reviewed current knowledge of the effect of forest type, ground cover and climate on interception, covering a wide range of forest types and climate. Characteristics such as trees.ha^{-1}, branch angle, the uniformity or lack of uniformity in crown height, the nature and thickness of the bark layer, leaf shape and inclination, and leaf area index are all influencing interception. They assumed that it is difficult to draw general conclusions about interception losses by particular forest types and climates because characteristics of a forest that affect interception are not easy to identify and quantify and data almost always depend on the type of rainfall and other meteorological conditions during the study period.

Jost et al. (2004) focused on the role of tree species upon horizontal and vertical patterns of soil water recharge. The horizontal and vertical spatial distribution is altered due to both the throughfall and stemflow patterns and due to plant soil feedback mechanisms: Higher interception of spruce results in lower soil water recharge when compared to beech. In addition the higher macro-porosity of soils under beech lead to a quicker vertical distribution and more even soil rewetting compared to spruce.

5.4.1.2. Soil evaporation

Soil evaporation in mature stands is usually of less importance than interception losses. As in many studies intercepted rain evaporation and transpiration of a forest floor vegetation is included in this term, results of different studies are frequently not comparable! In the study of Kantor (1995) soil evaporation amounts to 6% for mature spruce and beech stands. During clear-cut phases and early stages of stand development there will be a pronounced effect of this term however. In a study by Katzensteiner (2000) soil evaporation during vegetation period at a clear-cut amounted to 10-20% of precipitation, compared to only 2% in the mature spruce stand. Müller et al. (2002) compared pine and beech stands in the NE-German lowlands under precipitation limited condition of forest growth. In the pine stand soil (and ground vegetation) consumed as much as 35% of precipitation compared to only 12% in the beech stand. In summary total EIT was similar for all situations and exceeded precipitation (Table 5.2).

Table 5.2. Water consumption of pine and beech stands in the NE German lowlands during vegetation period [mm] (Müller et al., 2002).

	Precipitation	I	T	E	EIT
Pine 84 a	360	104	148	126	378
Pine/beech 51/11a		83	220	72	375
Beech 101 a		86	256	44	386

5.4.1.3. Transpiration

Transpiration rates clearly depend on tree leaf surface areas (Santiago et al., 2000), vary according to levels of radiant energy, soil moisture, humidity, wind, and stomatal resistance imposed by vegetation (Pike and Scherer, 2003) and differ with tree species and canopy structure (Köstner, 2001).

Different from interception, plants are able to regulate transpiration in response to environmental conditions. Thus the effect of species on transpiration may be less pronounced as for interception losses. The comparison of transpiration rates sof spruce and beech in Kantor’s study shows only a minor difference between mature stands. Transpiration of replanted young-growth beech and spruce stands at the same site investigated by Kantor (1995) however show generally higher rates in the spruce stand compared to the beech stand, the difference increasing over time due to tree growth (Figure 5.2).

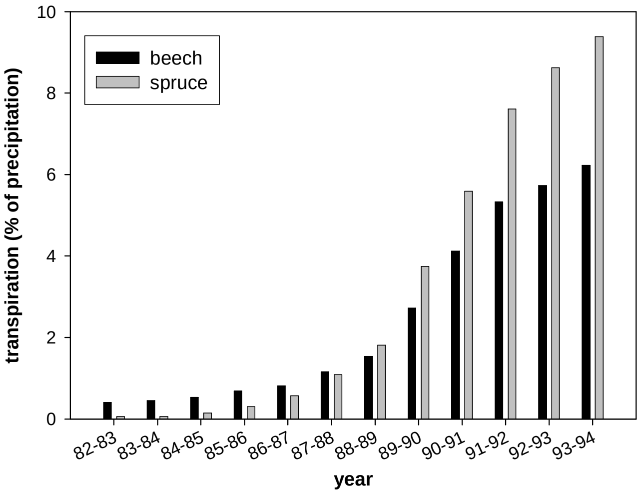

Figure 5.2. Transpiration rates in regrowing beech and spruce stands (Kantor 1995).

Contrary conclusions were drawn by Hietz et al. (2000). A considerably higher transpiration rate of beech trees than spruce trees of comparable DBH was measured. They assumed that coniferous trees were more economic in the consumption of water than broad-leaved trees. Schipka et al. (2005) report high values of canopy transpiration of beech for Central Europe of around 210 – 350 mm, showing no clear dependence on precipitation regime, in a review of several studies. Experimental results were obtained by different methods, sap flux measurements, soil water balance and micrometeorological approaches.

Coniferous and deciduous trees further differ in rooting depth (Canadell et al., 1996) which is strongly determining the water supply during precipitation-free periods and thus transpiration rate. Moreover deciduous angiosperms are leafless during the winter, so the potential transpiration varies more widely in deciduous forests than in coniferous forests as a function of seasonal dynamics in leaf area (Moore et al., 2004).

Besides the variation of transpiration-rates in relation to tree species, changes of transpiration with increasing tree age appear to be relevant.

While canopy LAI seemed to be a suitable predictor of canopy transpiration in European beech forests independently of stand age, canopy transpiration/LAI decreased with increasing stand age in stands of Norway spruce (Köstner, 2001). Ryan et al. (2000) reported also a higher transpiration per unit leaf area of 12 m high, 40 years old, compared to 36 m high, 290 years old, Ponderosa pine trees – a hint on physiological changes of leaf conductance with tree age.

5.4.2. Site preparation – fertilisation

Fertilisation may influence ET by alteration of leaf area and by changing stomatal behaviour. In addition indirect effects on water uptake by changing shoot to root ratio, rooting pattern and conductivity of the xylem have been shown (Ewers et al., 2001).

Ewers et al. (2001) compared effects of fertilisation and irrigation on stomatal conductance of young *Pinus taeda* and *Picea abies* stands grown on nutrient poor soil. Fertilisation doubled respectively tripled leaf area of *P. taeda* respectively *P. abies*, regardless if irrigation was applied. In fertilized, non irrigated *P. taeda* stands canopy stomatal conductance decreased sharply, while the combination of irrigation and fertilisation caused a large increase in stomatal conductance. In *P. abies* fertilisation caused no change in stomatal conductance unless irrigation was applied.

Earlier reports by Guehl et al. (1995) gave evidence of reduced stomatal conductance after fertilisation of *Pinus pinaster*. Ripulline et al. (2004) confirmed an increase of WUE after N fertilisation for *Pseudotsuga menziesii* and *Populus x euroamericana,* while they found no evidence for a change of stomatal conductance. There are however a number of studies showing either no or even a negative effect of N fertilisation on WUE.

Hubbard et al (2004) showed a tremendous increase of water use due to fertilisation in *Eucalyptus saligna* stands, also an effect of increased leaf area, while on the long run canopy conductance did not change. In this case water use efficiency could not be improved by fertilisation.

In a study by Powers and Reynolds (1999) *Pinus ponderosa* showed increased WUE due to fertilisation only if drought stress was not too extreme – a clear indication of dependence of fertilisation effects on site conditions.

In summary the increasing leaf area due to fertilisation should lead to higher interception losses, even if transpiration is not increased due to improved stomatal control.

5.4.3. Stand management and harvesting

5.4.3.1. Rotation, silvicultural regime

To visualize the effect of choice of tree species and thinning regime, different scenarios have been calculated using the BROOK90 model (Figure 5.3).

The modeling results indicate a distinct difference between beech and spruce over the rotation length. EIT of beech remains rather constant from year to year and the reaction to thinning operations is moderate. This is partly due to the fact that beech has a rather plastic canopy, and leaf area is recovering quickly after thinning. Under the situation of unlimited water supply the transpiration of beech is rather high compared to spruce, in the dormant season the effect of stand density on evaporation is only moderate. In case of water limitation the EIT of beech is however lower than EIT of spruce (except for the heavily thinned variant) – a result of lower interception rates (Table 5.3).

Table 5.3. Scenario results of EIT and water yield for spruce and a beech stand for low, average and high precipitation rates.

	Minimum			Average			Maximum		
Scenario	Precipitation	EIT	Water yield	Precipitation	EIT	Water yield	Precipitation	EIT	Water yield
Spruce A	587	386	153	787	453	335	1148	502	752
Spruce B		416	126		490	298		540	713
Spruce C		425	122		504	284		554	695
Beech		400	162		491	297		570	668

In contrast to beech the development of leaf area in the moderately thinned spruce stand peaks earlier and shows a steady decline from age 25 onwards. This is reflected in the behavior of EIT. Reduction of leaf area by thinning reduces water consumption continuously. For a whole rotation in this case water consumption between early and heavily thinned stands with short rotation and stands thinned moderately from below differs as much as 50 mm.a^{-1}.

Comparing a spruce stand and a beech stand with similar LAI's and heights for the site conditions mentioned above, one can clearly see the difference in partitioning of available water in different seasons. The higher water consumption of Norway spruce compared to beech confirms with values from the literature (Table 5.4).

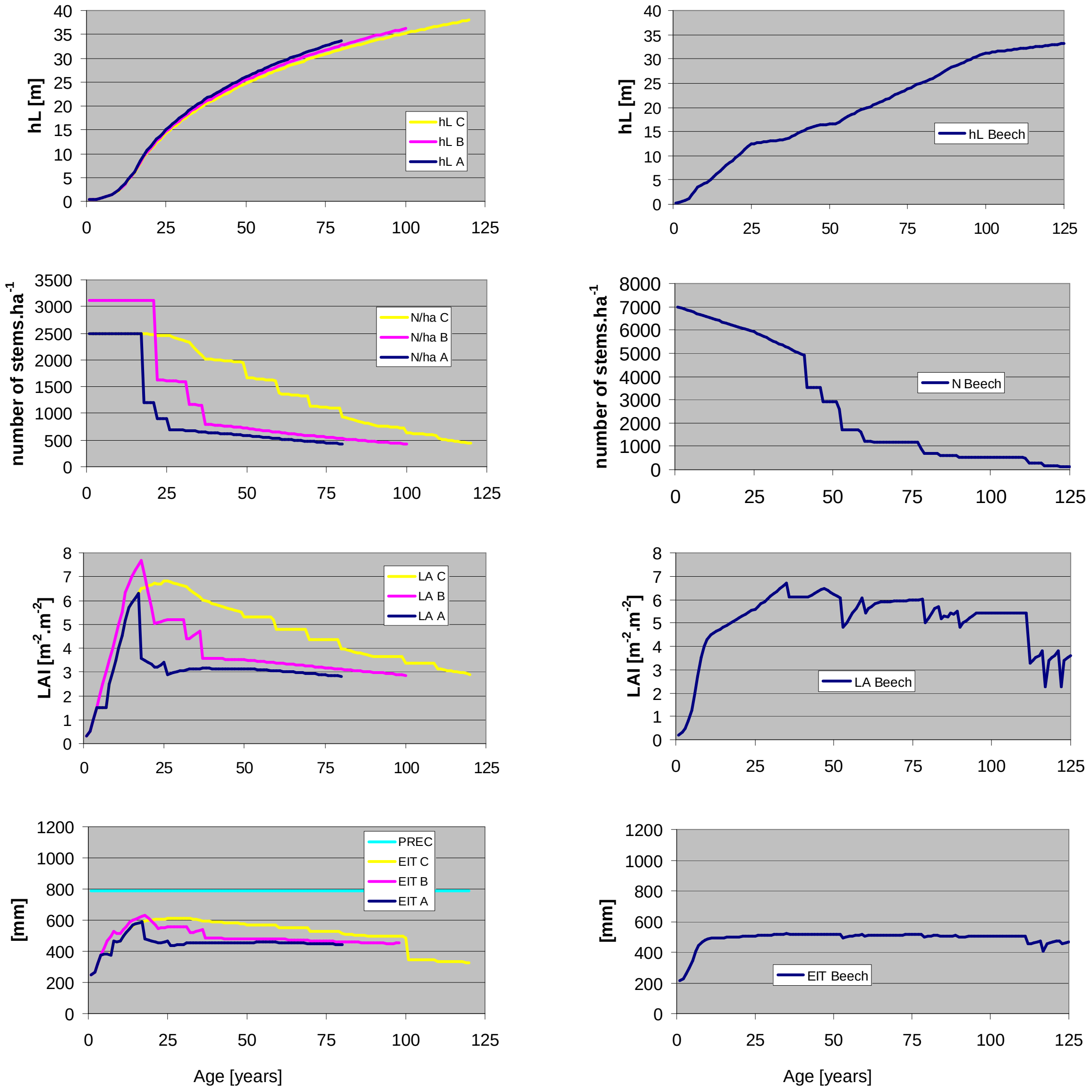

Figure 5.3. Stand development of a spruce stand with different degrees of thinning and different rotation (A: early and heavy pre-commercial thinning, 80 years rotation; B: less intensive selection thinning, 100 years rotation; C: moderate thinning from below, 120 years rotation) and a beech stand (thinning from below, shelterwood cut at age 110): hL…mean stand height [m] (Lorey), N…stem number, LA…leaf area index; and EIT: Evapotranspiration [mm] modeled by BROOK90 for a precipitation of 800 mm.

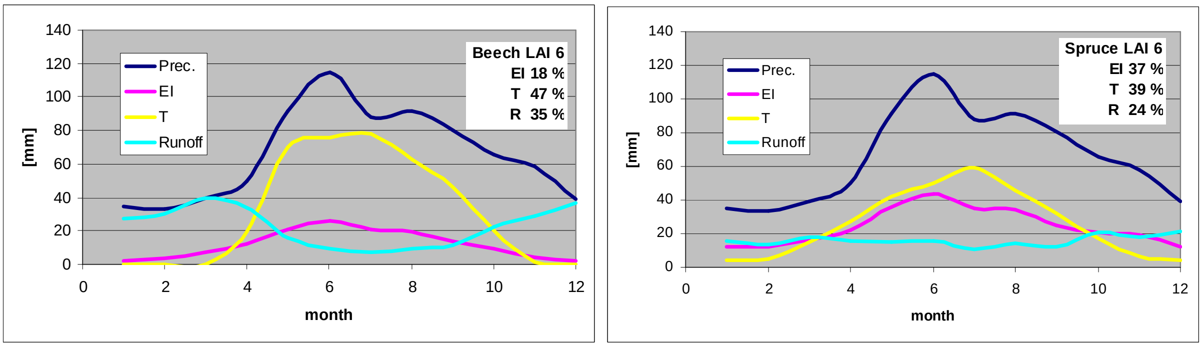

Figure 5.4. BROOK90 model results of the water cycle for a spruce stand and a beech stand with LAI 6.

Table 5.4. Water cycle of European forest ecosystems: Comparison of model results for generic systems of European beech and Norway spruce and literature values.

Reference	Site		Precipitation	E	I	EI	T	ET	**EIT**	surface runoff	interflow	seepage	**R**
Benecke 1984	Solling	beech	1060		187			287	**474**			589	**589**
	1969-1975	spruce			305			335	**640**			423	**423**
Klimo, 2007	Orlické hoty Mts., Cz	beech	1296	77	87		181	258	**345**	21	20	909	**950**
after Kantor, 1995	1976/77-1980/81	spruce		80	212		192	272	**484**	13	19	773	**805**
Model results	Generic system	beech LAI 6	787			141	374		**515**			273	**273**
	25 a time series	spruce LAI 6				288	308		**596**			191	**191**

E	soil evaporation (including ground cover)
I	intercepted rain evaporation
T	transpiration
R	total runoff

For continuous cover forestry, a rather even water consumption over the years is assumed, leading in total to higher EIT values as for age class forests over a rotation.

5.4.3.2. Thinning

Intercepted rain evaporation

The removal of intercepting surfaces through thinning results in a decrease of intercepted rain evaporation and consequently a higher percentage of precipitation is reaching the forest soil. Depending on the degree of LAI reduction and canopy opening by the forester a modification of the climatic characteristics occurs to a greater or lesser extent. After thinning of a pine stand and underplanting with deciduous trees, (Knoche, 2004) observed a decrease of interception by 8% of annual precipitation (729mm) to 30% as a consequence of the strong reduction of stand basal area. On a watershed scale interception loss of at of a beech-spruce-fir forest in the Bavarian Alps of a fully stocked stand was 20-30%. Thinning of 55% lead to a bisection of interception loss (Breisameter, 1996).

Hager (1988) observed a close relation of remaining stem number and interception after pre-commercial thinning of Norway spruce stands. With increasing tree number an exponential rise of interception and an exponential decay of throughfall is shown in Figure 5.5.

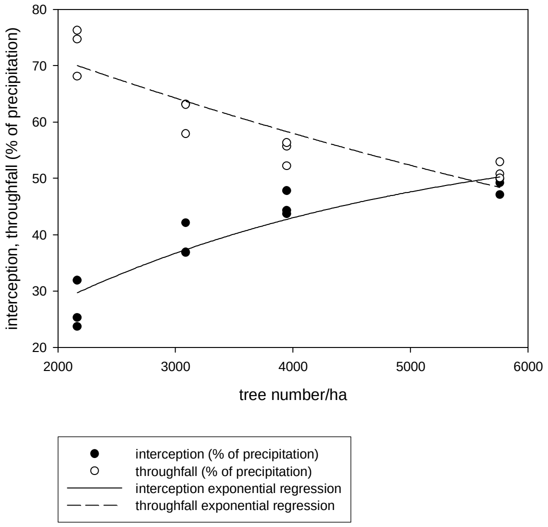

Figure 5.5. Relation of intercepted rain evaporation and stem number after pre-commercial thinning of Norway spruce (Hager, 1988).

Intercepted snow

As snow storage is often an order of magnitude larger than that for rain (Lundberg and Halldin, 2001), canopy dimension is an important factor determining the water balance in snow-dominated areas.

Snow course measurements in northern Finland's boreal forests showed that interception loss from gross precipitation increased with forest density and approached 30% for a forest with the highest density class (Lundberg and Koivusalo, 2003). Storck et al. (2002) studied snow interception and canopy effects on snow accumulation and melt in a coniferous stand in Oregon. Over the study period, approximately 60 % or snowfall was intercepted by the canopy. Sublimation rates were however low and amounted to 100 mm per winter season. Approximately 72% and 28% of the remaining intercepted snow was removed as meltwater drip and mass release.

In general the inverse relationship between snow accumulation and forest density causes differences in snowmelt patterns and peak runoff.

Soil evaporation

Evapotranspiration of soil and vegetation is affected by the increased amount of solar radiation and precipitation reaching the ground as a result of forest thinning (Pike and Scherer, 2003). In thinned areas, higher light levels correspond to an increased exposure to evaporative forces. Light, temperature and wind are increased, compared to shaded areas. Molchanov (1960) reported an increase of evaporation of 0.8% as a result of 15-25% thinning.

Transpiration

Either an increase as well as a decrease of transpiration after thinning is reported. In general forest stands have higher evapotranspiration rates than other vegetation types and so any opening in the cover results in a reduction in the amount of water consumed and transpired (Aussenac, 2000). It is believed that a decrease of transpiration rates is the consequence of forest thinning due to a reduction of transpiring surfaces.

Lower transpiration rates of a *Quercus robur* stand after thinning were measured by (Vincke et al., 2005). The Leaf Area Index had decreased from 4.1 in the control plot to 3.4 due to thinning.

Though the transpiration of individual trees may increase due to thinning, like shown in a study by Morikawa et al. (1986) in a 31-year old *Chamaecyparis obtuse* stand, whole stand transpiration will usually decrease. The author's explanation for increased single tree transpiration is an increase of the foliage biomass per tree and greater canopy exposure after thinning.

A higher physiological activity of thinned stands in the long-term was also shown by Peterson et al. (1997). Thinned loblolly pine stands had a slowed live crown recession and increased crown diameters resulting in increased photosynthetic surface area per tree. The crown diameters of thinned threes increased 82 % while the crown diameters of control trees increased only 20% over a 6 year study period.

Water yield

The removal of intercepting surfaces of the forest canopy affects the generation of runoff directly and results in higher water availability which contributes to soil moisture and/or streamflow. Most catchments show an initial increase in annual yields after thinning, followed by an unsteady decline depending on the rate and type of revegetation. A crown cover reduction from 60 to 14% resulted in an increase in streamflow of approximately 20% of annual rainfall after 3 years, compared to a streamflow yield of 6% annual rainfall before thinning (Ruprecht et al., 1991).

The initial increase of water yield after forest harvesting is directly related to the basal area removed (Cornish, 1993). So it can be assumed that the magnitude of the effect after thinning will also depend on the basal area removed.

Peak runoff

Snow accumulation and melt have a significant influence on hydrological processes in high latitudes like montane and boreal forest environments. Peak flows are affected by removing canopies as a consequence of altered snow accumulation and melt. Canopy removal of a coniferous catchment in

Colorado increased precipitation reaching the forest floor by about 40%, the peak snowpack water equivalent increased by 35% (Stottlemyer and Troendle, 1999). A thicker snowpack accumulates after removal of forest canopies which leads to an altered rate of spring snowmelt and a shift of timing of peak water yield.

5.4.3.3. Harvesting

Forests differ from cleared land in two hydrologically significant ways: they (1) have high rates of evapotranspiration (ET) and (2) their soils allow rapid infiltration of rain water (Giambelluca, 2002). After forest harvesting more precipitation is reaching the ground due to the lack of interception and transpiration. The consequence is a lower soil moisture deficit which means that less rainfall is required to commence streamflow (Bari et al., 1996). In addition the alteration of soil physical properties due to compaction may impact runoff patterns. A paired catchment study, setup in a Eucalypt forest in Western Australia showed that the spatial average saturated hydraulic conductivity was considerably lower for the harvested catchment. According to these results the authors suggest that logging had a significant long-term impact on the hydraulic conductivity of surface soil.

Transpiration

The application of forest management practices reduces the maturity of the forest stand. As water use is determined by tree age, large differences between old and young stands occur. (Moore et al., 2004) found that transpiration in young stands is increased compared to old growth forest. This finding provides evidence that forest management alters site water balance via elevated transpiration in young-kept stands.

As stands age changes in the amount and vertical distribution of leaf area occur. Old trees have lower whole-tree leaf-specific hydraulic conductance than young trees of the same species. The leaf area index (LAI) typically increases with stand age, but it declines in some late successional forests (Ryan et al., 1997). Species-related differences in water use contribute to a decreased stand-level water use with stand age. For example in old forests, shade-tolerant species, which have a reduced water usage per unit sapwood are more abundant. Surprisingly, Granier et al. (2000) did not find significant differences in forest transpiration when comparing beech stands of different age.

Water yield

(MacDonald and Stednick, 2003) explained that the primary effect of forest harvest is a change in evapotranspiration which drives a change in runoff. The reduction of interception losses and an elimination of transpiration cause an increase of water yield for the first years until the clear-felled area becomes re-vegetated.

Hibbert (1967) reviewed 39 catchment experiments and assumed that a reduction of forest cover generally increased the water yield. The response to the treatment was highly variable and, for the most part, unpredictable.

Stednick (1996) reviewed paired catchment studies in the USA. He assumed that changes in annual water yields from harvesting of less than 20% catchment area or forest cover cannot be determined by streamflow measurements. Further he summarised a 25 mm increase in annual yield after 10% reduction of a forest stand. Reviewing 94 catchment experiments, Bosch and Hewlett (1982) assumed a general 40 mm increase in annual yield for every 10% reduction in conifer forest cover. An annual increase in stream flow by 28 mm after 90% clearance of pine plantations in Queensland were reported by Bubb and Croton (2002). The study of Robinson and Dupeyrat (2005) shows that felling of mature coniferous forests in upland Britain can have measurable effects on river flows even if the size of the cut area was generally less than 5-10% of the catchment.

The increase in streamflow following deforestation is due to a decreased interception by the crowns, reducing the evaporative losses. A decrease in canopy interception storage is associated with a higher infiltration of precipitation and increased water yield might be the result. By investigating mountane cloud forests in Latin America (Bruijnzeel, 2001) concluded that the clearing of forests results in an increase in the total volume of streamflow, typically by 100-400 mm/year. A catchment study of an

Eucalypt forest in Western Australia showed an increase in annual streamflow duration and magnitude after harvesting. Streamflow peaked one year after logging (~18% of annual rainfall) due to increased throughfall following the logging (Bari et al., 1996).

Hornbeck et al. (1993) generalised that the effect of forest cutting was an initial increase in water yield which occurs promptly after the cutting. The magnitude is related to the percentage of reduction in basal area as found in previous summaries too (Douglass and Swank, 1972; Bosch and Hewlett, 1982).

A comparison of 11 watersheds suggested that reductions in basal area must approach 25 % to obtain measurable responses in annual water yield. Above that threshold there is some variability in first-year responses among watersheds with similar basal areas cut (Verry et al., 2000). The differences usually can be explained by factors such as configuration and timing of cutting, location of cutting in relation to the stream channel or source area, and whether regrowth was controlled. The results of catchment studies (Figure 5.6) were used for a meta-analysis.

As shown in earlier studies below a threshold of 20 % area cut no clear response of runoff can be detected (Figure 5.7). The slope of the partial regression for water yield increase as a percentage of area cut is similar to earlier analyses and hardly differs between deciduous hardwoods and hardwoods and conifers. There is however a clear difference in the response to precipitation. At high annual precipitation rates the effect of forest cover is more pronounced for conifers than for hardwoods – probably an effect of the high intercepted rain evaporation of conifers.

Forest removal increases the water yield, the subsequent return towards pre-treatment streamflows and below is associated with forest growth rate, canopy cover and soil depth (Cornish, 1993). Under natural regrowth water yield increases diminish rapidly, nearly disappearing within 3-10 years in most cases, but lasting for decades in some situations (Hornbeck et al., 1993). Results from a paired catchment study in Western Australia showed that as vegetation regenerates all the streamflow components decrease (Bari et al., 1996). The increases of water yield after deforestation can be prolonged by controlling the natural regrowth for example by herbicide application.

Peak flow
As living trees consume large amounts of precipitation, the harvest of trees causes an increase of water available for forest streams. The effect of forest harvesting on stormflows might depend on the size of the catchment. A literature review with regard to land use impacts on hydrologic regimes by (Kiersch, 2000) shows an observable impact on average flow and peak flow just in basins smaller than 100 km². The role of catchment sizes on riparian and hillslope dynamics was investigated by (McGlynn et al., 2004). They found no clear relationship between catchment size and new water contribution to runoff in storms, but the lag times of responses increased systematically with catchment size.

Robinson et al. (2003) investigated forest impacts on peak and low flows in a European context and collected and analysed data from 28 small basins across Europe. Different stages of a plantation forest cycle from site preparation before planting to post-felling conditions were studied. It was assumed that complete clear felling could increase moderate peak flows immediately downstream.

Data from the Glenturk catchment (Ireland) showed a delay of the hydrograph peaks in a 15-year old conifer forest compared to an 8-year old stand and a stand after forest felling. The flow per hour peak was lowest in the 15-year old stand (Figure 5.8).

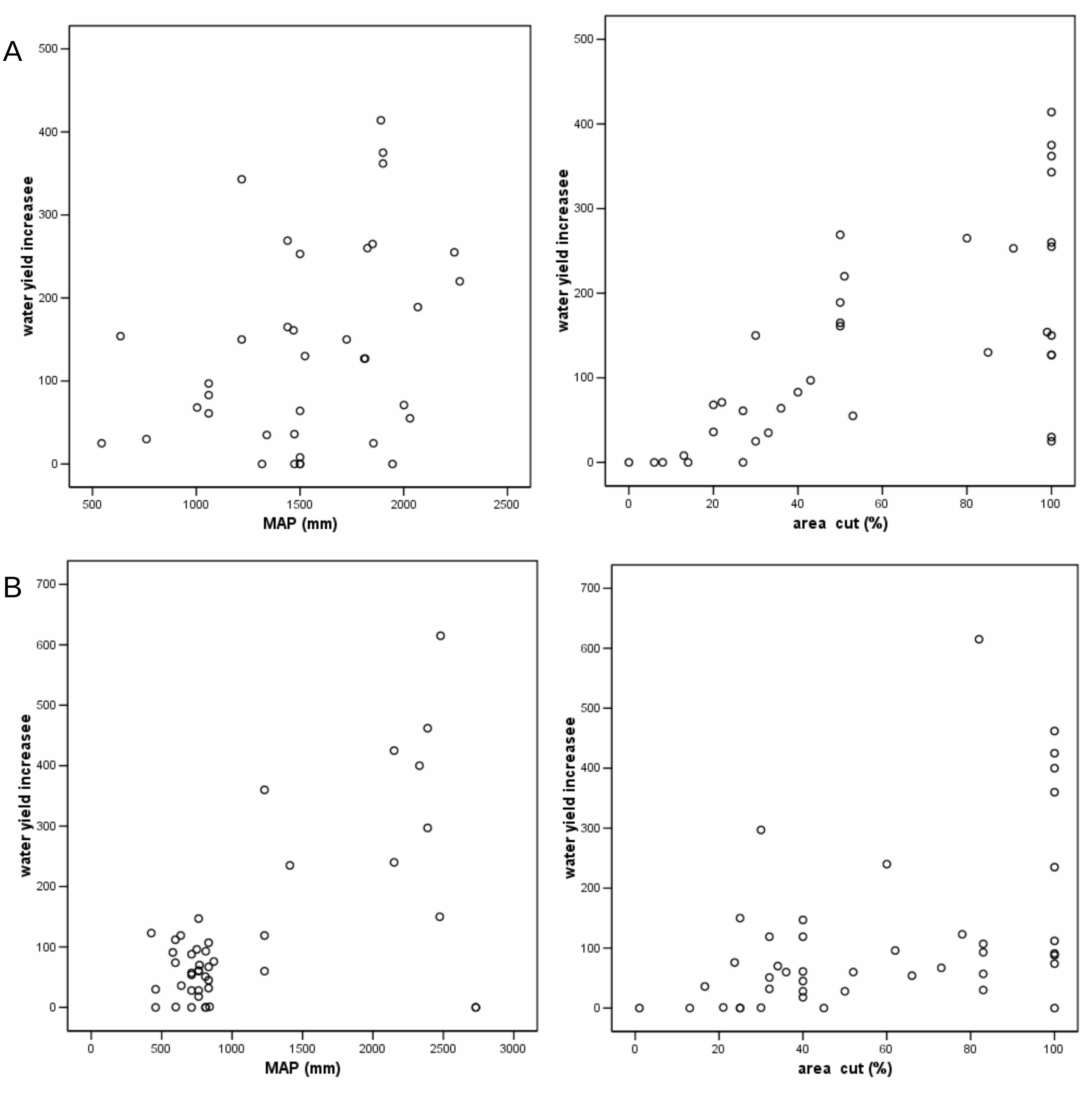

Figure 5.6. Mean annual streamflow as a function of mean annual precipitation and percentage of area cut for catchments with A. deciduous hardwoods (n=36) and B. coniferous forests (n=43) (Meta-analysis of data from earlier reviews (Best et al. 2003; Bosch and Hewlett 1982; Sahin and Hall 1996; Stednick 1996 and MacDonald and Stednick 2003).

An early study by Harr and McCorison (1979) measuring the size and timing of peak flows in Western Oregon showed however that clear-cut logging reduced the size of an annual peak flow by 32%. A delay of all peak flows of nearly 9 hours could be observed after clear-cutting. These results were attributed to the site specific snowmelt patterns. In contrast to that, Robinson and Dupeyrat (2005) found no evidence that forest felling of mature coniferous forests in upland Britain had a significant influence on peak flows. And forest harvest (23.7%) in the Coon Creek watershed, a 1673 ha catchment (Wyoming) did not cause a significant increase in peak discharge (Troendle et al., 2001).

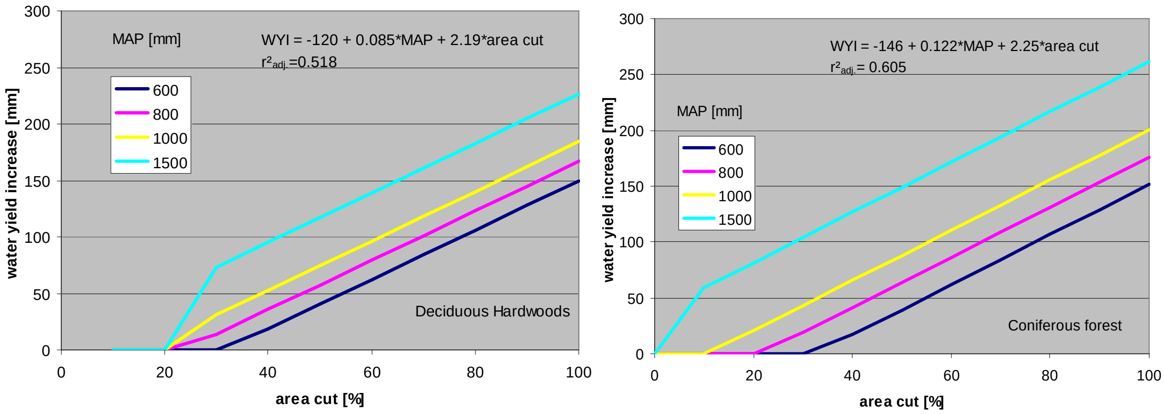

Figure 5.7. Results of a multiple linear regression of water yield increase (WYI) as a function of mean annual precipitation (MAP) and percentage of area cut (Regression only valid for precipitation > 600 mm and percent area cut > 20 %!) for watersheds covered with deciduous hardwoods or coniferous forests respectively. Response is visualized for four MAP levels.

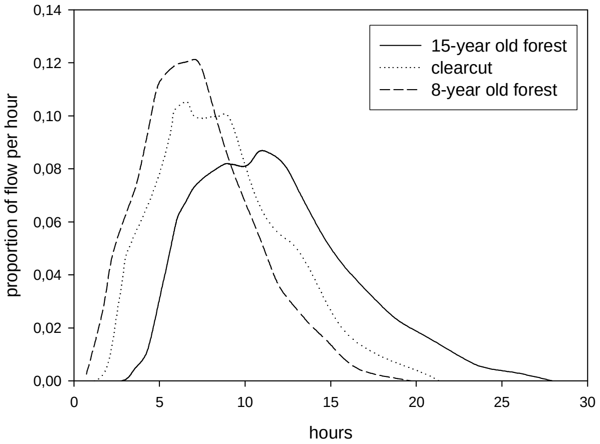

Figure 5.8. Storm unit hydrographs for drained coniferous forest (after Robinson et al. 2003).

5.4.4. Vegetation changes

Concerning vegetation changes, the conversion of grassland (cropland) to forest and vice versa often modifies evaporative water losses as a result of plant effects on water demand and supply. Calder (2003) indicates that interception losses are higher from forests than shorter crops primarily because of increased atmospheric transport of water vapour from their aerodynamically rough surfaces.

Van der Salm (2002) discusses the effect of afforestation of arable land controversially. The points of reduced groundwater recharge on the one hand, the reduced risk of flooding and the reduced demand for artificial drainage of lowland areas on the other hand are rised. Recent studies of chronosequences of afforestations in the Netherlands (Van der Salm et al., 2005) and in Scandinavia (Rosenqvist et al., 2007) confirm higher EIT of Norway spruce compared to common oak stands of comparable age. In the study of van der Salm et al. (2006) water recharge declined from 485 mm per year at the arable land to 172 mm in the 18 year old spruce and to 100 mm in 13-14 year old spruce stands. A comparison of effects of afforestation of arable land is available in Heil et al. (2007).

Nosetto et al. (2005) investigated eucalyptus tree plantations in Argentina and found out that they evaporated 80% more water than grasslands. Tree plantations therefore caused higher evaporative water losses and so the vegetation type has a key role in the hydrological cycle.

Afforestation tends to reduce groundwater recharge and net water availability because the trees intercept part of the precipitation and, owing to their deeper root system, transpire more water than grasses (Iroumé and Huber, 2002).

On the other hand a decreased peak runoff but a higher continuous base flow under forest cover is expected.

5.4.5. Other management options

5.4.5.1 The role of watershed management with respect to peak runoff

Compared to other land use types forest areas show considerabely lower runoff coefficients (Table 5.5). This is of particular importance for the question of torrent and stormflow prevention.

Table 5.5. Runoff coefficients for storm-return periods less than 25 years by hydrologic soil groups B (sandy loam soil), C (clay soil) and watershed slope range (0-2%, 2-6% and >6%], selected values from (Chang, 2006).

		B			C	
Land Use	0-2%	2-6%	>6%	0-2%	2-6%	>6%
Pasture	0.18	0.28	0.37	0.30	0.40	0.50
Meadow	0.14	0.22	0.30	0.24	0.30	0.40
Forest	0.08	0.11	0.14	0.12	0.16	0.20

Surface runoff under various scenarios of management Tyrolean Alps was measured by (Markart et al., 2007). Runoff was determined with small scale sprinkling experiments. On sites with rough surface structure like forest ecosystems and alpine dwarf-shrubs, peak flows were lower and delayed compared to alpine meadows and intensely managed areas. The results showed that forested soils hardly ever reach the water holding capacity even under sprinkling with high precipitation amounts and precipitation intensity.

The modelling study of the upper Schesa catchment (0.781km²), located in Vorarlberg near Bludenz demonstrates peak-flow differences under 4 land-cover variations (Table 5.6). Scenario 1 is intensely grazed by cattle and horses (current condition), Scenario 2 implies further deterioration, Scenario 3 represents low-to free from grazing conditions and partly forest closure, Scenario 4 is completely forested with trees and shrubs (Markart et al., 2006). It was assumed that the rougher the surface of a site was, the lower is the portion of the surface runoff.

Table 5.6. Peak runoff ($m^3 s^{-1}$) for four different scenarios in the Schesa catchment (Markart et al., 2006).

	Scenario			
	1 (intensely grazed)	2 (deteriorated)	3 (low grazing intensity, partly forested)	4 (forested)
peak runoff ($m^3 s^{-1}$)	8.1	8.6	6.2	4.2

5.4.5.2 Road construction

After road construction (Jones and Grant, 1996) detected changes in peak discharges in basins ranging up to 600 km² in the Western Cascades, USA. The increases are attributable to changes in flow routing rather than to mere changes in water storage due to vegetation removal. Though a synergistic effect may exist between forest roads and harvest as related to their effects on flows The addition of roads to clear-cutting in small basins produced a quite different hydrologic response than clear-cutting alone, leading to significant increases in all sizes of peak discharges in all seasons (Jones and Grant, 1996). The results support the hypothesis that roads interact positively with clear-cutting to modify water flow paths and speed the delivery of water to channels during storm events, producing much greater changes in peak discharges than either clear-cutting or roads alone.

Road construction combined with patch clear-cutting ranging from 10-25% basin area produced significant, long-term increases in peak discharges (Jones and Grant, 1996). (MacDonald et al., 2001) found that relatively undisturbed vegetated hillslopes generate runoff only during the largest storm events. In contrast, unpaved roads commonly generate runoff when rainfall exceeds 6 mm.

The concentration of runoff from nearly impervious road surfaces and intercepted subsurface flow into ditches effectively increases the drainage density, shifting the distribution of water on hillslopes and potentially increasing peak flows of streams (Luce and Cundy, 1994).

5.5. Synthesis

Forests, compared to other land use types are characterized by considerably higher water consumption. Thus, under condition of limited precipitation, afforestation may reduce water yield compared to other land use options. In situations where stormflow generation is a problem, afforestation may decrease peak runoff and thus prevent erosion and formation of mudflows.

Forest management itself has different impacts depending on the scale under observation.

On a stand and a small watershed scale harvesting regime will have the highest impact on water consumption and runoff patterns. Clear-cutting more than 20 percent of a watershed will increase runoff rates in a proportional relation to precipitation amount and cleared area. Depending on site conditions and regrowth rates, the impact may last from few years if forest regenerates quickly, to decades if the site is heavily disturbed.

In addition the construction of roads, skid trails and landings, as well as soil compaction from harvesting operations may increase peak runoff.

Choice of tree species has a moderate impact upon water consumption, but alters the pattern of snow distribution and interception. While evergreen coniferous forests have rather high interception throughout the year, deciduous forests have lower interception in the dormant season. This effect is partly compensated by higher transpiration rates in the vegetation period. There may however be pronounced plant-soil-feedback mechanisms. Deeper rooting and higher activity of soil macro-fauna can improve soil structure under beech compared to spruce or pine, thus leading to improved infiltration rates and decreased surface runoff.

Increased water consumption due to fertilisation may occur due to increased interception losses by increased leaf area. The effect is considered small.

On the scale of a forest enterprise the water consumption may be influenced by rotation length and thinning regime. Maintaining lower stand closure due to heavy and early pre-commercial thinning and selection thinning, leading to shorter rotation length, may reduce water consumption continuously compared to stands thinned from below.

6. The impact of forest management on water quality

by

Per Gundersen

6.1. Introduction

Clean freshwater is essential for life. In Europe pressures from intensive agriculture, industry, urbanisation, tourism and climate change suggest that guaranteeing water quality for humans and ecosystems will continue to be difficult and costly (EEA, 2005). In this context water from forest are now increasingly valued as natural and clean water resources. Water resources from forests are already in use for drinking water purposes in several regions and in numerous of the major European cities. The natural protective function of forest with respect to water quality and water-related hazards rely on the continuous vegetation cover and the relative low intensity of human intervention. It is indeed documented that forest streams and groundwater is of good quality compared to water from other land uses (e.g., US-EPA, 1995; Callesen et al., 1999; Gundersen et al., 2006; Thornton et al. 2000). In Northern Europe, afforestation on agricultural land takes place as part of a strategy to restore protective functions and thereby improve water quality and to secure water resources in the future (Hansen et al., 2007).

In Europe, the increase in air pollution over the last 4-5 decades is recognised as a threat to the quality of waters from nature and less intensively managed land such as forests, although significant reductions in emissions have occurred (EEA, 2005). At the same time, the use and management of forested land (forest harvest, fertiliser and lime addition, ditching, road building, soil preparation, weed control, change of tree species etc.) has also intensified (Worrell and Hampson, 1997) and may thus to some extent degrade the protective function of forests on waters. Strategies on forest management for water quality protection are currently discussed (Rothe et al., 1999; Brooks and Lust, 2000; 2001). With new demands for biofuels, use and management of forests may be further intensified. To be able to predict and possibly mitigate losses of water quality, operational knowledge about forest management measures is needed.

In the EFORWOOD project the ambition is to be able to predict an array of impacts from scenarios of change in forestry, including quantification of impacts on the protective function of forsts such as on the quality of forest waters. As a first step this chapter focuses on the impact of a range of individual management operations (listed in Table 1.2) on water quality in order to identify and illustrate which management options may have the most impact on water quality and under which conditions. Our analyses build on existing data syntheses and reviews, but for several topics we have complied new tables on the water quality response to management. The second step will be to aggregate the effect of the individual operations into evaluations of the impact of management alternatives (preliminarily defined in Table 1.3). This process will run in parallel with the further detailing of the management alternatives in EFORWOOD; however, our preliminary evaluation is presented in chapter 7.

6.2. Concepts and indicators

In the EFORWOOD module 2.1.1 specific indicators were identified and described (Raulund-Rasmussen et al., 2006). For water quality the identified indicators were the concentrations of nitrate, dissolved organic carbon (DOC), pH, dissolved aluminium, heavy metal and xenobiotics as the main issues/indicators related to the quality of water (Table 1.1). For heavy metals and xenobiotics the compounds need to be specified for a further analysis of the impact of forest management on these compounds.

The indicators relate to water leaving the forest ecosystem, i.e. seepage water and small streams, and are thought to articulate the protective function (or potential loss hereof). However, the impacts associated with elevated concentrations of the indicator compounds are mainly observed in "down-stream" water receptors such as streams, lakes, rivers, groundwater and coastal marine waters. For the impacts in streams and small lakes, changes in indicators may be relevant at short timescales (days to weeks) and small spatial scales (hectares), whereas for large lakes, rivers and marine waters the loads of indicator compounds over large areas and longer timescales (year to decade) are relevant. For groundwater the size of the reservoir is important for the relevant spatial scale but often the impact will only be relevant at timescales from decades to a full forest rotation. For the water quality indicators we thus have to consider the short-term and small spatial scale as well as the long-term and broad scale effects of forest management operations.

Knowledge on the impact of forests and management operations will be of increasing importance as the EU regulation related to water quality the Water Framework Directive (WFD) is implemented in the member states. The basis for WFD is river catchments (or groundwater reservoirs) where the main principle is that water bodies should be restored to a good ecological and chemical status by 2015. Thus the contribution and quality of water from forests in catchments with significant forest cover needs to be quantified. The quality of water from forests will also relevant in relation to the Nitrate Directive and the Drinking Water Directive.

The indicators cover a relative broad field of issues and environmental impacts. In this chapter on the impact of individual forest management operations we focus on the nitrate and acidity (including Al^{3+}) indicators. First of all because nitrate leaching from forests seems to be the most pressing European wide issue. Not so much due to elevated nitrate leaching from forest but mainly due to the pressures on water resources from agricultural land uses. Secondly, because most forest management operations will have an effect on N cycling, since it is the most important plant nutrient. Thirdly, because nitrate leaching is an acidifying process in it self and thus is important for the response of the acidity and aluminium indicators (Dise et al., 2001). Strong mobilisation of Al^{3+} related to nitrification and nitrate leaching has been shown in acid soils (e.g. Mulder et al., 1988; Wilpert et al., 2000).

The impact of forest management on the other indicators (DOC, heavy metal and xenobiotics) will be of more local importance and will only be mentioned under the management operations where they are most relevant.

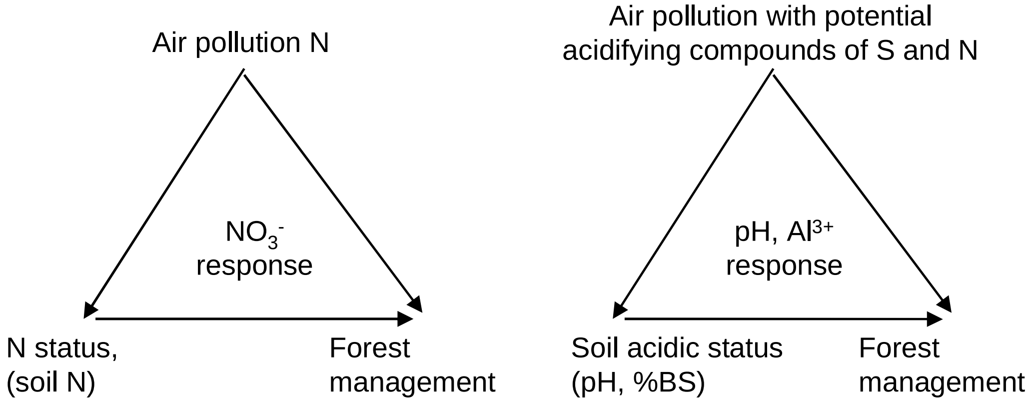

Figure 6.1. The water quality response (nitrate and acidity) to forest management operations is difficult to isolate from the impacts from the response due to air pollution. The status of the soil (which can be modified by air pollution) is also important in determining the response to management.

Elevated deposition of N and S compounds from air pollution is a widespread phenomenon in Europe, which both directly and indirectly (through changes in the soil) may affect water quality. At the same time the effects of management on water quality will be dependent on the status of the soil (with respect to N and acidity) prior to performing the management activity. Thus the impact of a management operation on water quality arise from a triangle of factors (Figure 6.1), where the impact of management often can not be fully separated from the influence from air pollution and its interaction with soil conditions. The long-term impact of air pollution on soils (the left arrows in the Fig. 1 triangles) and its possible interaction with management is an issue which will be discussed and analysed thoroughly in chapter 3 on Soil Quality.

The impact of air pollution on nitrate leaching in Europe is summarised in the following as a background for the evaluation of that impact caused by management activities. There is a threshold at 10 kg N ha^{-1} yr^{-1} in throughfall input below which almost no nitrate is leached (Gundersen et al., 2006). N deposition explains approximately half of the variability in N leaching. Part of the remaining variability could be explained as an effect of ecosystem ‘N status’, that may be described by interrelated variables like foliar N content, litterfall N flux, forest floor C:N ratio and mineralisation rate. For coniferous forests, needle N content above 1.4%, and/or forest floor C:N ratio lower than 25 were thresholds for elevated nitrate leaching. There is some evidence that the threshold in C:N ratio may be more generally valid since mineralisation increase with decreasing C:N and nitrification does only occur in the forest floor at C:N ratios below 24-27 (Gundersen et al., 2006).

The impact of potential management operations may in addition to air pollution and soil status be influence by other conditions such as climate (boreal, temperate, Mediterranean) and age phase (young, mature, old growth). Thus to analyse the effects systematically we need to consider a matrix of forest stages and conditions and the range of potential management operations (Table 1.2), but for this chapter we aggregate the information and only include the most important combinations of conditions and management operations.

6.2.1. Nitrogen, acidity and Water Quality

The mobility of N in soils, i.e. the transport of N in soil water, largely depends on the form of N (NH_4^+, NO_3^- or dissolved organic N (DON)). Ammonium is absorbed on the soil cation exchange complex and is thus quite immobile in the soil profile. Consequently, NH_4^+ concentrations are generally low in seepage water and very low in streams (< 10 µg NH_4-N L^{-1}). Ammonium usually contributes less than 5% to the total dissolved N concentration in soil water except in extremely NH_4^+ loaded soils (Dise et al., 1998a). Concentrations of DON are below 0.6 mgN L^{-1} and often even below 0.1 mgN L^{-1} in both seepage water from well-aerated soils (Andersen and Gundersen, 2000; Michalzik et al., 2001) and streams (Campbell et al., 2000; Lovett et al., 2000; Perakis and Hedin, 2002). In pristine forests streams, DON constitutes the dominant N leaching loss in the order of 1-3 kg N $ha^{-1}y^{-1}$, since nitrate concentrations were very low in those streams (Campbell et al., 2000; Perakis and Hedin, 2002).

Nitrate is the constituent in seepage and stream water that responds to flux changes (e.g., increased forest N-input, decreased forest N-uptake or increased N-mineralisation). Nitrate is highly mobile in soils and production in excess of plant and microbial uptake requirements will be transported through the soil profile. Gundersen et al. (2006) compiled survey data from the literature on nitrate leaching from mature undisturbed forests. Observed nitrate concentrations span a wide range (0-50 mgN L^{-1}) in soil water and a lower and much narrower range (0-3 mgN L^{-1}) in small streams. The frequency distributions were highly skewed towards the lower end of these ranges indicating that forest waters mostly have low nitrate concentrations. Based on the compiled surveys, Gundersen et al. (2006) suggests 0.5 mg N L^{-1} for streams/catchments and 1.0 mg N L^{-1} for seepage (annual mean concentration) as tentative limit values above which nitrate leaching is considered as elevated.

The potential effects of dissolved N in forest waters are related to i) the toxicity of inorganic N, ii) eutrophication and iii) acidification:

Toxicity of inorganic N

When ground and surface water is used for drinking water nitrate contamination is of special concern. High nitrate concentrations have been linked with methemoglobinemia (blue-baby) syndrome in human infants and is suspected to increase the frequency of some types of stomach cancer. The drinking water standard for nitrate adopted by The World Health Organisation is 11.3 mgN L^{-1} (50 mg NO3 L^{-1}) (WHO, 1998). This standard is used in the European Community (EEC, 1998), whereas USA and Canada use 10.0 mgN L^{-1} (US-EPA, 1986). Ammonium is of concern in aquatic systems. It is not toxic in itself, but it is in equilibrium with a low concentration of very toxic ammonia in water. The ammonia-ammonium equilibrium is pH dependent and shift strongly in favor of ammonium at pH<6.5. For chronic exposure at pH 6.5 to 7.0 ammonium concentrations should not exceed 2.0-2.3 mgN L^{-1}. For acute exposure the limit is 10 times higher (Binkley et al., 1999).

Eutrophication

Increased N levels in forest waters may impact productivity and species composition in nutrient-limited surface waters. Locally coastal waters may be affected if N export increases from forested catchments (Fleischer and Stibe, 1989; Hessen et al., 1997; Hinga, 1991). There is no general ecological standard for N concentration in stream/surface water, since the effects of increased N concentrations may be very dependent on P availability, but generally N inputs control the productivity in estuaries and coastal waters, whereas P is the most limiting nutrient in lakes (Vitousek et al., 1997)

Acidification

Long-term deposition of N and S has decreased the capacity in soils and fresh waters to buffer acids in moderate to high deposition regions. Both proton (H+) producing- and proton-consuming processes with N occur in soils, but a net acidification only occurs when nitrate is leached from the system (Gundersen and Rasmussen, 1990). Each 14 kg N ha^{-1} yr^{-1} of nitrate leached is equivalent to production of one kmol H^{+} ha^{-1} yr^{-1}. Depending on the acid status of the soil, base cations and/or Al will be leached with the nitrate. As the deposition of S-compounds has decreased, an increasing fraction of the acidity in acid sensitive surface waters is related to nitrate (Stoddard et al., 1999). Nitrate and aluminium are usually positively correlated in acid soil and surface waters (Dise et al., 2001). Thus NO3 has become the major anion that cause alkalinity changes and fluctuations in Al^{3+} concentrations. The sensitivity to acidification and the risk for damage to fish in surface waters is given by alkalinity and the concentration of ionic aluminium (Havas and Rosseland, 1995). Acidity and/or Al^{3+} will only reach problematic levels on soils that are acidified (pH<4.5) (an exception is snowmelt acidification pulses but they are not related to management) and only if the management activity lead to soil acidifying processes.

6.3. Methods and approaches

As a starting point for the analysis in this chapter we used the recent synthesis by Gundersen et al. (2006). They review the world-wide literature on nitrate leaching from forest including the impact of some important management operations. However, the emphasis was on the causes for nitrate leaching separating three types of N cycle disruptions leading to nitrate leaching: elevated N inputs, reduce N uptake in trees and enhanced mineralisation of soil N.

Here we build on to the data compiled by Gundersen et al. (2006) but with a main focus on European data and including recently published studies. Thus for each of the management operations listed in Table 1.2 we have searched the literature for new data on nitrate leaching and for data on other impacts on water quality.

Ideally data compilations for each management operations should amount to datasets suitable for a proper meta-analysis. However in reality the amount of studies, their replications and inhomogeneity make this type of statistical approach difficult to pursue. Only for liming and clear-cut operations there may be a sufficient amount of studies available, but for many of the desired variable for such an analysis the site and data owners would need to be contacted. Thus in the following sections we present the tables compiled and base our conclusions on general observations of the material as in a more traditional review.

In the context of water quality, the concentration of N (mg N L^{-1}) is the relevant unit of measurements, whereas for understanding the forest N budgets the flux of N leaching from forest ecosystems (kg N $ha^{-1}y^{-1}$) is the relevant unit. Provided seepage or stream water fluxes are reported as well, conversion between the two units is simple. Where relevant and possible we will use both units.

6.4. Impacts of forest management operations

6.4.1. Tree species choice

Tree species choice is and has been an important management option in plantation forestry. Various tree species influence the soil biogeochemistry and create ecologically distinct spatial environments. Plant-mediated characteristics such as litter quality and root structure contribute to the chemical and physical characteristics of the soil; e.g. coniferous species generally have higher forest floor C:N than deciduous species (Aber et al., 2003). Increased N deposition adds a new dimension to the interplay between plant species and soil biogeochemistry. Nitrogen deposition on coniferous forest is approximately two-fold higher than on deciduous forests caused by the more efficient filtering effect of conifers due to their evergreen foliage and higher leaf area (e.g. Kristensen et al., 2004). This is shown by an enrichment of throughfall for N compounds in paired comparisons of coniferous and deciduous stands (Brown and Iles, 1991; van Ek and Draaijers, 1994; Rothe et al., 2002; Wilpert et al., 2000) (Table 6.1). In central to northern Europe, deciduous species have been replaced by conifers (mainly Norway spruce) on a considerable part of the forest area indirectly leading to increased N deposition to the European forest area in the current air pollution climate.

Table 6.1. Differences in seepage water nitrate concentrations (or leaching flux) between paired mature stands of conifer (C) and deciduous (D) species at the same site (n=1 unless otherwise noted).

Site		**Species**	**Nitrate in drainage**		**Deposition**	**Years**	**Reference**
			mg NO_3-N L^{-1}	*kg N $ha^{-1}y^{-1}$*	*kg N $ha^{-1}y^{-1}$*		
Three forest areas in Bavaria, Germany	C	Norway spruce (n=22)	0-13[a]		20-30	1998-2000	Rothe and Mellert 2004
		Silver Fir (n=3)	0-2				
	D	Beech (n=11)	0-1		n.d.		
		Other (n=5)	0-2				
Höglwald, Germany	C	Norway spruce	7-14	21	30	1985-89, 94-97	Kreutzer and Weiss 1998
	D	Beech	0-1	1	15		
Schongau, Germany	C	Norway spruce	4-8			1994-97	Rothe 1997, 1998
	D	Beech	1-3				
Solling, Germany	C	Norway spruce	2.9	12.5	31	73/71-85 volume weighted mean	Matzner 1988
	D	Beech	0.3	1.5	25		
Convent Forest, Germany	C	Norway spruce (n=2)		15	26		Wilpert et al. 2000
	D	Beech and Beech/S.fir		4	17		
Aubure, France	C	Spruce		15	19	1993-95	Dambrine et al. 2000
	D	Beech		1.5	9		
Stubbaröd, Sweden	C	Norway spruce	0.6	2.3	32	1991-92; 50-75 m from edge	Påhlsson and Bergkvist 1995
	D	Beech	0.6	3.1	24		
Forellenbach, Germany	C	Norway spruce	1-4		16	1992-1995	Kleemola and Forsius 2000
	D	Beech	0.2		12-14		
Mont-Lozere, France	C	Norway spruce		1.8	10.9	1981-1986	Durand et al. 1992
	D	Beech		0.2	7.9	1981-1986	
Hoogmoerhei	C	Corsican pine	16	53	96	1998-2000	De Schrijver et

de, Belgium	D	Silver birch	6.4	21	27		al. 2004
Nordrhein-Westfalen, Germany	C	Pine		~3	11.2	1985-1987	Van Grinsven et al. 1991
	D	Oak/Beech		~0	5.6		
Lindet, Denmark	C	Norway spruce	2	1.3	24	1994-97	Hansen (ed.) 2003
	C	Sitka spruce	8	9.1	35		
	C	Douglas fir	0.5	2.1	27		
	D	Beech	<0.1	0.1	15		
	D	Oak	0.1	0.3	11		
Nødebo,	C	Norway spruce	0.6	0.5		2000	Christensen et al. 2006
Denmark	D	Beech	11.3	39			
Vallø, Denmark	C	Norway spruce	3.1	1.9	23 [b]	2004-2005	Christensen 2006
	D	Beech	9.8	17	11		
	D	Oak	0.1	0.3	9		
	D	Lime	0.7	2.1	12		
	D	Maple	<0.1	0.1	9		
Mattrup, Denmark (afforestation on former agricultural field)	C	Norway spruce	20		37 [b]	2004-2005	Vesterdal, personal communication
	D	Beech	13		10		
	D	Oak	14		9		
	D	Lime	9		19		
	D	Maple	0.8		12		
	D	Ash	0.7		9		

[a] Highest concentration were found in the forest with the highest deposition.

[b] One year of throughfall measurements only. Significant canopy uptake may be the reason for the lower N input in decideous than coniferous throughfall.

The observed differences in N deposition between conifers and deciduous are reflected in higher output of nitrate from the soil under conifers (Table 6.1) when compared to deciduous species at the same site. Rothe et al. (2002) compared three pairs of similar age beech-spruce stands in Germany and found that the throughfall N input was more than doubled in spruce compared to in beech. The cycling processes in the forest floor resulted in similar amounts of ammonium, DOC and DON for both species, but below 20 cm soil depth N in soil solution was almost entirely in the form of nitrate under spruce due to strong nitrification. Similar patterns were found at 12 other European spruce-beech pairs located directly adjacent to each other (Figure 6.2). Rothe et al. (2002) concluded that as the deposition level increases the species-related difference in nitrate leaching also increases. Both the data used in Figure 6.2 and listed in Table 6.1 show a number of beech stands with low levels of seepage nitrate and full N retention over the whole input range. A regional comparison of several spruce and beech stands in S. Germany again confirmed full N retention under beech, but elevated nitrate concentrations under spruce (Rothe and Mellert, 2004). But recent data from nutrient rich site in Denmark (Christensen, 2006; Christensen et al., 2006; Vesterdal, personal comm.) show relative high nitrate leaching from beech (Table 6.1) even higher than from spruce. This could be due to their relative young age (25-30 yrs). The species difference observed from adjacent pairs also seems contradictory to observations from monitoring sites, where deciduous have higher nitrate concentrations than conifers at similar input (Kristensen et al., 2004). However, the deciduous monitoring sites were growing on more nutrient rich soils (higher N status) than the conifers (Kristensen et al., 2004).

Species differences other than filtering efficiency, such as hydrological differences or differences in nutrient requirements, may also influence N cycling response among species when grown on the same soil. Thus species-mediated differences in N cycling and leaching are not just a matter of differences between deciduous and coniferous tree species. Differences are also apparent between various species of conifers or of deciduous (Table 6.1). The enrichment of throughfall was higher in Douglas fir than in Scots pine (Ek and Draaijers, 1994) and higher in Sitka spruce than in Norway spruce and Douglas fir at Lindet, DK (Hansen, 2003), which then again was reflected in nitrate leaching (Table 6.1). Preliminary data from tree species trial with broadleaf species (Vallø and Mattrup sites in Table 6.1) reveal important differences in N retention among broadleafs (Christensen, 2006; Vesterdal, personal comm.). Maple and ash could retain all N inputs on a former agricultural field at high deposition (Mattrup), but on an old forest soil and intermediate deposition (Vallø) oak and lime also had high N retention and no nitrate leaching (Table 6.1).

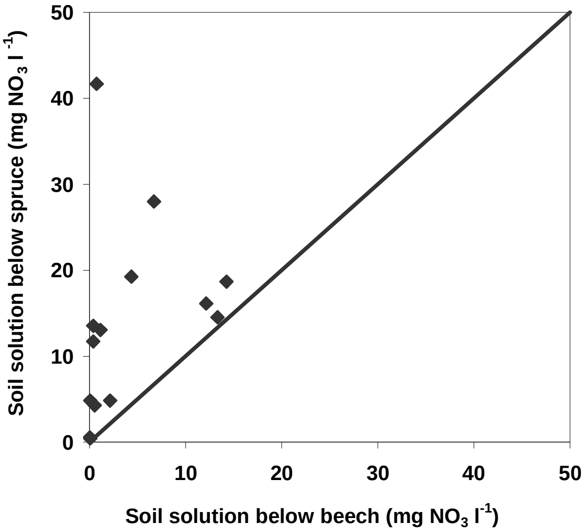

Figure 6.2. Nitrate concentration in seepage water from Norway spruce vs beech in adjacent stands on similar soils across Europe. Modified from Rothe et al. (2002).

The differences between species imply that management practices favouring certain species may delay or accelerate N-leaching and the potentially negative changes associated with this process. In Central and Northwest Europe, where deciduous species have been replaced by planting of conifers over the last century, trends to return to the original deciduous species may (by means of the lower N input in deciduous) alleviate the relatively widespread elevated nitrate levels in seepage water. However, we have no documentation that this will occur. The N stored in the soil over the rotations with high N deposition under conifers may limit N retention when the original deciduous species are reintroduced. Research on the effect of conversion from confers to broadleafs are need as well as studies comparing nutrient cycling characteristics of the range of broadleaf species that may potentially be used in this conversion.

In relation to nitrate leaching, N abundance of N-fixing trees such as alder (*Alnus* ssp) and black locust (*Robinia pseudoacacia*) in forest stands or catchments make a special case since they significantly increase N input to the ecosystem.Black and red alder have been used as a silvicultural tool to improve soil fertility and the N status of the system (Tarrant and Trappe, 1971). Currently alder and robinia are estimated to make up approximately 1% of the European forest area (Köble and Seufert, 2001). This may be the reason why only one European study of nitrate leaching from an N-fixing stand (Robertson et al., 2000) was identified in the literature.

Annual symbiotic N-fixation by alder stands ranges between 50 to 200 kg N ha^{-1} (Binkley et al., 1992; Bormann and DeBell, 1981). Accordingly, high levels of nitrate in soil water (Binkley et al., 1992; Van Miegroet et al., 1992b; Robertson et al., 2000) and in streams (Brown et al., 1973; Compton et al., 2003; Willard et al., 2005) have been reported from forests with N-fixing species. Hurd and Raynal (2004) also found higher nitrate concentration in soil solution, groundwater and stream water in alder dominated wetlands compared to non-alder wetlands. The mean annual concentration of nitrate in soil water under alder from stands in the US and UK ranged from approximately 3 to 6 mg N L^{-1} (Van Miegroet et al., 1990; 1992b; Robertson et al., 2000), which are only comparable to soil water concentrations found in high deposition regions in Europe (Gundersen et al., 2006). The production and leaching of nitrate from the topsoil resulted in decreased pH in alder soils (Van Miegroet and Cole, 1985). Furthermore, the organic layer C:N ratio was only 14 in alder compared to 21 in an adjacent Douglas fir (Van Miegroet et al., 1990), which indicates that the retention capacity is low under N-fixing species.

An investigation of 26 small catchment streams in in the Oregon Coast Range (a low N deposition region in west USA) showed that the per cent cover of red alder in these catchments explained 72% of the variability in stream nitrate concentrations (Compton et al., 2003). For each 10% increase in alder cover nitrate concentration increased 0.2 mg N L^{-1} and leaching loss increased by about 3 kg N ha^{-1} yr^{-1}. Nitrate concentrations were more strongly related to alder cover in the whole catchment than in the riparian area alone, which indicates that leaching from upland alders were important (Compton et al., 2003). Cutting of alder stands and thereby drastically reducing the fixation N input decreased nitrate levels post cut (Homann et al., 1994; Mann et al., 1988; Robertson et al., 2000). Thus elevated N leaching is readily reversible when fixation input is reduced.

6.5. Site preparations

6.5.1. Physical manipulations

6.5.1.1. Ditching

Improved drainage by ditching in forest ecosystems on wetlands is a common and prerequisite practice for the utilisation of areas for forestry. The drainage of wetlands and subsequent oxidation of organic compounds can mobilise large amounts of stored organic N. In Denmark, for example, Callesen et al. (1999) found concentrations up to 120 mg NO_3-N L^{-1} following ditching of a nutrient rich bog characterised by a low C:N-ratio. The effect of draining wetlands upon forest growth is well-documented, whereas it's effects on other ecological processes such as mineralisation, nitrification and nitrate leaching is less well studied. Drainage may result in a direct transport of leached nitrate to the streams, where denitrification may otherwise have been a significant sink for nitrate retained in the wetlands.

Silvicultural practices (thinning, clear-cutting and site preparation) in drained wetlands showed increased levels of suspended sediment and nutrients compared to undisturbed controls (Bormann and Likens, 1979; Briggs et al., 2000; Shepard, 1994). In an extensive study of 20 boreal catchments in Sweden and Finland, Lepistö et al. (1995) found that drainage explained 29% and 67% of the export of ammonium and organic N, respectively, to forest streams. The increase in organic N export up to 5 kg N $ha^{-1}y^{-1}$ indicated increased erosion due to the drainage activities (Lepistö et al., 1995). They did not find any correlation between nitrate and drainage, which may be due to a lack of nitrification and/or high absorbing and retention capacity in boreal peaty soils. Even in the first years after ditching, Åstrøm et al. (2002) found practically no effect of ditching on N export in a boreal catchment.

6.5.2. Chemical treatments

6.5.2.1. Fertilisation

Forest fertilisation with N is a common practice in intensively managed plantations and occurs on about a half million of hectares yearly in the industrialised world (Binkley et al., 1999). Fertilisation is aimed at N-limited forest where a growth response is expected. Thus fertilisation with N have ceased in the regions of Europe with elevated N deposition. Binkley et al. (1999) compiled world-wide data on N concentrations in soil water and in streams after forest fertilisation for a comprehensive review on water quality impacts. They found peak nitrate concentrations in seepage water up to >10 mg L^{-1} after fertilisation in 50% of the studies. The concentration stayed above 10 mg NO_3-N L^{-1} for at least a year in 34% of the studies (Figure 6.3). Some of the highest concentrations in Figure 6.3 were reported from treatments with repeated fertilisations and high doses (Binkley et al., 1999). The N concentration in seepage water, however, was higher than the concentration in streams due to denitrification and other N processes in the riparian zone, in-stream removal and mixing with other water. The highest reported annual average nitrate concentration in stream water was 4 mg NO_3-N L^{-1} and Binkley et al. (1999) thus concluded that forest fertilisation commonly leads to only moderate increases in stream water N concentrations. The greatest increases in nitrate leaching come from: i) direct application to streams; ii) use of fertilisers containing nitrate; and iii) the application of high rates (or repeated doses).

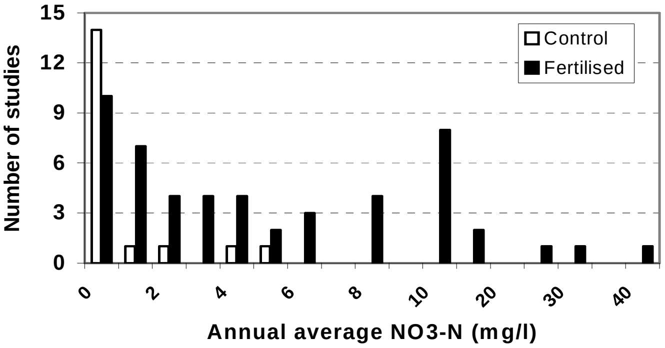

Figure 6.3. Nitrate concentration in seepage from control and fertilised plots 1-5 years following fertiliser application. The x-axis is divided in 5 mgN L^{-1} intervals for levels above 10 mgN L^{-1}. Modified from Binkley et al. (1999).

The highest N concentration in stream water (4 mg NO_3-N L^{-1}) was reported from West Virginia after fertilisation in an almost N-saturated forest (Adams et al. 1997). The same high leaching loss was observed in a number of N-saturated forests after fertilisation in Japan (Mitchell et al., 1997; Ohrui and Mitchell, 1998) and in northern Florida, USA fertilisation of a pine plantation on former agricultural land (soil C:N ratio 10) practically all the added N was leached (Lee and Jose, 2005). In contrast, decreased nitrate leaching was achieved by high dose fertilisation with P and K at N-saturated conditions (Stevens et al., 1993) due an increase in N uptake by the trees and possibly a stimulation of growth, suggesting that P and/or K were limiting nutrients. This illustrate that forest fertilisation with other elements than N could be considered in high N deposition regions and that fertilisers with an N-P-K stoichiometry optimal for forest growth may be relevant in general (e.g. Tamm, 1991).

The conclusion from Binkley et al. (1999) as well as from other studies (Ring, 1995; Shepard, 1994) is that operational forest fertilisation poses little or no risk to water quality. Operational fertilisation is aimed at increasing forest growth and as such done in N-limited forest with high N retention capacity. Fertilisation in regions with elevated N inputs from the atmosphere (>5-10 kgN $ha^{-1}yr^{-1}$) probably have limited effect on tree growth and may accelerate the rate towards N-saturation.

6.5.2.2. Liming and wood ash recycling

During the last decades nutritional imbalances and accelerating forest soil acidification have been reported, especially in northern and central European coniferous forests (Ingerslev, 1997). This has been related to acid rain and enhanced atmospheric deposition of N compounds. Application of lime (i.e. calcite or dolomite) has been suggested as a tool to counteract the acidification of forest soils and the loss of base cations (Huettl and Zoettl, 1993). Recently application of wood ash has received attention as an alternative to lime and as a means to recycle nutrients removed from the forest ecosystem in logs.

In general, there is an extensive literature on long-term effects of liming on both soil biology and chemistry. The effects include increase in soil pH, increased base saturation (Derome et al., 1986; Johnson et al., 1995) and reduction in Al release (Derome et al., 1986; Keersmaeker et al., 2000) as intended, but also on alterations of the C and N cycling (Arnold et al., 1994; Matzner and Meiwes, 1990; Simmons et al., 1996). One recent concern about liming is the increased levels of nitrate in the soil solution as observed in a number of studies (Table 6.3; De Boer et al., 1993; Geary and Driscoll, 1996; Marschner et al., 1992; Nilsson et al., 2001). Liming may cause decreased forest floor C:N ratio (Kreutzer, 1995) and increased forest floor pH, which stimulates net nitrification (Persson et al., 2000b). Kreutzer (1995) found that the increase in nitrate below the rooting zone was due to nitrification in the mineral soil because DON accounted for the major increase in N-flux from the forest floor after liming. The shift from ammonium to nitrate was followed by a decrease in mineral N retention in the mineral soil from 88 % in the control soil to 40 % in the limed treatment (Nilsson et al., 2001). However, in a number of north European studies on N poor soils, liming did not significantly increase leaching of N (Table 6.3; Hindar et al., 2003; Ingerslev, 1997; Lundell et al., 2001; Nohrstedt, 1992; Persson and Wirén, 1996). In a lysimeter study with and without tree roots included in the cores, Lundell et al. (2001) only observed nitrate leaching in the absents of roots and at the most N rich site (C:N ratio 24) indicating the importance of the plant sink as well as of the soil microbial immobilisation of N at the more N poor sites.

The high potential nitrification observed in the number of liming experiments might indicate that the risk and extent of other N cycle disruptions is higher in limed forests. However, liming promoted the establishment of ground vegetation (Bartsch et al., 1999; Bauhus and Bartsch, 1995) by favouring species with high nutrient demand (Keersmaeker et al., 2000). This may increase the resilience of the ecosystem to nutrient losses after disturbance.

Liming has a major impact on a large number of biological and chemical processes, which influence N mobility and plant requirements for N. It is not clear what controls the nitrate leaching response to liming. The dose of lime (or wood ash) seems to increase the response (Person and Wiren, 1996; Shäffer et al., 2002; Table 6.3), and doses larger than 3 t ha^{-1} all caused a nitrate response. The greatest absolute response in nitrate concentrations appeared at those sites in Germany and the Netherlands which already leached some nitrate and where N deposition was also relatively high (Table 6.3). Further, the study by de Boer et al. (1993) indicates a larger response in old stands, where the N uptake by trees probably is low.

6.6. Stand management and harvesting

Disturbances of the forest cover by management (clear-cut, thinning etc.) may have variable intensities from an almost complete uncoupling of the tree uptake to a minor change in the uptake rate. The intensity of the disturbance as well as the capacity of the ecosystem to repair itself by regrowth (ecosystem resilience) are important for the duration and the extent of the N-cycle disruption. In this section we focus on the controls on nitrate leaching when the plant cover is disturbed by management; however, we also included studies from windthrown sites where trees are removed by salvage cuttings.

6.6.1. Clear-cutting

The major single man-made disturbance applied to forest ecosystems is clear-cutting. Plant uptake is disrupted, and decomposition rate will change depending on local conditions (Yin et al., 1989). In general net mineralisation and nitrification increase as a result of decreased immobilisation in litter by microbes (Prescott, 1997; Smolander et al., 1998). However, if large amounts of logging residues with high C:N ratio is left on the site microbial immobilisation may increase and delay the mineralisation response. Furthermore, the outflow of seepage and runoff water is increased due to lower evapotranspiration (Knight et al., 1991; Qualls et al., 2000; Swank et al., 1988). A comparative study of nitrate losses from trenched plots in 19 forest sites in the USA suggests that the most important processes limiting the nitrate leaching response are 1) Processes preventing or delaying ammonium accumulation (e.g. ammonium immobilisation in soil organic matter and logging residues with high C:N ratio, ammonium fixation or ammonium uptake by regrowing vegetation), 2) Processes preventing or delaying nitrate accumulation (e.g. biological denitrification or uptake by regrowing vegetation), and 3) Processes preventing or delaying nitrate mobility (e.g. lack of water or chemical denitrification) (Vitousek and Melillo, 1979; Vitousek et al., 1979).

The effect of clear-cutting on leaching of nutrients has been followed in numerous studies since the late 1960s, where the classic experiments at the Hubbard Brook Experimental Forest (HBEF) in North Eastern USA (Likens et al., 1970) illustrated the dramatic increase in leaching of nutrients from a catchment scale clear-cut. In this experiment the vegetation recovery was delayed by herbicide application leading to massive nutrient losses (e.g. leaching of 150 kgN ha^{-1}yr^{-1}). Later, other studies of clear-cuts more close to commercial practice were performed at HBEF (Figure 6.4; Bormann and Likens, 1979) and numerous studies have been preformed in Europe (Table 6.2). In general, the nitrate concentration in soil and stream waters increase with peak nitrate concentrations within 2-3 years after clear-cut (Figure 6.4; Table 6.2). The nitrate concentration often returns to pre-cutting levels within relatively short time, normally 3-5 years, especially if clear-cut is performed without any other disturbances (e.g. site preparation and herbicide application) (Vitousek et al., 1992).

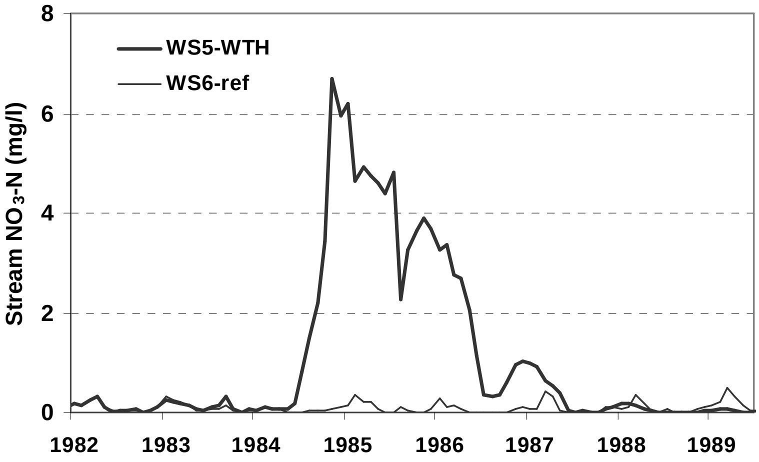

Figure 6.4. Typical temporal response of nitrate concentration in stream water after clear-cut. Data from Hubbard Brook Experimental Forest in New Hampshire. All of the trees were removed from watershed 6 (thick curve) in the winter of 1983-84. Watershed 5 (thin curve) remained undisturbed throughout the period. The data were provided by Gene E. Likens through funding by the National Science Foundation and The A.W. Mellon Foundation.

Among the studies in Table 6.2 the highest responses in nitrate concentration in stream or seepage water (the difference in concentration between cut and reference stands) were observed in Central Europe (5 mgN L^{-1} as a mean over the region) followed by Northwest Europe (mainly UK) and Northeast (Sweden, Finland). This pattern of regional response follows the general trend in deposition N among the regions with Central Europe receiving the highest deposition.

To further investigate if the nitrate response increase with N deposition or with N status, we compiled information on N deposition, forest floor C:N ratio and pre-cut litterfall N flux from sites in Table 6.2. Such data were only sparsely available in the literature we compiled. As a surrogate we used the nitrate concentration in seepage water from the intact reference stand as a proxy for N status (Figure 6.5) expecting that the most N-saturated sites would show the largest response. There was, however, no clear trend in the nitrate response (expressed as the difference between pre- and post-harvest condition) with increasing pre-cut nitrate concentration (Figure 6.5). Yet, differences in evaporation amount among sites with increased post-cut water flux could dilute post-cut concentration differently and a strong reduction in dry deposition post-cut may also influence the observed relationship. Nevertheless, a number of other factors, e.g. harvest intensity, site preparation, plant recovery, site quality and erosion appear to significantly influence the magnitude and duration of leaching losses after clear-cut. Alder forests in UK were excluded from Figure 6.5 since pre-cut levels of nitrate were high (Table 6.2) due to high N input from N fixation. In these forests, cutting decreased nitrate concentration levels (Table 6.2; Homann et al., 1994; Mann et al., 1988; Robertson et al., 2000) probably due to the reduction in fixation N input and decreased evapotranspiration.

6.6.1.1. Harvest intensity

The degree of biomass removal in connection with clear-cutting may influence the magnitude of export of N from the system. Whole tree harvest (WTH) compared to conventional or stem-only harvest (CH) removes up to 2-4 times more N from the forest due to lower C:N ratios in foliage and branches (Fahey et al., 1991; Johnson et al., 1982; Mroz et al., 1985; Møller, 2000). WTH has therefore been seen as a way to counteract the effect of N deposition and reduce the leaching of N in high deposition areas in Europe (Lundborg, 1997). In contrary, other researchers have pointed out that the substantial removal of nutrients by WTH was incompatible with sustainable forestry.

There are studies showing both increased and decreased nitrate leaching after WTH compared to CH (Hendrickson et al., 1989; Mann et al., 1988; Stevens et al., 1995). Across the studies, we found no general difference in nitrate leaching following CH and WTH but site specific differences. The combined effect of biomass harvest regime and site specific conditions may influence several processes, e.g. regrowth and nutrient uptake of the vegetation and N mineralisation and immobilisation of soil organic matter and logging residue, which exert important controls on N-retention and nitrate leaching after clear-cut (Vitousek et al., 1979; Vitousek and Melillo, 1979). This indicates that responses in nitrate leaching to CH and WTH are a function of pre-existing site conditions.

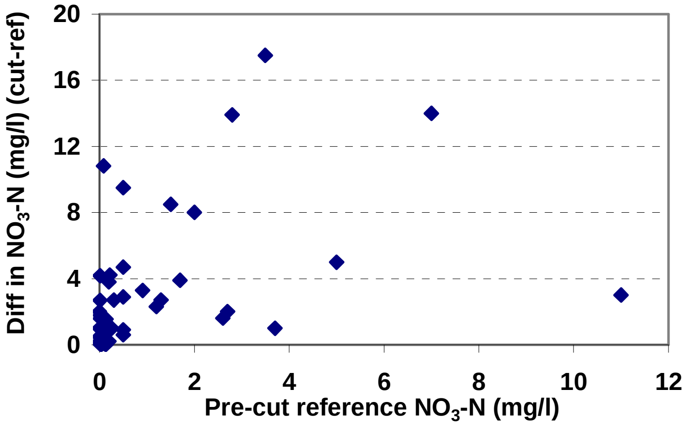

Figure 6.5. The increase in mean annual seepage water nitrate (mg NO_3*-N* L^{-1}*) after clear-cut (nitrate response), i.e. the difference between cut stands and intact reference stands, plotted as a function of pre-cutting nitrate concentration. Data are calculated from Table 6.2.*

Decomposing logging residues may be an important sink for N due to the high C:N ratio in the material. Decomposers can retranslocate large amounts of N into the dead wood to a point where the C:N ratio is low enough to allow nitrogen release (Vitousek and Melillo, 1979). Further, woody debris may prevent erosion in well-drained areas. Briggs et al. (2000) observed increased nitrate export after WTH in well-drained areas compared to poorly drained. A number of studies found a reduction of nitrate leaching with the amount of woody debris left at the site, especially if chipped or buried (Carlyle et al., 1998). Also Stevens et al. (1995) found that woody debris after CH was a net sink of N for three years following clear-cutting whereas it became a source of N in the fifth year. However, Olsson et al. (1996) found long-term increase in forest floor and upper mineral soil C:N ratio after WTH, which may have a positive effect on the long-term N-retention, at least at sites exposed to high N-deposition. Nitrate leaching following CH and WTH were compared over the second year after clear-cut in a Sitka spruce forest in UK using lysimeters (Emmett et al., 1991a,b). Leaching was reduced 90% by WTH compared to CH. The effect was partly attributed to a better establishment of grass with logging debris removed by WTH. In contrast, at a more N limited Swedish site, Olsson and Staff (1995) observed lower ground cover after 8 and 16 years following clear-cutting in WTH plots compared to CH.

CH and WTH are often associated with other disturbances, which may influence the magnitude of nitrate losses. Often the slash is piled to make replanting easier on CH sites, but N leaching can be substantial under slash piles (Rosen, 1988; Staaf and Olsson, 1994). WHT include mechanical disturbance with delayed re-growth of ground vegetation but favouring replanting, damage to forest tree seedlings, compaction of the soil, and less woody debris to support biodiversity and retain water and nutrients (Mou et al., 1993). Furthermore, substantial amounts of other nutrients are removed by WTH (Johnson and Todd, 1998; Johnson et al., 1982; Yanai, 1998; Møller, 2000). Nitrogen will be replaced at high N deposition sites but weathering may not be able to supply a new stand with sufficient base cations and phosphorus at many sites. Consequently WTH may result in soil acidification and reduced long-term growth potentials. In some areas where WTH is practised, the logging debris is used for bio-energy. The wood ash may be returned to the logged areas to counteract loss of base cations and phosphorus.

In general, the impact of WTH or CH on seepage nitrate is depending on pre-existing site conditions. The difference in seepage nitrate between stands with different degree of biomass utilisation is minor compared to the large effect of clear-cutting and in flux terms the difference in nutrients removed by harvest (Mann et al., 1988; Stevens and Hornung, 1988; Vitousek and Matson, 1985).

6.6.1.2. Plant cover establishment and site preparation

One of the important controls on the magnitude and duration of elevated nitrate concentration after harvest is the recovery of the plant N sink illustrated by many studies (Bormann et al., 1977; Boring et al., 1988; Fahey et al., 1991; Klimo and Kulhavy, 1994; Rothe and Mellert, 2004; Weis et al., 2001). Emmett et al. (1991a,b) found a 80-90% reduction in nitrate leaching when more than 50% cover of grass was established. Accordingly, Mellert et al. (1998) found high negative correlation between vegetation cover and nitrate in soil solution (r^2=0.7) at windthrown and cleared sites in Germany (Figure 6.6). From other areas in Germany similar negative relationships between ground vegetation cover and nitrate in soil solution was observed by (Huber, 2005; Rothe and Mellert, 2004). This is explained by both a higher total plant biomass (weed and trees) in weedy areas and a higher N-accumulation in weeds compared to the tree culture (Smethurst and Nambiar, 1989).

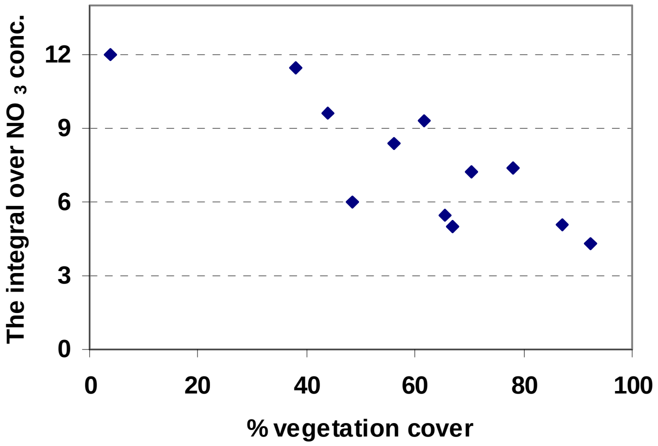

Figure 6.6. The relationship between the integral of seepage nitrate concentration (g NO_3 L^{-1}) over time and total vegetation cover (%) in plots cleared after windthrow in Bavaria, Germany. The y-axis is a combined measure of the extent and duration of the nitrate response to clearing. Redrawn from Mellert et al. (1998).

Although weeds may improve N retention after harvest, weeds exert a strong competition in regeneration or establishment of a new plantation and weed control in this period greatly improves tree growth in a number of species (Chang and Preston, 2000; Munson and Timmer, 1995; Sutton, 1995). Weed control includes a number of methods, e.g. herbicides, site preparation involving mechanical removal, mulching, and inter-specific plant competition, which influence the leaching of nitrate after harvest in highly different ways.

Application of herbicides is the most common practice of weed control and often the most cost-effective method as well. A number of studies have found increased soil temperature and moisture along with increased nitrate concentrations in the soil after herbicide treatment (Lambert et al., 1994; Munson et al., 1993; Ogner, 1987 a, b; Vitousek and Matson, 1985, Vitousek et al., 1992), thus mineralisation and nitrification was probably stimulated. Callesen et al. (1999) include a case study where repeated herbicide treatments of grass lead to peaks in soil water nitrate up to 75 mgN L^{-1}. In a study in moderate well-drained soils in Central Maine, Briggs et al. (2000) observed that application of herbicide the fourth year following harvest resulted in an increase in nitrate in soil solution similar in duration and levels to that after clear-cutting. Herbicide was applied to inhibit growth of competitive hardwoods interfering with conifer development.

Site preparation, such as disking, ripping etc., performed to improve soil conditions or as weed control may have large effects on both the magnitude and the duration of increased nitrate in seepage water. Intensive site preparation (e.g. disking) increased N mineralisation, nitrification and nitrate losses (Vitousek and Matson, 1985). Further, soil preparation may increase the risk for erosion and export of suspended particles to forest streams (Bormann and Likens, 1979; Briggs et al., 2000). Common practices in Northern Europe are disc trenching or mounding with mixing of forest floor and mineral soil. Such disturbances may result in increased organic matter decomposition and impose the risk of nitrate leaching (Smolander et al., 2000). A number of studies have focused on alternative weed control where either fast growing tree species or ground cover species are used as strong competitors to the weed species. Willoughby (1999) found that planting densities above 10.000 stems/ha could more or less out-compete weeds. Weed control performed as inter-specific competition keeps the vegetation cover constant high. This can be done by the use of tree species that are less affected by competing weeds, e.g. pioneer species like oak and birch.

In a long-term perspective, leaching of N after harvest will be followed by a period of rapid re-growth and high N retention (Klimo and Kulhavy, 1994; Richter et al., 2000; Rothe and Mellert, 2004). Forest re-growth in White Mountain National Forest in New Hampshire on areas subjected to logging or fire a century ago, indicates that removal of biomass N may actually increase the long-term retention of N (Goodale and Aber, 2001).

6.6.1.3. Site quality and nitrogen availability

Work by Wiklander (1983) emphasised the effect of site quality, e.g. N-status on post-harvest nitrate. He found that the highest concentration of groundwater nitrate (4 mgN L^{-1}) appeared on a high quality site while concentrations were only around 1 mgN L^{-1} on low quality sites (Table 6.2). However, the duration of elevated nitrate concentrations was limited to 2 years at the high-quality site and lasted 5-10 years after clear-cut on the lower-quality sites. This may, at least partly, be an effect of differences in the re-establishment of the vegetation cover.

The data for the observed nitrate responses (Figure 6.5) are inconclusive on the hypothesised effect of N status on the response. Additional controlled experiments leaving out effects of vegetation recovery and amounts of debris have touched upon this issue. On low productivity boreal sites in Sweden, no leaching was measured in three years after clear-cut even at sites, which prior to cutting had received high amounts of fertiliser (except in the highest accumulated dose of 1800 kg N ha^{-1}) (Ring, 2001). However in the fifth year, leaching of nitrate tended to increase in all treatments and continued to increase at all fertiliser levels at least to the 10th year after felling. The increase was positively correlated to fertiliser dose (Ring 2001) and number of years after clear-cut. Firstly, this confirms the delayed response in nitrate leaching at low fertility sites observed by Wiklander (1983). Secondly, it demonstrates a profound effect of site N-status on nitrate leaching illustrated in this case by a N status gradient created by increasing N fertiliser doses. Recently, a survey of soil nitrate concentrations at 29 clear-cut sites in south Sweden confirmed an increase in soil water inorganic N concentration from <1 to 4 mgN L^{-1} with deposition increasing from 15 to 25 kg N ha^{-1} yr^{-1} (Akselsson et al., 2004; Figure 6.7). Over this deposition range inorganic N concentration in soil water was constantly low (full retention of deposition N) at almost all sites with growing forests (Figure 6.7), but after clear-cut part of the accumulated deposition N was apparently leached. Estimates of N leaching over the first 5 years after clear-cut was <5 kg N ha^{-1} yr^{-1} at low N deposition and 25-35 kg N ha^{-1} yr^{-1} at high N deposition (Akselsson et al., 2004).

Litterfall N flux is one of the indictors of ecosystem N status (Gundersen et al., 1998b) and by re-examining a dataset from USA (Vitousek et al., 1982) a threshold for elevated risk of nitrate losses after disturbance was indicated at c. 50 kg N ha^{-1} yr^{-1} of litterfall N flux in the intact forest before disturbance (Gundersen et al., 2006). Above this level of litter N input to the soil the sinks for N in plants and microbes may be 'saturated' and in the event of plant disturbance there is no further potential for increased immobilisation in microbes.

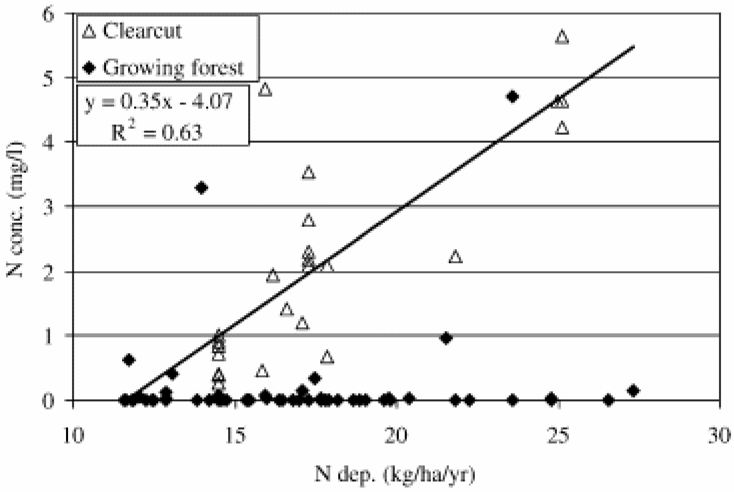

Figure 6.7. Measured inorganic N in soil water (mean for 1-6 years) as a function of modelled deposition of inorganic N for coniferous forest in southern Sweden (Akselsson et. al. 2004). Triangles are clear-cut sites (n=24) and diamonds are growing forests (n=117). The regression line is derived from data from clear-cuts. Reprinted from Akselsson et. al. 2004 with permission from Elsevier.

6.7. Other silvicutural regimes

Alternatives to clear-cut harvest include regeneration in gaps, selective cutting or thinning creating un-even aged stands. Stand age is an important factor determining N uptake rate in forest ecosystems. The retention of N in forests declines dramatically after crown closure. Nitrate leaching losses increase as the growth rate of the forest decreases (De Boer et al., 1993; Goodale et al., 2000; Emmett et al., 1993; Rothe and Mellert, 2004; Stevens et al., 1994; Vitousek and Reiners, 1975; Van Miegroet et al., 1990). Regeneration within gaps or beneath the old trees combined with selective cuttings may create forests of mixed ages with continuously higher N demand and N retention capacity on a regional scale. However, it is not generally documented whether high N demand can be sustained over the long-term by selective cuttings.

Regeneration in gaps are important in unmanaged natural forest succession and may be an alternative to clear-cutting in "close-to–nature" forestry, aiming at reduction of the negative consequences of the clear-cut forest management system (Larsen, 1995). However, several investigations in European beech forests report seepage water nitrate concentrations from gaps above the WHO threshold for drinking water quality of 11.3 mgN L^{-1} (Bauhus and Bartsch, 1995; Bartsch et al., 1999; Ritter et al., 2005; Ritter and Vesterdal, 2005). Thus the situation in gaps compared to the situation after clear-cut, although it was thought that uptake by surrounding trees might reduce nitrate leaching faster than after clear-cut. Those stands were all in high deposition areas, and Wilpert et al. (2000) concluded that under the present emission regime (Northwest Europe), even small gaps (<0.1 ha) can cause high nitrate output. Accordingly, Ritter et al. (2005) found no effect of gap sizes (20 and 30 m diameter) on the high level of nitrate leaching from gaps. In a study of gaps in high-elevation spruce-fir forests in Canada (at low N deposition) soil nitrate concentrations were also significantly elevated in the gaps up to 7 years after cutting and no effect of gap size (0.1 to 10 ha) was observed (Prescott et al., 2003). But removal of single trees did not result in any increase of soil nitrate. The effect of gaps on soil water quality has to be evaluated in relation to the management of the whole forest or catchment. The spatial scale has not been included in these studies as they are based on plot trials. The magnitude of response on whole catchment soil water quality has not been evaluated, but if gaps are only created at one to a few per cent of a forest area at the same time effects on water quality are expected to be minimal.

Prescott (1997) studied a number of alternative silvicultural practices to clear-cutting in British Columbia and found low and almost even levels of forest floor nitrification in old-growth, shelter wood and patch-cut stands compared to high levels in clear-cut stands. Accordingly, Feller et al. (2000) found significantly higher seepage nitrate in clear-cut in the same region compared to the uncut forest but a smaller or insignificant increase in nitrate concentration in areas cut 50% and 71%. Bäumler and Zech (1999) observed an immediate increase in topsoil ammonium concentration followed by a moderate increase in soil solution nitrate with a peak concentration already after 6 months after 40 % thinning in mixed mountain forest in south Germany. The concentrations were back to pre-cutting conditions after one year. At three other South Germany sites, Weis et al. (2001) found almost no increase in nitrate leaching from selective cuts in Norway spruce in the first year, but significant increases after clear-cut of adjacent stands (Table 6.2). Survey data from another forest area in south Germany confirm insignificant nitrate response to thinnings up to 60% of growing stock (Rothe and Mellert, 2004). Briggs et al. (2000) observed no increase in nitrate concentration after a 12-fold reduction in stem density after thinning in balsam fir and red spruce in Maine.

In general, removal of all or some trees has an impact on the water flow. Thinning treatment in a lodgepole pine removing 60% of the trees increased water outflow by 92% compared to 277% after clear-cutting (Knight et al., 1991). Increase in soil nitrate was detectable up to two years after thinning, but the increase was low compared to the nitrate concentration found in the clear-cut stand. In conclusion, thinning of up to 60% seems to keep the plant uptake function intact and no increase in nitrate leaching is observed.

The discrepancy between the high increase in seepage nitrate in gaps compared with shelter wood systems might be explained by several factors. First, gaps create vegetation-free areas for regeneration whereas thinning removes competition between neighbouring trees during years with high N demand from the vegetation. Second, the soil in larger gaps may be exposed to the sun with microclimatic effects on mineralisation and nitrification (Joslin and Wolfe, 1993).

6.8. Other management options

6.8.1. Road constructions, fencing, other infrastructure

In mountainious or hilly areas building of infrastucture in particular roads both permanent roads and temporary trail for harvesting have impacts on hydrology and may lead to erosion (section 3.7.2). This may potentially also affect water quality most likely by increasing the DOC leaching. The impacts are highly dependent on the local conditions (slope, soil type, rain pattern etc.) as well as on the precautions taken in road building, and therefore difficult to predict.

6.8.1.1. Buffer strips protecting riparian zones.

In N-saturated agricultural lands buffer strips and small wetlands along streams are known to have vast potentials for denitrification and N retention that are used in environmental management (Hefting and de Klein, 1998 and references therein). The riparian zone in forests (the transition zone from upland to wetland) can effectively remove nitrate by denitrification (Ashby et al., 1998; Konohira et al., 2001; Lowrance, 1992) and by vegetation uptake (Bischoff et al., 2001) as well as reduce sediment and DOC transport. As discussed in section 6.2, ditching is a common practice in forestry that may impair the riparian N sink and increase DOC leaching (Lepistö et al., 1995). Protection and restoration of riparian zones and possibly creation of small wetlands may be an option to protect aquatic systems from N leaching (Polyakov et al., 2005). The complex issue of riparian management in forests was recently discussed by Verry et al. (1999) and reviewed by Broadmeadow and Nisbet (2004).

6.9. Risk of natural hazards promoted by management

Forest management may have indirect impacts on water quality due to the effects of management on the risk and frequency of natural disturbances of the plant cover such as windthrow, pest diseases, avalances and fires. One example is the severe level of forest damages from recent hurricanes that are thought to be aggravated by the use of exotic conifer trees in even-aged monocultures in central and northwest Europe. The impact on water quality from windthrown forests may be comparable to that of clear-cut management, since salvage logging often with some delay will appear as a clear-cut. In section 6.6.1 and Table 6.2 we include the information from clear-cut plots that originated from windthrows and there were no signs of differences in water qualty impact compared to commercial clear-cutting. However, an imortant difference is the scale of the impact, where commercial clear-cuts affect some hectares within a particular forest distric hurricane windthrow may affect most of that district. If it is situated on a groundwater resservoir the whole reservoir will experience elevated nitrate leaching for 3-5 years, which potentially may impair water quality.

Another example could be the risk that moncultures enchanced development of insect pests. When insects defoliate or kill trees over larger areas the effects on N leaching may be similar to those of windthrow and clear-cut. A recent bark beetle attack damageing 85% of the Norway spruce forest in the highlands of Bavaria, Germany illustrated this potential risk (Huber et al., 2004, Huber, 2005). Elevated nitrate leaching was observed in all damaged stands investigated at least up to 7 years after the death of the trees, which is longer than after conventional clear-cut. The nitrate concentration increased from almost zero to 7 mgN L^{-1} and stayed at this level for 5 years (Huber, 2005) and the total loss of N by nitrate leaching was above 500 kgN ha^{-1} for the 7 year period covered by measurements (Huber et al. 2004). At the same time the leaching of Al increased 10 fold (Huber et al., 2004). Similarily elevated nitrate leaching was observed in USA following the spreading of a gipsy moth infestation (Lovett et al., 2002).

In some areas, fire suppression may contribute to the development of N saturation by allowing N to accumulate in soil and forest floor organic matter. This effect of fire suppression has been shown in fire-adapted mountain ecosystems in California (Fenn et al., 1996). Beside the direct loss of soil organic N by burning, the fire will initiate resprouting and regeneration of a new forest vegetation that will be a strong sink for N, as observed by Goodale and Aber (2000) even 100 yrs after the last burn. Prescribed burning has been suggested as a tool to manage soil N storage (e.g., Fenn et al., 1998). This option is relevant in forests that have developed with fire, but settlement often restricts the use of burning in management. This issue may be particularly relevant for the meditarainian regions, however fire suppression is part of forest management in almost all regions in Europe and small prescribed burning could be used as management tool. As for the application of wood ash, burning may increase soil pH and stimulate nitrification with the potential for a temporary increase in nitrate leaching after the fire.

6.10. Synthesis

In this chapter we reviewed the impact of forest management operations on water quality with a main focus on nitrate leaching.

Deciduous and coniferous forest types react differently to the moderate to high N deposition in parts of Europe, since deciduous forests among other factors are on richer soils. When established on the same soil on adjacent sites, conifer forests receive higher N deposition and exhibit higher nitrate loss than deciduous forests. An exception is alder forests, which show substantial nitrate leaching, due to N-fixation inputs. There were indications that there may be significant differences in the cycling and N retention capacity among broadleaf tree species.

Fertilisation with N have ceased in regions with elevated N deposition. In other regions fertilisation poses limited risk to water quality, when applied to N-limited forests. Fertilisation with P and K may be used to restore the plant N sinks in forests already leaching N.

Responses in nitrate leaching after liming with more than 3 t ha^{-1} may increase with N deposition and in older stands. A nitrate leaching response to soil warming was found at a N-saturated site and not at an N-limited site. However, data on these types of N cycle disruption are too sparse to allow general conclusions on controlling factors.

Nitrogen cycle responses to clear-cut are well studied. Nitrate losses peak after 2-3 years and are back to pre-cut levels after 3-5 years. The extent and duration of the nitrate response seems connected to the recovery of the vegetation sink. Also the harvest intensity is of importance. There is evidence that N losses increase with deposition and are higher at N-rich sites. A threshold for elevated risk of nitrate losses after disturbance was indicated at c. 50 kg N ha^{-1} yr^{-1} of litterfall N flux in the intact forest before disturbance. Less intensive disturbances like thinning and selective harvest (of up to 60% of the trees) have only minor effects on N loss.

The variability of N status between forests can in part be an effect of former century-old land-use patterns with forest utilisation (e.g. logging and litter raking) decreasing N status and cultivation and agricultural use increasing N status (and decreasing C content). These old land-use impacts will translate to differences in mineralisation rates and other N status parameters. Forests now planted on modern agricultural soils will most probably appear as N-saturated forests.

6.10.1. Recommendations for future research

During our work with this synthesis we have identified several gaps in our ability to understand and predict nitrate leaching from forests. Also there is a need for better information on DOC leaching in relation to management. In the following we briefly discuss areas requiring detailed future research.

The choice of tree species influence water quality both directly through differences in N-cycling and indirectly through differences in evapotranspiration (Chapter 5). Although differences between conifers (mainly Norway spruce) and broadleafs (mainly beech) are well documented there a need for more comparisons of N cycling among tree species grown on the same soil. Differences among broadleafs, but also among conifers, need to be investigated. We found some indications that such differences could be significant and important for making decisions on replanting.

The temporal variation of nitrate leaching (or N budgets) throughout full rotations should be studied to get information on the full rotation average. In perspective of groundwater protection it may not be a problem with high N losses right after clear-cut if there are long periods in the aggrading phase with low (or zero) losses. Observations from chronosequences could be evaluated along with modelling of rotation length N budgets. This also involves aspects of rotation length. There may be trends to reduce rotation length to increase biomass removal and the consequences hereof need to be evaluated. The information on rotation time scales is particularly needed for the evaluation of management alternatives (Table 1.3). Since futher work is needed on this issue, we only made a first qualitative evaluation of management alternatives in chapter 7.

The impact of the contemporary changes in forest management strategies focussing on forest restoration (including conversion of conifer plantations to deciduous forests, liming, re-establishing natural hydrology, and near-to-nature management) is not well known, although these changes are assumed to be beneficial to the environment.

In Europe, N deposition have stabilised or slightly decreased in some regions, but for major parts of Europe particularly those dominated by agriculture the N deposition will continue to be elevated. Thus options and strategies for managing or improving forest N retention may be relevant in these regions to sustain or improve the protective function of forest on water quality. Such strategies, including replacement of conifers by deciduous trees, shorter rotations, continuous cover, whole-tree-harvesting, fertilising with limiting nutrients, prescribed burning, and protecting/improving the riparian zone, were discussed briefly in Gundersen et al. (2006). But these options are not well documented. More documentation and practical demonstration of these strategies is needed to meet the challenge of protecting the good quality of forest waters in regions of excess N deposition.

Table 6.2. Nitrate concentrations in seepage or stream water (mg N L^{-1}) after clear-cutting in European forests. Studies on sites cleared after windthrow are included as well.

Clear-cut		**Species**	**Control [a]**	**Year 1-2**	**Year 4-5**	**Note**	**Reference**
			----------------*mg N L^{-1}*		-------------		
S. Sweden	Ground-water	Low site qual., n=3	0	0.3 – 1	0.3 – 1	Some delay (one year) assumed since concentrations are for groundwater.	Wiklander 1983
		Medium site qual., n=3	0	1 – 2	0.3 – 1		
		High site qual., n=1	0	4	0		
S. Sweden, regional study	Seepage	Conifers, n=29	0-0.5 [b]	1-6		Mean over first 4-5 yrs; conc. increase with N deposition in the range 15-25 kg N ha^{-1}y^{-1}	Akselsson et al. 2004
Gisburn, NW England, UK	Seepage	Oak	0	<0.5			Robertson et al. 2000
		Alder	3.4	1.4			
		Scots pine	1.3	4			
		Norway spruce	0.5	5.2			
Beddgelert, N.Wales, UK	Seepage	Sitka spruce	0.5 -0.9	1.7 – 5.4	0.3	Flux change 10 to 70 kg N ha^{-1}y^{-1}	Stevens and Hornung 1988; Reynolds et al. 1992
Plynlimon, Wales, UK	Seepage	Sitka spruce					Reynolds et al. 1992
		- podzol soil	<0.1	3.9 – 4.6			
		- gley soil	1.1	0.7 – 1			
	Stream		0.5	1.9			
Kershope, N. England, UK	Stream	Sitka spruce	1.3	3.6 – 4.3	2.2		Reynolds et al. 1992
Afon Hafren, Wales, UK	Stream	Sitka spruce	0.5	3.4	0.9	Similar increase in nitrate conc. was observed in groundwater	Neal et al. 2004
Ravels State Forest, Belgium	Seepage	Scots pine	~7	21		High deposition, 52 kg N ha^{-1}y^{-1}	Keersmaeker et al. 2000
Eiseneck, Austria	Seepage	Norway spruce	~0	3-6		Input 8 kg N ha^{-1}y^{-1} Flux change 0.5 to 15-27 kg N ha^{-1}y^{-1}	Katzensteiner 2003
Convent Forest, Black Forest, SW Germany	Seepage	Norway spruce		15	2-3	Input 25 kg N ha^{-1}y^{-1}, details missing	Wilpert et al. 2000
Ebersberg Forest, Bavaria, Germany	Seepage	Norway Spruce	0-2	17	1-3	Input 25 kg N ha^{-1}y^{-1}, windthrow Values up to 20 mgN L^{-1}	Rothe et al. 1998; Weis et al. 2001
Eurasburger Forest, Bavaria, Germany	Seepage	Norway spruce	(11)	14		Input approx. 30 kgN/ha/yr. Growing stands in the area is used	Rothe et al. 1999

						for control.	
Windthrown, site 10, Bavaria, Germany	Seepage	Norway Spruce	<5	10	4	Windthrow	Mellert et al. 1998
Münchner Schotterebene, Germany	Seepage	Conifer, n=3	0-2 [b]		6-10	Survey in forest distict. Input 20 kgN/ha/y	Rothe and Mellert 2004
Flossenbürg, Germany	Seepage	Norway Spruce	1.7	5.6		Input 19 kg N $ha^{-1}y^{-1}$	Weis et al. 2001
Höglwald, Germany	Seepage	Norway Spruce	5.6	14	3	Input 31 kg N $ha^{-1}y^{-1}$	Huber et al. 2004
Pfarrwald-Michelbach-Bilz, Germany	Spring water	Norway spruce, some Oak and Beech	2.6 3.7	4.2 4.7		Windthrow	Schlär 1999
Korfdorf Forest, Giessen, Germany	Stream	Beech	2.7		4.7	Stepwise cut over 5 years. Flux change 4 to 14 kg N $ha^{-1}y^{-1}$	Hüser et al. 1996
Mont-Lozere, S. France	Stream [c]	Norway spruce	0.2 - 0.4	2 - 4		Flux change 3.9 to 35 kg N $ha^{-1}y^{-1}$	Durand et al. 1992

[a] Uncut control or pre-treatment concentration

[b] range for intact forests in the region or forest district

[c] concentrations were roughly estimated from fluxes, actual year water fluxes were not reported

Table 6.3. Seepage water nitrate concentration (mg N L-1) after forest floor liming (or wood ash application). Fluxes are given in ( ) whenever available.

Site, stand type	**Dominant tree species**	**Soil type**	**C:N ratio forest floor**	**Rainfall**	**Nitrogen deposition**	**Lime treatment Dose (and type)**	**Nitrate in control stands**	**Nitrate in limed stands**	**Treatment effect**	**Reference**
				mm	*kg N* $ha^{-1}yr^{-1}$	*t/ha*	*mg* NO_3*-N* L^{-1} *(kg N* $ha^{-1}yr^{-1}$*)*			
Seepage										
Klosterhede, 56 years Denmark	*Picea abies*	Haplic podzol	26	860	15	1.7 ($CaCO_3$) 0.7 ($MgSO_4$)	0.01	0.01-0.74	Yes	Ingerslev 1997
Adirondack Mountains, USA	Deciduous	Spodosol	24-49	1230		6.9 ($CaCO_3$)	1-2 (soil water) 0.1-0.4 (stream)	1-5 0.4-1.4	 Yes	Geary and Driscoll 1996, Cirmo and Driscoll 1996
Uddevalla, Sweden Vallsta, Sweden	*P. abies, Pinus sylvestris*	Podzol Podzol	n.d.			1 ($CaCO_3$) 1 ($CaCO_3$)	0.2 <0.03	0.2 <0.03	No No	Nohrstedt 1992
Hasslöv, Sweden	*P. abies*	Haplic podzol	20-27	1100		3.45 ($CaCO_3$) 8.75 ($CaCO_3$)	0.27 (1) 0.27 (1)	0.98 (5) 1.99 (10)	Yes	Persson and Wiren 1996, Nilsson et al. 2001
Öringe, Sweden	*P. abies*					3.35 ($CaCO_3$)	(3)	(10)	Yes	Persson and Wiren 1996
Åled, Sweden	*P. abies*	Dystric Regosol			14	4.2 (wood ash)[a]	0-1	0-6	Yes	Högbom et al. 2001
Grunewald, Germany	*Pinus sylvestris*	Cambic Arenosol	28	580	12	6.1 (dolomite)	8.2 (27)	12.6 (44)	Yes	Marschner et al. 1992
Pfälzerwald, Germany	Pine and Beech					3 (dolomite)	0-0.5	0-0.5	No[b]	Schüler 2001
Höglwald, Germany	*P. abies*	Alfisol	23-27	850	30	4 (dolomite)	~15 (33)	~30 (55)	Yes	Kreutzer 1995, Kreutzer and Weiss 1998
Convent Forest, Germany	*P. abies*	Cambisol		1400	~25	2.4 (wood ash)[c] 12 (wood ash)[c] 18 (wood ash)[c]	~3	~1 ~3-7 >10	No Yes Yes	Shäffer et al. 2002
Swiss Plateau, Switzerland	*P. abies*	Dystric Cambisol	16	1076		8 (wood ash)[d]	~2 [e]	~10	Yes	Brunner et al. 2004
St. Anthonis, Netherlands	*Pinus sylvestris* *Pseudotsuga menziesii* *Quercus robur*	podzol	20-24	450		3 (dolomite)	(1.6) young pine (6.9) old pine (3.6) young fir (13.9) old fir	(10.8) (12.7) (12.9) (36.5)	Yes	De Boer et al. 1993

Site, stand type	**Dominant tree species**	**Soil type**	**C:N ratio forest floor**	**Rainfall**	**Nitrogen deposition**	**Lime treatment Dose (and type)**	**Nitrate in control stands**	**Nitrate in limed stands**	**Treatment effect**	**Reference**
				mm	*kg N* $ha^{-1}yr^{-1}$	*t/ha*	*mg* NO_3*-N* L^{-1} *(kg N* $ha^{-1}yr^{-1}$*)*			
							(2.5) young oak (9.4) old oak	(1.1) (17.6)		
Stream water										
Hubbard Brook, USA	Mixed broadleaf	podzol				0.85 Ca (wolastonite)	0-4[f]	0-2[f]	No	Groffman et al. 2006
Gjastad, Norway	*P.abies* *P.sylvestris*	podzol	19-21	1200	15	2.9 (dolomite)	0.07(0.7)	0.07(0.7)	No	Hindar et al. 2003

[a] Ca content equivalent to 3 t lime/ha

[b] Except for a few episodes.

[c] Ca content equivalent to 3, 15 and 22.5 t dolomite/ha, respectively.

[d] Ca content equivalent to 6 t lime/ha

[e] Pre-treatment nitrate conc.; Control plots had higher pre-treatment nitrate conc. which decreased over the 4 yrs of observation, whereas the concentration increased at the wood ash treated plots.

[f] Both the control and the Ca-treated catchments had declining concentration throughout 5 years of monitoring, due to recovery from a disturbance (ice storm damage).

7. Synthesis

by

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The nature forest reserve, in which the objective is old-growth forest including all natural disturbances and without any management at all will serve as our reference for judgement of management operation effects on the environmental services (e.g. Peterken, 1996).

Several forest management operations cause only local and short-term effects on the services whereas others cause significant effects both in time and space. The single most thorough manipulation of the system is introduction of tree species outside their natural distribution which cause fundamental changes in the authentic biodiversity relying on long-term continuity (Schnitzer and Borlea, 1998). Although succession and adaptation to changes in the environment continuously takes places, the influences of such changes are mostly minor compared to the effects of introduction of alien species. Tree species also have an impact on the water balance (transpiration, interception and run-off), the acidity and nutrient content of water leaving the system, on carbon stocks and sequestration, and on soil acidity. From an environmental point of view, negative impact can often be related to introduction of conifers on land naturally covered by beech or mixed broadleaves.

At the time of stand regeneration many single operations like soil tillage and weed and pest control will be carried out. Most of these operations are part of intensive forest management alternatives aiming at establishment of homogenised even aged stands containing only one or few tree species. Such stands are also often regularly thinned and regenerated following a traditional clear-cut. The single isolated operations may not have great impact on the services but together as part of an intensive management alternative the gross effect may be significant, mainly on biodiversity. Water quality and water quantity indicators will be negatively influenced as well but to a lesser extent depending on the impact of operations following the introduction of alien species. In some cases however, e.g. as a consequence of conifers introduction on sites natural for mixed broad leaved species, carbon sequestration may increase significantly whereas carbon stock on site may decrease.

Harvesting of biomass influences the biodiversity negatively due to less dead wood left in the forest. Furthermore, harvesting cause compaction, export of nutrients and soil acidification depending on the intensity of the biomass export. A decrease in the nutrient capital might be crucial where the natural restoring capacity is limited. However, compensation both for nutrient export and soil acidification by fertilisation and liming are possible to reduce the negative impact. Large clear-cuts might have great impact mainly due to lack of forest climate, return the forest succession and loss of nutrient and possible soil particles. Rotation length considered as a single parameter might not have great impact on the services. For biodiversity, retention of sufficient mature trees might be more important.

7.1. Effect of forest management alternatives

In table 7.1 the effects of the five management alternatives described by Duncker et al. (2007) and in table 1.3 are evaluated by use of a score system where the nature reserve is reference (score 0). The five management alternatives will be used in the EFORWOOD project scenarios analysis.

Table 7.1. Summarised most likely gross effects of the forest management alternative defined in table 1.3 on biodiversity, carbon stocks and sequestration, water quality and quantity, and soil quality indicators defined in table 1.1.

Forest management alternatives	Biodiversity	Carbon		Water quality	Water quantity	Soil quality
		stocks on site	seque-stration			
Forest nature reserve	0	0	0	0	0	0
Close to nature forestry	-/0	-/0	0/+	0	0	0
Multifunctional Forestry	-/0	-/0	-/0/+	-/0/+	-/0/+	-/0
Intensive even-aged forestry	-	-	-/+	-/0	-/0/+	--/0
Wood-biomass production	--	--	--/++	--/0	--/0	--/0

0 no gross effect, the reference
- and -- a moderate and a highly significant negative influence
+ and ++ a moderate and a highly significant positive influence

7.1.1. Biodiversity

It is possible to manage low intervention and multiple purpose forestry without significant negative influences on biodiversity, but there might be a risk, especially for the multiple purpose alternative if goods for other purposes such as recreation are problematic for biodiversity. Intensive and wood-biomass forestry might always influence biodiversity negatively. On the other hand, planted forests might also restore biodiversity in degraded landscapes. The most significant negative influence on the biodiversity might be due to introduction of alien tree species and intensive harvesting leaving no dead biomass and soil tillage.

7.1.2. Soil quality

Nutrient stores in the nature reserve are mostly relatively constant because the exchanges to the surroundings are modest. Air pollution has negative influences due to an increase in nutrient leaching from the system and acidification but can often not be attributed to forest management. Harvesting of biomass might cause a significant decrease in the soil content of almost all nutrients and an increase in soil acidification depending on the weathering capacity of the soil minerals and the kind and intensity of biomass removal. Change in tree species might accelerate the negative nutrient balance and acidification both due to increase in biomass harvesting and increased deposition of air pollution compounds. Negative nutrients balances and acidification might be counteracted by wise fertilisation and liming but this is seldom done. On steep slopes clear-cut or intensive harvesting might be problematic due to a potential increase in loss of soil material due to erosion. Modern intensive forestry often includes heavy machine trafficing with possible negative influence on soil structure due to compaction.

7.1.3. Carbon sequestration and stocks on site

In the long-term, the nature reserve should in principle have a balance in carbon sequestration because photosynthesis and autotroph and heterotroph respiration balance each other. However, this is barely observed in the short-term since environment is not stationary (climate change, nitrogen deposition, ozone deposition) and may shift significantly the carbon balance positively or negatively. The stock on site in the ecosystem, both in biomass, dead wood, and the soil are often very high in natural reserves compared to managed forests. On the other hand, management increases productivity through a more or less intense rejuvenation of the tree stand. Shortening of the rotation age and rejuvenation increases photosynthesis through site improvement and fertilisation and it will definitely decrease the heterotrophic respiration because biomass is removed from the system. This might, on the other hand, reduce the amount of carbon stocks in the biomass and soil. Very intensive forestry alternatives offer a possibility for a very significant increase in carbon uptake despite a decrease in soil stock. Site preparation practices such as ploughing or drainage might be a special problem because very large soil stocks due to long-term built up might be released very fast.

7.1.4. Water quantity

The evatransporation from nature reserves is often relatively high due to a permanent high plant cover and transpiration when the temperature is above a certain threshold. Moderate negative influence from forest operations might be attributed to tree species manipulation, e.g. when broadleaves are converted to conifers or high consuming eucalypts. Clear-cuts, depending on size, will increase the runoff and may cause local temporary flooding. It might be possible to manage forests aiming at high surplus of water, e.g. by introducing tree species with a low leaf area index and by relatively high thinning rates. Increased water consumption due to fertilisation may occur due to increased interception losses by increased leaf area.

7.1.5. Water quality

The content of problematic compounds from nature reserves is mostly low. Mainly compounds coming from outside sources, e.g. nitrogen due to air pollution might be a problem. But this may not be attributed to the forest or forestry *per se*. Short-term negative influence on water quality might be due to clear-cut, intensive soil tillage, drainage, excessive fertilisation or use of nitrogen fixing tree species. Very intensive forestry alternatives might offer specific problems if e.g. pesticides are used unwise. In the case of regions with elevated N deposition (>10-15 kgN/ha/yr) water quality may be impaired. Forest management such as choice of conifer species, clear-cuts and heavy liming may aggravate this, but on the other hand, conversion of conifers stands to broadleaf stands, multiple age management, and increased utilisation may be used to mitigate effects of air pollution on forest waters.

7.2. Trade offs

A management alternative aiming at optimising environment services as defined in this report is not unambiguous. In general, the nature forest reserve by definition offers the reference biodiversity, and high carbon stocks and clean water. Nature forests are also mostly both highly resistant and resilient which secure ecosystem functions also in the future. However, mature nature reserves did not sequester much carbon and the water run-off might also be slightly lower than some alternatives characterised by a lower leaf area index. If carbon sequestration has highest priority the very intensive alternatives might be optimal. On the other hand, such alternatives generally have lower carbon stocks in the system because harvesting and other operations might cause a rapid release of a part of the carbon stock. Therefore, a specific statement including running sequestration and decrease in stock is necessary for short-term evaluation of gross carbon impact.

Air pollution also influences nature reserves and there are clear indications that especially the no harvesting regime in areas with high nitrogen load may cause leaching of nitrate and hence accelerated soil acidification (Ritter and Vesterdal, 2006). Such negative impacts might be counteracted by biomass harvesting and fertilisation.

7.3. Uncertainties and gaps

Although a huge number of experiments have been conducted aiming at quantification of impacts of management operations on the services several gaps still exist. The most fundamental problem for a synthesis like this is the short-term perspective of the experiments aiming at explaining effects that often have long-term impacts. Several examples of such legacy effects are described e.g. for soil tillage and changes in tree species on biodiversity. Furthermore, most of the investigations and experiments rely on a reductionistic research approach whereas the impacts of the operations are on a system or a landscape level and should be assessed as such. Forest reserves for references are few and most if not all of them have been or are still influenced by man.

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