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Preamble

This deliverable gathers five individual manuscripts intended for publication in a special issue of Annals of Forest Science. Those manuscripts are derived from work undertaken in WP2.2 as part of sustainability impact assessment of forest management alternatives (as defined in WP2.1) and specifically deal with impacts on five main environmental services from forests. Those manuscripts are ready for submission (1) or under final revisions (2, 3, 4, 5) before submission.

The manuscripts are entitled:

1. The impact of forest management on biodiversity, by de Jong et al.
2. Impact of forest management alternatives on water budgets and runoff processes, by Katzensteiner et al.
3. The impact of forest management on soil quality, by Hansen et al.
4. The impact of forest management on water quality, by Gundersen et al.
5. The impact of forest management on the carbon cycle, by Loustau et al.

Key words forest management operations; forest management alternatives; impact indicators; biodiversity; carbon sequestration; soil quality; water quality; water quantity.

Table of contents

1	THE IMPACT OF FOREST MANAGEMENT ON BIODIVERSITY	3
2	IMPACT OF FOREST MANAGEMENT ALTERNATIVES ON WATER BUDGETS AND RUNOFF PROCESSES.....	27
3	THE IMPACT OF FOREST MANAGEMENT ON SOIL QUALITY.....	56
4	THE IMPACT OF FOREST MANAGEMENT ON WATER QUALITY IN EUROPE –.....	90
5	THE IMPACTS OF FOREST MANAGEMENT ON THE CARBON CYCLE.....	120

1 The impact of forest management on biodiversity

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Abstract

Biodiversity is fundamental for the delivery of forest based environmental services. Identifying the management alternatives that can deliver sustainable forestry (with even higher production than today) while meeting conservation objectives is of great importance to both policy makers and forest managers. The different forest management alternatives are comprised of a range of management prescriptions which can then be applied to the tree, stand or forest unit. These operations take place over the life cycle of the forest from regeneration to harvesting. The aim of this paper is to present an evaluation and synthesis of the known effects of specific forest management operations on biodiversity.

This review clearly shows that the impact on biodiversity by different forestry activities is well known. However, it is also known how it is possible to the impacts, and there are a number of examples on how to combine forestry and biodiversity conservation in the landscape. The most important of these is to attempt to mimic natural disturbance regimes which can improve structural complexity. Landscape considerations take account of habitat loss through fragmentation which can be mitigated against through restoring connectivity through habitat networks. Measuring the impact of the different management alternatives requires biodiversity indicators that can help inform our understanding of the ecological dynamics and conservation value of tree s and stands within a forest

Introduction

Forests and other wooded land cover roughly 30 % of the total land area of Europe and deliver a wide 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 the concept of sustainable management is in place, and 25% of the forest area is now under some form of certification that includes measures 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 (over-aged) forest, climate change and pressure for intensification of forest utilization 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).

The aim is to present an evaluation and synthesis of the known effects of specific forest management operations (Table 1) on biodiversity and to rank the importance of the different effects pointing out the most important and influential management operations.

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 (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 policy 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 Spannios 2008). 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. 20-40m³/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. Landscape ecology is defined as “the study of the interactions between the temporal and spatial aspects of a landscape and its flora, fauna and cultural components” (Dover and Bunce, 1998). The term landscape ecology was first coined by the German biogeographer Carl Troll at the end of the 1930s (Farina, 1998). Troll hoped that a new science could be developed that would combine the spatial, ‘horizontal’ approach of geographers with the functional, ‘vertical’ approach of ecologists. Landscape ecology also occupies an important bridge between pure and applied ecology, with great potential for the integration of emerging theories e.g. island biogeography, metapopulation models and how these can be applied to address issues such as habitat fragmentation.

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 have been diminished. 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).

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.

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. *Nothofagus 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 almost 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 micro-arthropod 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 correlated 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 and structure of plantations can be extremely beneficial to biodiversity. For example in the UK, naturally created gaps in upland spruce forests are often colonized 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 tree 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).

Site preparation

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* (von Sydow 1997, Örländer and Nilsson 1999). 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.

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 (short rotation forestry, e.g. for production of biofuel or Christmas tree production) or space (nurseries) the pesticide

usage is also intensive (Ravn and Andersen 1997). 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.

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). Mycorrhiza 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 (Wikars 1992, Ehnström 1997). The fungal pathogen *Rhizina undulata* gets virulent when exposed to temperatures 35-45°C (Petersen, 1971).

Stand management and harvesting

Clear-cut system

Large scale 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 (Schroeder et al. 2006). 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, therefore restoration of biotopes by using fires is still important (Niemelä, 1997).

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 (Matthews, 1991).

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. Mycetophilidae, Ökland, 1994).

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 have different impact on 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, of several, important differences 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., 2004). 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 (Kruys and Jonsson, 1999; Nordén et al., 2004).

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 commercial forestry (Christensen and Emborg 1996). 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 (Rune, 2001). Peterken et al. (1992) and Humphrey (2005) have recommended extending rotations in temperate plantations to benefit species associated with late succession 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.

Other management options

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.

The exclusion of grazing animals from areas of woodland by fencing can have both positive and negative effects on biodiversity. Natural forest ecosystems include some level of grazing by large herbivores, though to what degree is widely debated (Vera 2000 and Hodder et al 2005), and the impact of grazing as a natural disturbance varies a lot between different forest ecosystems. Deer fencing is used to encourage and protect tree regeneration but can lead to a decline in ground flora diversity. This results in the development of a dense sward and the build up a vegetation mat, which ultimately can inhibit the regeneration the fencing was designed to promote. Bird strikes on deer fencing can effect populations of iconic and protected species such as Black Grouse (*Tetrao tetrix*) and Capercaillie (*Tetrao urogallus*) as well as a range of raptors. In boreal forests the moose is a key species and its browsing creates opportunities for a number of species (Persson et al. 2000, Edenius et al. 2002). On the other hand a very high population density of moose is a threat both against forest production and biodiversity. In some cases fencing is the only possibility to produce aspen and rowan, which are important for conservation.

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.

Possibilities to decrease the negative impact on biodiversity

Mimicking natural disturbance regimes

Both natural and cultural disturbances have varied during history, and the result of this is a dynamic ecosystem, which can be very difficult to restore or conserve (Lindbladh and Bradshaw, 1998; Lindbladh et al., 2000; Lindbladh et al., 2007a). 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 (Lindbladh et al., 2003; Granström, 2001). 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 (Mielikäinen and Hynynen, 2003). 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.

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 taxa (Hedenås and Hedström, 2007; Schroeder et al., 2006; Kaila et al., 1997). 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 be 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 (Mielikäinen and Hynynen, 2003). 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.

Landscape considerations

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 (Villard et al., 1999).

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 should be varied at the landscape scale simply to improve general structural heterogeneity (Berg et al., 2002). However, careful planning is required to ensure a balance between this aspiration and the need to ensure conservation of specific priority species and spatial-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.

Some types of forestry have big negative impact on biodiversity, e.g. intensively managed forest plantations. In general the number of species in these habitats is very low and it is dominated by a few species with general habitat requirements. However, on the landscape level the significance of intensively managed forest plantation depends on how large an 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 limited 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.

Habitat loss, fragmentation and restoring habitat networks and connectivity

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 (Berg et al., 2002). 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 (Berg et al., 2002). 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 (Leroux et al., 2007). 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 biodiversity in general also is 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 (Brunet and von Oheimb, 1998). 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 (Sverdrup-Thygeson and Lindenmayer, 2002). 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 (de Jong et al., 2004; Bütler et al., 2004). Further development and scientific evaluation is needed.

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.

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 forest management 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 issues which are important for biodiversity regardless of context. The biggest threat against biodiversity is loss of the structural diversity and of forest key

habitats (this includes plantation forestry including alien species), leading to fragmentation, leading to dispersal- and genetic problems. Another threat is lack of suitable management of protected areas. Some species adapted to very specific natural disturbances is not possible to combine with forestry. A network of protected areas in which natural disturbances are allowed must be included in forestry strategies also in the future. However, for most species forest management and species conservation can be combined. In order to succeed with this important actions are:

- Develop strategies on the landscape level
- Identify important key biotopes, e.g. areas with long continuity of tree cover or continuity and high abundance of specific substrates. These areas should be set aside or managed with some kind of adapted forestry methods.
- Increase the connectivity between stands. In some cases this means that restoration of biotopes is the most urgent action
- Increase the structural diversity in the stand (increase the abundance substrates such as dead wood, big trees, old trees and increase the number of tree species)

With regard to the sustainability of forest management it is important also to consider the specific role of the intensive forest management and biomass production alternatives, which are somewhat out with the envelope of multi-purpose forestry since there is an overriding focus on timber or biomass production. The consequences of these forms of forestry depends mainly on the location in the landscape, the area it occupy and what kind of biotope that has been removed.

Clear-cutting is the most obvious shift in the forest succession, and a lot of discussions on conservation have focused on the consequences of clear-cutting on the biodiversity. One result of this discussion is that nowadays some trees and dead wood are left on the cut area for conservation purposes. However, this review demonstrate that other forestry operations, such as drainage, thinning and the use of alien species, might have as high or higher impact on the biodiversity.

The conclusion of this review is that even though more information is needed to increase the efficiency of conservation in forests (see below) the knowledge of the impact of different forestry operation on biodiversity is well known for a large number of organisms. It is also shown that it is possible to combine conservation of species with efficient forestry.

Biodiversity indicators

In order to combine forestry with conservation it is necessary to identify and set aside hot-spots, as well as important structures for biodiversity (Lindenmayer et al., 2000). One possible solution is to use different indicators and there is widespread interest in the use of biodiversity indicators to represent and monitor complex phenomena and processes that are not susceptible to complete description. There is a need to identify and to make these indicators operational of ecosystem and biodiversity responses to both environmental change, and changes in forest policy and practice. Climate change imposes new challenges for the effective monitoring of ecosystems and biodiversity. To date there has been a focus on modelling species specific responses but for forest ecosystems there is a need to develop ways of assessing changes in the structure and composition of communities and to evaluate how climate change will amend biodiversity through range shifts with consequent changes in existing communities and the development of novel assemblages.

Methods vary according to requirements, and range from measures which function as “Barometers” of change to those which provide indicators of specific aspects of biodiversity, or the impact of changes in forest management on ecosystem services. The indicator concept uses characteristics of an easily measured feature such as an organism or aspect of forest structure as an index of certain attributes of the system that are otherwise too difficult or expensive to measure directly.

Biodiversity indicators can be summarised into 3 categories:

1. Structural indicators which are usually easily quantifiable through forest inventories and can be described as: physical pattern, spatial pattern and temporal pattern,
2. Compositional biodiversity indicators are usually developed through gathering data in the field: species diversity, genetic diversity and biotope diversity,
3. Functional biodiversity indicators that allow us to understanding of ecosystem function of forest biotopes: ecological process, natural disturbance and nutrient cycling.

To be effective functional biodiversity indicators should be simple and not expert based systems. They should combine the structural and compositional elements of forest biotypes and the ecological processes that drive them. They need to be able to work effectively in the different management alternatives applied to them. This would allow the complex ecological phenomena and processes to be monitored

Gap of knowledge

1) Stand level management

Biodiversity can be enhanced in forests by modifying structure and composition (Quine et al., 2007). It has been proposed that management methods based on mimicking natural disturbance regimes should be developed as a means of diversifying structure and composition (Quine et al., 1999; Bengtsson et al., 2000; Franklin et al., 2002; Humphrey, 2005). However, this assumption remains to be tested in different forest management alternatives. Potential methods range from modifying harvesting practices such as the use of Variable Retention harvesting (e.g. Aubry et al., 2004), to selective-cutting-methods and other lower impact silvicultural systems including the use of variable density thinning (Carey, 2003).

Clear missing components of most managed forests are large old trees and decaying wood, but the role of these as habitats for native species needs to be investigated (Humphrey, 2005). More research is also needed on the benefits of diversifying tree species composition (Hartley, 2002).

2) Landscape level planning

There is increasing evidence that plantations can make a positive contribution to biodiversity at the landscape scale (Brockerhoff et al., 2008). Research is needed on the impact of the spatial configuration of different stand types on the movement and dispersal of key species through the landscape. The facilitation of such movement is thought to be crucial to the the survival of some woodland species affected by climate change (Opdam and Wascher, 2004).

Strategies for landscape planning may include striving for a balance of structural phases at the landscape scale, but this assumption remains to be tested, and it may also be the case that a focus on key structures is necessary, e.g the spatial configuration of old-growth patches (Humphrey 2005).

3) Indicators and thresholds

There is an urgent need to develop guidance for managers on the development of indicators of biodiversity (Ferris and Humphrey, 1999). Although progress has been made in this respect (Smith et al., 2008), there is a need for experimental testing of the efficacy of these indicators and the development and validation of rapid assessment methodologies.

The issue of thresholds also needs to be investigated (Eiswerth and Haney, 2001). This is essentially a species-driven process as different species require different threshold levels, for example in the amount and quality of deadwood habitats (Humphrey et al., 2004).

4) Habitat restoration

Plantations of exotic species support novel assemblages of native species (Hobbs et al., 2006), and there is uncertainty regarding the way in which these syncretistic ecosystems will develop over time (Lindenmayer et al., 2008) especially given climate change. Options need to be researched such as methods for “restoring” plantation to the original woodland cover, or for accelerating ecosystem development towards multi-aged, multi-species systems.

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2 Impact of forest management alternatives on water budgets and runoff processes

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Abstract:

One of the major services of forests is the provision of water in sufficient quality and quantity. The current literature review summarizes experimental results on impacts of different forest management alternatives upon water budgets of forest ecosystems and forested landscapes. Impacts of the choice of tree species, stand management and harvesting upon the terms of the water balance equation are discussed. Response of runoff to harvesting is expressed as a linear dependence of annual precipitation and percentage of area cut based on published results from paired catchment experiments. The applicability of hydrological models to predict the reaction of water yield to forest management is shown for generic forest systems.

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 also has a tremendous influence on the protective function of forest ecosystems. Thus a thorough understanding of forest 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 regarding the effects of forests on the water cycle started in the late 19th century by pioneers in forest hydrology such as Engler in Europe (Keller, 1988) or Bates and Henry in the USA (Douglass and Hoover, 1988; McCulloch and Robinson, 1993). The famous paired catchment study carried out in the Swiss Emmental showed increased annual runoff in a sparsely forested watershed in comparison to a fully forested catchment. Furthermore, the regulating effect of forests on flow rates: a decrease of peak flow during rainy periods and an increase of base flow during dry periods, as well as effects on snowmelt were documented (Engler, 1919). A large number of catchment studies have been established since then.

Measurements of runoff at weirs give integral results of water yield for whole catchments. A first review by Hibbert (1967) of 39 experimental catchments has subsequently been updated by Bosch and Hewlett (1982) with 55 additional experiments. More recent reviews by Stednick (1996), Sahin and Hall (1996), Best *et al.* (2003) or Mac Donald and Stednick (2003) more or less confirm the general patterns previously observed in the first investigations: a reduction of water yield with 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 re-growth and site conditions in general cause a high variation in the response of water yield to management treatments. The response of the magnitude of peak flows is even more complex, though a general extenuating effect of forests can be concluded.

Forest management impacts the hydrology of watersheds by alteration of species composition and forest cover and structure on the one hand; and by construction of roads, skid trails and landings on the other hand. In catchment scale studies clear separation between the effects of road construction from the effects of stand treatment is usually impossible. Direct surface runoff from forest roads is mainly directed into ditches, which drain into stream channels. In sloping terrain additional subsurface components of runoff may be intercepted. Both components are delivered rapidly to the stream network and can increase the magnitude of peak flows. Discussion of the significance of these effects is, however, controversial (MacDonald and Stednick, 2003).

Regression based models derived from statistical analysis of results gained through direct measurements are one method used to predict possible management impacts on hydrological processes. Another approach is the use of mechanistic hydrological models to predict 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 concerning the effects of afforestation on water yield (Feger *et al.*, 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 *et al.*, 2007).

Complex meteorological situations, particularly in mountainous terrain; spatial variability of site conditions; and non-homogeneous forest stands still make precise forecasts of the effects of forest management alternatives uncertain. The current review should provide an overview of the present state of knowledge regarding the impact of forest management alternatives on water consumption and modification of peak runoff. A statistical analysis of data from the literature serves as a basis for the general quantification of management impacts upon water yield.

The review should aid the evaluation of forest management alternatives (FMA's) in the European forest wood chain with respect to the environmental services 'water yield' and 'mediating effect on peak discharge'.

2. Materials and methods

2.1 Concepts and indicators

The impact of forest management upon the water budget of forest stands/watersheds is evaluated based on the terms of the water balance equation (Equation 1).

$$\text{Equ. 1} \quad P_o = E_s + I + T + \Delta SW + \Delta GW + Q$$

Po...Precipitation, E_s ...Soil evaporation, I...Intercepted rain evaporation, T...Transpiration, ΔSW ...delta soil water storage, ΔGW ...delta ground water storage; Q...runoff; all units are [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. A second indicator is runoff, as this value is frequently given as an output of paired catchment studies. This term is meaningful under the assumption that over longer periods, $\Delta SW + \Delta GW$ will equal zero.

With respect to the protective function of forested systems, the impact upon surface runoff and peak discharge is also of importance. The analysis of hydrographs from paired catchment studies (control versus treated watershed) provides information on such response patterns. Peak discharge is dependent upon a number of factors – timing and intensity of precipitation, snow accumulation and snowmelt patterns, site factors such as 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. Results gained from watershed investigations are therefore usually of limited general validity.

2.2 Review

The current study includes a literature review of publications regarding the effects of forests and forest management upon hydrological processes. The review includes results pertaining to 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, which affect ET) to fertilisation. With respect to the service 'reduction of peak runoff' few references to published experiments and runoff coefficients are included.

The results are structured into species effects, effects of site preparation and fertilization, effects of stand density (including thinning and harvesting), effects of road construction and effects of land use changes (grassland or arable land to forest). A synthesis is drawn for different FMA's defined by Duncker *et al.* (2010a).

2.3 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 amount of literature has been generated. A literature search on paired catchment studies provided a broad range of data, comprising of reviews (Hibbert, 1967; Bosch and Hewlett, 1982; Sahin and Hall, 1996; Stednick, 1996; Best *et al.*, 2003; Pike and Scherer, 2003). which were completed by additional studies (Jones and Grant, 1996; Troendle *et al.*, 2001; MacDonald and Stednick, 2003; Robinson and Dupeyrat, 2005).

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

The available data were hierarchically classified regarding the affiliation to the Köppen-climate classification, the size of the investigated drainage area, the vegetation type and the management

practices. The focus was laid on climate zones C, temperate and D, continental climates according to (Köppen, 1931).

The response of runoff to changes in 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, as has been done by Sahin and Hall (1996).

2.4 Modelling approach

To show the applicability of physically based hydrological models as support tools for forest management decisions, modeling results for generic forest systems of different species (beech vs. spruce) and thinning intensity under steady site conditions are provided. The hydrological model BROOK90 (Federer, 1995) was used to calculate water balances for the generic forest systems. Details on parameterization are given in Katzensteiner and Eckmüllner (2010) and Duncker (2010b).

3. Results and discussion

The water balance of vegetated surfaces depends on meteorological parameters such as precipitation, radiation, air temperature, humidity and wind speed; soil parameters such as plant available water storage capacity; as well as a number of crop related factors such as height, roughness, ground cover, reflection and resistance to transpiration (Allan *et al.*, 1998).

3.1. Choice of tree species

3.1.1. Evaporation

3.1.1.1. Intercepted rain evaporation

The evaporation of precipitation intercepted by a 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 the evaporation of intercepted rain between tree species are well studied, in particular for spruce and beech (Brechtel, 1970; Benecke, 1984) and their relevance concerning interception, water balance and microclimate has been proven in many studies (Augusto *et al.*, 2002). The Solling study showed comparatively high evaporation of intercepted rain for spruce (29%) compared to beech (17%) (Benecke, 1984; Ellenberg *et al.*, 1986). The higher interception rate of Norway spruce can be attributed to greater leaf area, both in summer time and particularly in winter. The study of (Kantor, 1981, 1984, 1995) also showed greater interception losses in mature coniferous stands (16%) compared to broadleaved forests (7 %), but these values are much lower than the Solling results. The difference may be explained by differences in precipitation (higher precipitation rates on Kantor's sites). Of course there is a pronounced year to year variation depending on climatic conditions (Table 1).

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

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, guide a high share of precipitation as stemflow close to the trunk (up to 20% of the precipitation), conifers like spruce rather guide precipitation to the periphery of the crown (Beier, 1998; Peck, 2004). For spruce rarely more than 2% of precipitation reaches the forest floor as stemflow, except in very young stands. Pine shows stemflow values of approximately 3% and Douglas fir even up to 8%.

3.1.1.2. Soil evaporation

Soil evaporation in mature stands is usually of less importance than interception losses. In many studies intercepted rain evaporation and transpiration by forest floor vegetation is included in this term, therefore results of different studies are frequently not comparable. In the studies of Kantor (1984; Kantor, 1995) soil evaporation amounts to 6% for mature spruce and beech stands. During clearcut phases and early stages of stand development, however, this term will be pronounced. In a study by Katzensteiner (2000) soil evaporation during vegetation period at a clearcut 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 2).

3.1.2. Transpiration

Contrary to interception, plants are able to regulate transpiration in response to climate conditions and soil water availability. The main species specific property influencing the transpiration process is

canopy conductance, a stand property depending upon stomatal resistance (Körner *et al.*, 1979), leaf area and canopy structure (Schäfer *et al.*, 2000; Köstner, 2001).

Increasing transpiration can be observed over time in regenerating forest stands, as leaf area and atmospheric coupling increase. Transpiration of replanted beech and spruce stands at the same site investigated by Kantor (1995) showed generally higher rates in the spruce stand in comparison to the beech stand, with the difference increasing over time (Figure 1). In mature stands however, differences in the transpiration rates are negligible (Table 1). Contrary conclusions were drawn by (Hietz *et al.*, 2000). A considerably higher transpiration rate in beech trees than spruce trees of comparable DBH was measured. It was assumed that coniferous trees were more economic in the consumption of water than broad-leaved trees. In a review of several studies Schipka *et al.* (2005) reported high values of canopy transpiration, approximately 210 – 350 mm, for beech in Central Europe showing no clear dependence on precipitation regime. Experimental results were obtained by various methods including sap flux measurements, soil water balance and micrometeorological approaches.

Coniferous and deciduous trees may further differ in rooting depth (Canadell *et al.*, 1996), which strongly determines 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 in 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 independent of stand age, canopy transpiration/LAI decreased with increasing stand age in Norway spruce (Köstner, 2001). Ryan *et al.* (2000) measured greater transpiration for 12 m tall, compared to 36 m tall, ponderosa pine trees – another hint at physiological changes in leaf conductance with tree age.

3.1.3 Total water use and water yield

For European conditions a greater water yield can be assumed for broadleaved forests than for coniferous forests. This difference has been shown for spruce vs. beech (see preceding chapters) or spruce vs. oak (van der Salm *et al.*, 2006; Rosenqvist *et al.*, 2009). In pine stands, high evaporation rates from ground vegetation and the forest floor may lead to a low water yield.

Care has to be taken with *Eucalyptus* replacing oak woodlands in Southwestern Europe. Even if water use efficiency is high for this genus (Stape *et al.*, 2004; Whitehead and Beadle, 2004; Forrester *et al.*, 2009), it has a fast growth reaction and thus increasing water consumption with increasing availability of water (Stape *et al.*, 2004).

3.2. Site preparation – fertilization

In general, site preparation will restrict encroachment by clearcutting vegetation, reduce transpiration rates, and until canopy closure increase soil evaporation. Due to reduced competition and improved soil conditions, growth of forest saplings will be accelerated. Thus, net effects on water yield may be of minor importance.

Fertilization may influence ET by altering leaf area and by changing stomatal behaviour. Additionally, indirect effects on water uptake by changing the shoot to root ratio, rooting pattern and conductivity of the xylem have been shown (Ewers *et al.*, 2001). The authors compared effects of fertilization and irrigation on stomatal conductance of young *Pinus taeda* and *Picea abies* stands grown on nutrient poor soil. Fertilization doubled and tripled leaf area of *P. taeda* and *P. abies* respectively. In fertilized, non-irrigated *P. taeda*, canopy stomatal conductance decreased as a response to increasing leaf area index, while the combination of irrigation and fertilization caused a large increase in stomatal conductance. In *P. abies* fertilization caused no change in stomatal conductance unless irrigation was applied. Earlier reports by Guehl *et al.* (1995) gave evidence of reduced stomatal conductance after fertilization of *Pinus pinaster*. Ripulline *et al.* (2004) confirmed an increase of WUE after N fertilization 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 fertilization on WUE.

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

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

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

3.3 Stand management and harvesting

3.3.1. Harvesting

3.3.1.1. Final harvest

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. After forest harvesting more precipitation reaches 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. Greater surface runoff rates due to soil compaction have been shown in a number of studies (e.g. Croke *et al.* (1999)).

In snow dominated areas, intensity and timing of runoff will additionally be influenced by dynamics of the snow cover. Jost *et al.* (2007) demonstrated that for such watersheds, forests accumulate between 27 and 39 % less snow than clearcuts, consequently altering runoff during snowmelt periods (Jost *et al.*, 2009).

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 elimination of transpiration cause an increase in water yield for the first few years until the clearcut 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 harvesting 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 less than 20% of a catchment area or forest cover cannot be determined by streamflow measurements. Reviewing 94 catchment experiments, Bosch and Hewlett (1982) assumed a general 40 mm increase in annual yield for every 10% reduction in conifer forest or eucalyptus forest cover and 25 cm for deciduous forests. The study by 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 – a realistic scenario at the scale of forest enterprises.

The increase in streamflow following deforestation is mainly due to decreased interception by the crowns, reducing the evaporative losses. A decrease in canopy interception associated with greater infiltration and increased water yield might be the result.

Hornbeck *et al.* (1993) generalised that the effect of forest harvesting was an initial increase in water yield, which occurs promptly after the cut. The magnitude of increase is related to the percentage of reduction in basal area, as found in previous summaries as well (Douglass and Swank, 1972; Bosch and Hewlett, 1982). 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 can usually 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 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 2). The slope of the curve for partial regression for water yield increase, as a percentage of area cut, is similar to earlier analyses and hardly differs between deciduous 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 evaporation of intercepted rain for conifers.

Forest removal increases the water yield, the subsequent return towards and below pre-treatment streamflow is associated with forest growth rate, canopy cover, and soil depth (Cornish, 1993). Under natural regrowth, increases in water yield 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 of the streamflow components decrease (Bari *et al.*, 1996). The increases in 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, harvesting trees causes an increase in water that is 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 Kirsch (2000) just shows an observable impact on average flow and peak flow in basins smaller than 100 km². The role of catchment size on riparian and hillslope dynamics was investigated by McGlynn *et al.* (2004). They found no clear relationship between catchment size and new contributions of water to runoff during 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 delay of the hydrograph peaks in a 15-year old conifer forest in comparison 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 4).

An early study by Harr and McCorison (1979) measuring the size and timing of peak flows in Western Oregon showed, however, that clearcut logging reduced the size of an annual peak flow by 32%. A delay of all peak flows by nearly 9 hours could be observed after clearcutting. These results were attributed to the site specific snowmelt patterns. In contrast to those findings Robinson and Dupeyrat (2005) found no evidence that 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 in Wyoming, USA, did not cause a significant increase in peak discharge (Troendle *et al.*, 2001).

3.3.1.2 Thinning

The removal of intercepting surfaces through thinning results in decreased evaporation of intercepted rain and consequently a higher percentage of precipitation reaches the forest soil. Depending on the degree of LAI reduction and canopy opening by the thinning operation, modification of climatic characteristics occurs to a greater or lesser extent. After thinning of a pine stand and under-planting with deciduous trees, Knoche (2005) observed a decrease in interception by 8% of annual precipitation (729 mm) to 30% as a consequence of the strong reduction of stand basal area. On a watershed scale, interception loss of a fully stocked stand in a beech-spruce-fir forest in the Bavarian Alps 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 3.

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 % of 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 increased exposure to evaporative forces. Light, temperature and wind are increased, in comparison to shaded areas. Molchanov (1960) reported an evaporation increase of 0.8 % as a result of 15-25 % thinning. In the total water balance, this effect may be of minor importance.

Transpiration

Both increases and decreases in transpiration have been reported after thinning. 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).

Lower transpiration rates of a *Quercus robur* stand after thinning were measured by Vincke *et al.* (2005) Though the transpiration of individual trees may increase due to thinning, as 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. Greater physiological activity of thinned stands in the long-term was also shown by Peterson *et al.* (1997).

Water yield

The removal of intercepting surfaces of the forest canopy directly affects the generation of runoff 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 canopy closure. A crown cover reduction from 60 to 14% in Australia resulted in an increase in streamflow of approximately 20% of annual rainfall after 3 years, compared to a streamflow yield of 6% of 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).

Peak runoff

Snow accumulation and melt have a significant influence on hydrological processes in high latitudes, such as montane and boreal forest environments. Peak flows are affected by removing canopies as a consequence of altered snow accumulation and melt. Canopy removal in a coniferous catchment in Colorado, USA increased precipitation reaching the forest floor by about 40%, the peak snowpack water equivalent increased by 35% (Stottleyer and Troendle, 1999). A deeper snowpack accumulates after removal of forest canopies, which leads to an altered rate of spring snowmelt and a shift in the timing of peak water yield.

3.3.2. Rotation and silvicultural regime

To visualize the effect of tree species choice and thinning regime over a rotation, different scenarios have been calculated using the BROOK90 model (Figure 5).

The modeling results indicate a distinct difference between beech and spruce over the length of rotation. 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 recovers quickly after thinning. Under conditions 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 the EIT of spruce (except for the heavily thinned variant) – a result of lower interception rates (Table 3).

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 continuously reduces water consumption. For a whole rotation in this case, water consumption between early and heavily thinned stands with a short rotation period 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 during different seasons (Figure 6). The greater water consumption of Norway spruce in comparison to beech agrees with values from the literature (Table 4).

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

3.4 Land use changes

3.4.1. Water use and groundwater recharge

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) indicated that interception losses from forests are greater than from shorter crops, primarily because of increased atmospheric transport of water vapour from their aerodynamically rough surfaces.

Van der Salm *et al.* (2006) controversially discusses the effect of afforestation of arable land. The issues 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 raised. Recent studies regarding chronosequences of afforestations in the Netherlands (van der Salm *et al.*, 2006) and in Scandinavia (Rosenqvist *et al.*, 2009) 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 on the arable land to 172 mm in the 18 year old oak and to 100 mm in 13-14 year old spruce stands. A comparison of the 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. 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.

3.4.2 The role of watershed management with respect to peak runoff

Compared to other land use types, forest areas show considerably lower runoff coefficients (Table 5). This is of particular importance for the question of torrent and stormflow prevention. Surface runoff under various scenarios of management in the 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.

A modelling study of the upper Schesa catchment (0.781km²), located near Bludenz in Vorarlberg, demonstrates peak-flow differences under 4 land-cover variations (Table 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 partial forest closure, Scenario 4 is completely forested with trees and shrubs (Markart *et al.*, 2006). It was assumed the rougher the surface of a site was, the proportion of surface runoff is lower.

3.5. Road construction

After clearcutting and 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 quite a different hydrologic response than clear-cutting alone, leading to significant increases of various magnitude in peak discharges in all seasons (Jones and Grant, 1996).

The results support the hypothesis that roads positively interact 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 only generate runoff 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). Effects may not be generalized and high variability of responses may be expected (King and Tennyson, 1984).

4. Synthesis

Forests, compared to other land use types are characterized by considerably higher water consumption. Thus, under conditions of limited precipitation, afforestation may reduce water yield compared to other land use options. In humid climates the effect of forest management upon water yield may be negligible. 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 % of a watershed will increase runoff rates in a proportional relationship to precipitation amount and cleared area. Depending on site conditions and regrowth rates, the impact may last from a few years if the forest regenerates quickly, to decades if the site is heavily disturbed.

In addition the construction of roads, skid trails and landings, soil compaction from harvesting operations may also 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 during 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 (Jost *et al.*, 2004).

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

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

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Annexes :

Table 1: Water balance of a spruce and a beech stand in the Orlicke Mountains (Cz.) in the hydrological years 1976/1977 – 1980/1981 (Kantor, 1995)

Table 2: Water consumption of pine and beech stands in the NE German lowlands during vegetation period [mm] (Müller *et al.*, 2002)

Table 3: Scenario results of EIT and water yield for a spruce and a beech stand for low, average and high precipitation rates

Table 4: Water cycle of European forest ecosystems: Comparison of model results for generic systems of European beech and Norway spruce and literature values

Table 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)

Table 6: Peak runoff ($\text{m}^3 \text{s}^{-1}$) for four different scenarios in the Schesa catchment (Markart *et al.*, 2007)

Table 1: Water balance of a spruce and a beech stand in the Orlicke Mountains (Cz.) in the hydrological years 1976/1977 – 1980/1981 (Kantor, 1995)

Hydrological year	Precipitation of an open area	Interception		Transpiration		Soil Evaporation		Surface runoff		Horizontal soil runoff		Seepage		Δ 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	-0.2
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	1.3
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	-0.4
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	0.4
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	0.3
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	-0.2
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	0.4
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	0.1
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	0.2
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
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

Table 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		104	148	126	378
Pine/beech 51/11a	360	83	220	72	375
Beech 101 a		86	256	44	386

Table 3: Scenario results of EIT and water yield for a spruce and a beech stand for low, average and high precipitation rates

Scenario	Minimum			Average			Maximum		
	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

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

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

Table 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)

Land Use	B			C		
	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

Table 6: Peak runoff ($\text{m}^3 \text{s}^{-1}$) for four different scenarios in the Schesa catchment (Markart *et al.*, 2007)

peak runoff ($\text{m}^3 \text{s}^{-1}$)	Scenario			
	1	2	3	4
	8.1	8.6	6.2	4.2

Captions of figures

Figure 1: Transpiration rates in regrowing beech and spruce stands (Kantor, 1995)

Figure 2: 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 or coniferous forests (a) deciduous hardwoods (b) respectively. Response is visualized for four MAP levels.

Figure 3: Relation of intercepted rain evaporation and stem number after pre-commercial thinning of Norway spruce (Hager, 1988)

Figure 4: Storm unit hydrographs for drained coniferous forest (after Robinson *et al.* (2003))

Figure 5: 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 6: BROOK90 model results of the water cycle for a spruce stand (a) and a beech stand (b) with LAI 6.

Figure 1

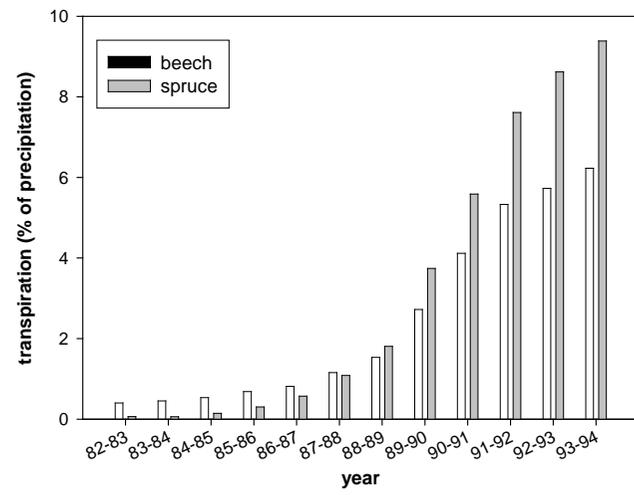


Figure 2 a

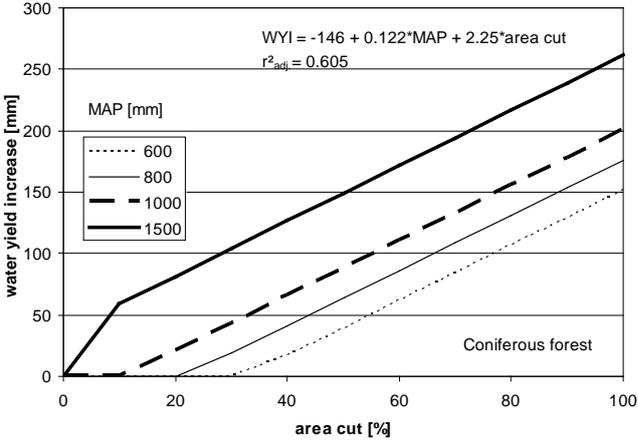


Figure 2 b

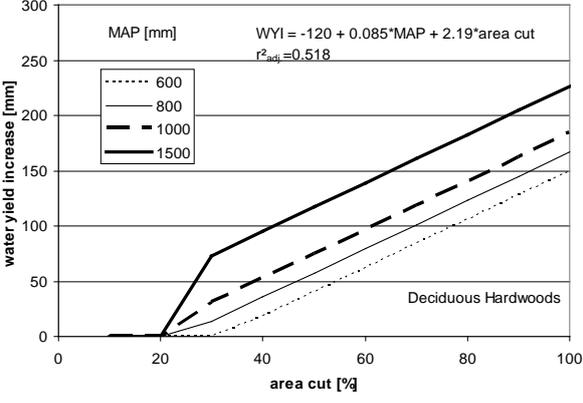


Figure 3:

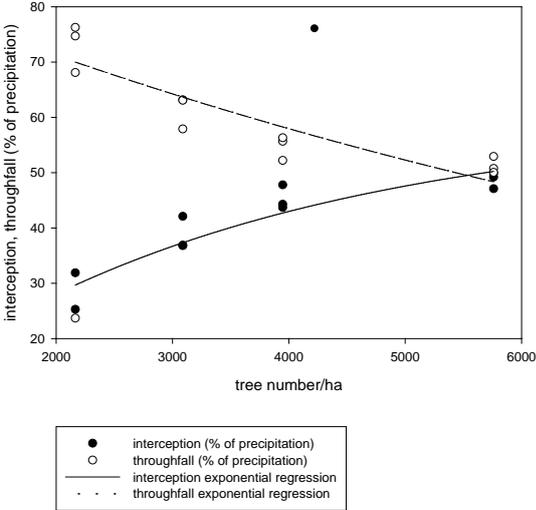


Figure 4:

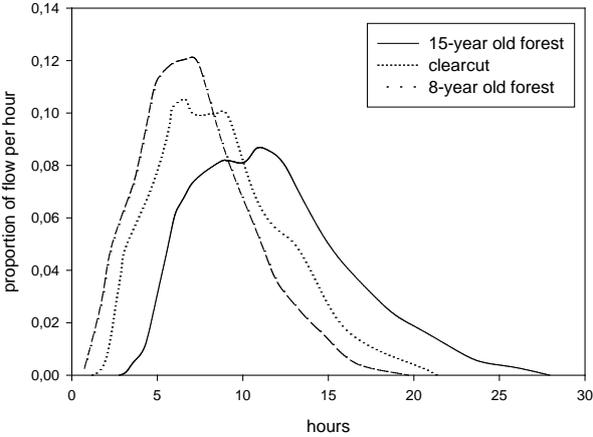


Figure 5:

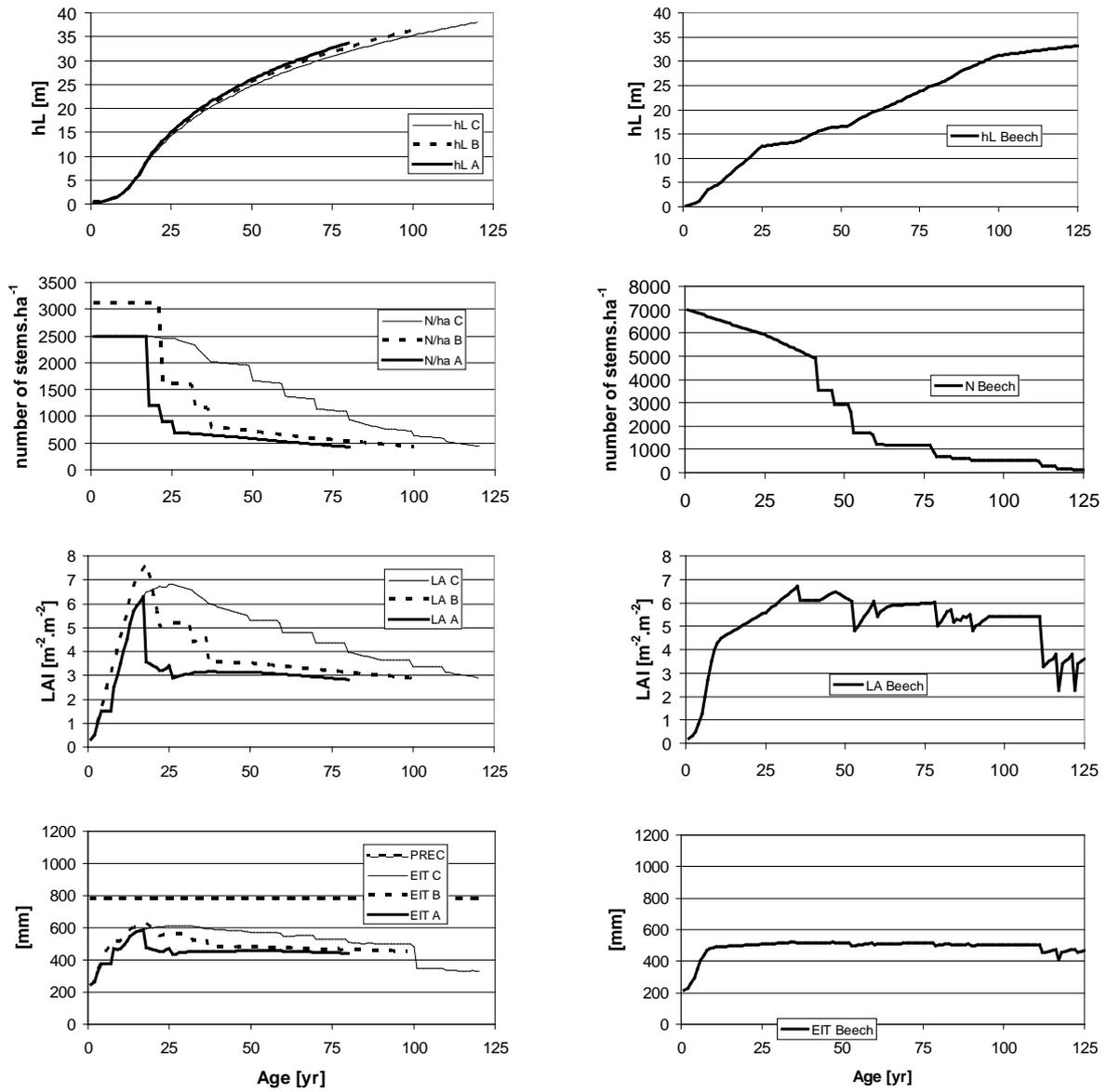


Figure 6a:

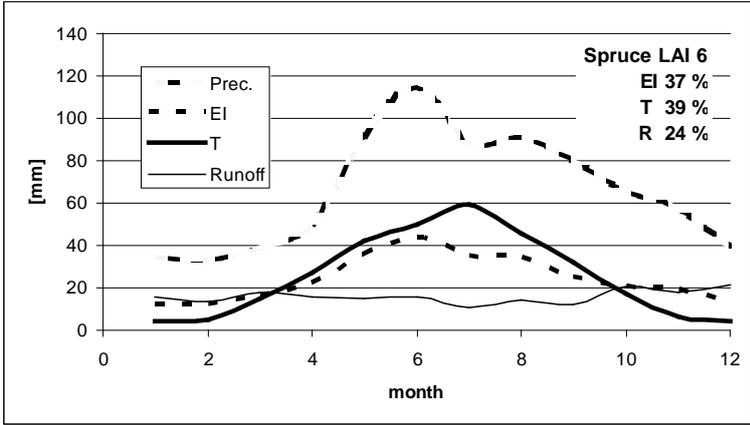
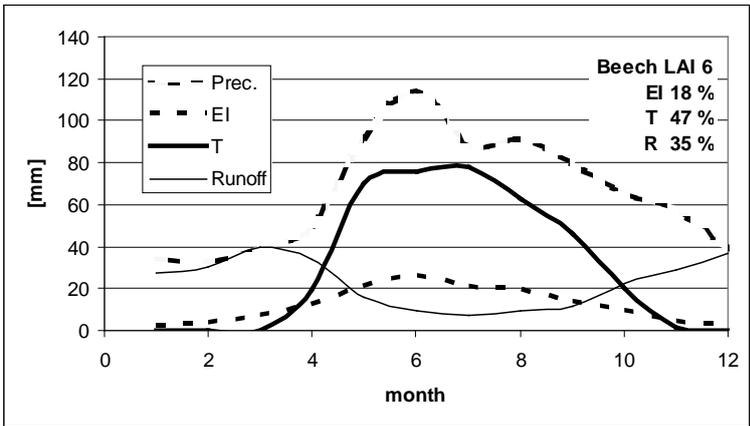


Figure 6b:



3 The impact of forest management on soil quality

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Abstract – The soil quality (SQ) on forest land is a prerequisite for services which are considered beneficial to society such as carbon sequestration, biodiversity, wood production and water for drinking water production. It is essential to manage the forests in a sustainable way to secure the SQ in a long-term perspective. In this paper, we identify and propose methods to quantify the impact of forest management on SQ in the temperate and boreal region. We have looked at several indicators to assess the impact of forest management on SQ, such as nutrient pools and fluxes (input-output budgets), pH, bulk density, porosity, soil formation rate and sediment yield as well as total carbon (C). Our analysis is based on meta analyses, reviews and scientific reports. We 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. Especially, input-output budgets were appropriate indicators for SQ in relation to the impact of harvesting operations. A 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. Today's modern intensive forestry includes heavy machine trafficking with great influence on SQ. A macropore volume <10% has been observed to restrict root growth. 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 it varies widely even within site. Many monitoring networks are available worldwide which can be used in future work on the establishment of further critical indicator values. This work will increase our understanding of how human disturbances impacts SQ and how to react to such changes in the forest ecosystem, ultimately leading to more sustainable use of the forest soil resource.

Resumé - missing

soil quality / forest management / indicators / available nutrient stocks / soil acidity / compaction / erosion

1. Soil quality (SQ) and forest societal services

The concept of soil quality (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).

As defined, the concept of SQ is both multidimensional and relative. The understanding of what is good SQ depends on the type of ecosystem and the purpose(s) for which it is managed (Patzel et al. 2000). In principle, SQ might be considered good for one purpose, but poor when assessed for another purpose. In practice, assessments of good SQ for one purpose will often coincide or overlap with assessments of good SQ for other purposes.

Within the limitations set by climatic conditions and parent material, the soil provides nutrients and water to support the growth and health of the vegetation, and it acts as an anchor for roots preventing the soil from eroding. Forest soils thus support the production of a variety of services that are

beneficial to society. The most obvious is wood production. However, environmental services as clean and abundant ground water for drinking purposes, clean streams and lakes, species and gene conservation and carbon sequestration are increasingly recognised and valued by society. Good SQ is a prerequisite for the provision of all such services and special attention should be given to the question of good SQ when a forest is managed for a multiple of purposes since trade-offs with regard to good SQ potentially exist.

The demand for forest products, mainly timber and pulp wood, has increased, and this has led to an intensified forest management with more mechanised operations, shorter rotations, draining, soil scarification, fertilisation, weed control, selection of more productive species, and breeding for faster growth (e.g. Moffat 2003; Johnston and Crossley 2002). These practices affect SQ. Soil erosion, nutrient removal, soil compaction and changes in soil organic matter (SOM) and soil water status are identified as the most important affected features (Haines et al. 1975; 215,230, Johnston and Crossley 2002] when we describe SQ.

Maintaining SQ is the key to maintaining healthy and productive forests. Negative impacts are naturally counteracted by soil formation and nutrient input from soil weathering and deposition. A fundamental requirement for sustainable management is that the impacts of forest management practices should not exceed the long-term site recovery [230]. Therefore, also SQ must be retained in the long-term [166, 156]. For example, erosion losses should not exceed soil formation rates and nutrient removals should not exceed nutrient inputs by weathering, N fixation and atmospheric deposition [230]. If the long-term self restoration capacity is exceeded, the impacts will cause soil degradation and lead to a reduction in SQ [230] and a permanent impairment of the productivity of the land (Powers 2006).

Soils differ in vulnerability, resilience and recovery time, and the degree to which forest SQ can be restored following degradation depends on the reversibility of individual impacts. Additionally, impacts need to be aggregated over at least one tree rotation before significant statements regarding for example nutrient sustainability can be made [230]. Recovery of the soil resource is possible to a smaller or larger degree, either by introducing less intensive management and letting the recovery processes act over long periods with no intervention or by improved treatments [230].

If intensively managed forests should continue to provide a wide range of societal services, forest management-induced changes in SQ should be detected before the long-term self restoration capacity of the soil is exceeded. This obligation is fundamental in international principles for sustainable forest management as outlined in the Montreal and Helsinki processes [186, Moffat 2003]. To fulfil these obligations, it should be clarified which management operations cause the largest or the most vital changes in SQ, and how these changes can effectively be detected and monitored. The need for quantitative tools and methods to assess and monitor SQ is apparent [7, 53] and flexible goals for SQ need to be established (Patzel et al. 2000) to allow for adaptation to different ecosystems and management strategies. These questions are addressed in this paper by:

- i. identifying the most important indicators as a tool to estimate SQ in forests;
- ii. identifying and describing the most important impacts of forest management strategies on forest soil processes;
- iii. analysing which SQ indicators are suitable in relation to different management purposes;
- iv. comparing management-induced changes in SQ quantitatively (whenever possible); otherwise comparing the nature and scale of the changes in qualitative terms;
- v. giving an overview of on-going SQ monitoring, and discussing how monitoring of management-induced changes in SQ can be improved for better quantitative prediction of forest management impacts on SQ under various conditions.

We focus on temperate and boreal forests, and whenever possible on a long-term time perspective, i.e. one rotation period or approximately 50-120 years. Our analyses are based on existing data syntheses and reviews supplied with newer scientific research.

3. Indicators of SQ

The use of indicators to assess the impact on SQ has been proposed for developing a sensitive and dynamic system to document the SQ and how it responds to management [186]. Indicators of SQ must be sensitive to changes in soil function (Moffat 2003; 186?) and they should be defined as quantifiable physical, chemical and biological parameters and processes. The indicators should be related to a defined level or rate to determine whether SQ is improved or degraded [186?]. Johnson and Todd (1998) [94] furthermore 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 a part of existing databases [7]. For our purpose, soil indicators responsive to forest management are especially of interest.

In literature, a list of indicators on SQ has been proposed through the last 15 years (e.g. Karlen et al. 1997; [186]). Among these indicators, we chose to work with SQ indicators connected to a forest-production perspective (Table 1).

2.1. Reduced nutrient input and availability

In the long-term perspective, wood production capacity is influenced by the inputs and outputs of nutrients. Hence, input-output nutrient budgets may provide a picture of the direction of change in SQ and therefore seem to be adequate indicators of SQ. Here we present two approaches of calculating such budgets; 1) the soil nutrient store method, 2) the nutrient balance method.

In method 1, the soil nutrient store is determined by normal soil sampling on two occasions to assess potential changes. In method 2, inputs to and outputs from the system are accumulated over a time period to calculate a balance for individual elements. Inputs are wet and dry deposition, weathering of soil minerals, N-fixation and fertilisation. However, the weathering rate is difficult to determine [174]. Outputs from the system take place through harvesting, leaching, erosion and vaporisation. If the budget is balanced, nutrient stores are neither increased nor decreased. A positive budget means that the considered element is accumulated in the system while a negative budget suggests that export exceeds import and as such the studied nutrient is depleted from the system causing the wood production capacity to be unsustainable over time. A thorough description of the method and the calculations involved is given by Ranger and Turpault (1999). The use of method 2 as a diagnostic tool for sustainable forest management is well described and used, e.g. by [172], Ranger and Turpault (1999) and [176].

The time interval in methods 1 and 2 must cover all relevant forestry operations as well as the entire stand development, since the distribution of nutrients within the forest alters gradually as the stand develops [133, 146]. Thus, for forests which are largely used for wood production a time interval of preferably one rotation period or more is appropriate.

2.2. Soil acidification

Soils become acid when the acid production rate exceeds the acid neutralisation rate. The most important acidifying processes in nearly all forest soils in humid climates are i) biomass growth and subsequent harvesting, ii) nitrification and subsequent leaching of nitrate and base cations, iii) production and leaching of organic acids during decomposition, and iv) deposition of N and S and subsequent leaching of nitrate and sulphate together with base cations. Soil acidification is counteracted by weathering [211, 174]. The intensity of the acidity can be measured by the pH.

A neutral pH indicates a sufficiently high weathering rate to compensate ongoing production of acidity in the system. At low pH, acidity produced or deposited from the atmosphere is accumulated in the soil indicating an insufficient acid neutralisation by chemical weathering. Soil acidification is, however, a natural process, e.g., the podzols in the boreal forests are naturally acidic and have low buffering capacities. The rate of soil acidification can be accelerated by human activities. Forest

management operations like harvesting and choice of tree species significantly accelerate natural increases in soil acidity due to growth.

The pH is assessed at two occasions to assess potential changes in soil acidification. As for the nutrient budget determination above, a time interval of preferably one rotation period or more is appropriate.

2.3. Compaction

Soil compaction takes place as a consequence of off-road driving in the forest [59B], for instance in conjunction with harvesting, and on forest roads and skid trails. The pressure this exerts on the soil may lead to increased bulk density and loss of soil porosity. The root environment will deteriorate and roots will have difficulties of extending during dry summers and during wet winters because of lack of oxygen [70, 224, 47, 218, 58] leading to lower production rates. An estimate of the compaction degree can thus be made from measurement of bulk density and soil porosity. A macropore volume <10% has been observed to restrict root growth [76, 171, 193]. 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 varies widely [193].

The soil porosity and bulk density are measured at two occasions to assess potential changes in compaction. Preferably the measurements should be performed before and after harvesting or other forest operation that might affect compaction considerably. The recovery time could be estimated by follow-up measurements at different times after the forest operation.

2.4. Soil erosion

Erosion is often a local phenomenon and a natural process where wind and water work together moving soil fragments from one place to another. The rate of soil loss (erosion) forms equilibrium with the rate of soil formation. The soil formation begins with the weathering of rock by physical or chemical processes and is a slow process. When soil loss is higher than soil formation, erosion is taking place.

Soil loss will often begin with splash erosion where rain drops hit the soil which creates a net downslope movement of particles. A flow of water and soil particles will move over land surfaces and the greater the flow velocity the more soil loss. Steep slopes will cause a faster flow. Vegetation slows down flow and hereby reduces the soil loss. Channelized flow into rills, gullies and canyons will further erode material downward through the soil. Soils mostly affected by erosion include sandy and loamy soils on steep slopes in upland sites and in high rainfall areas [41] where bare soil is most vulnerable.

Erosion processes can become accelerated when humans interfere. Forest operations like site preparation, harvesting, soil damages near surface water due to off-road driving and construction of skid trails and forest roads speed up erosion [230] so that soil loss is faster than soil formation. Soil horizon disappears and the eroded material sediments in lakes and streams and clogs them. Soil loss and soil formation before and after management operations will tell more about the amount of erosion. However, these processes are difficult to determine and can only be estimated inaccurately.

Theoretically, the rate of soil formation sets the upper limit for an acceptable rate of soil loss by erosion. Knowledge of soil's age together with information about changes in climatic and biotic factors enables rates of soil formation to be estimated in tons per hectare per year. The estimation of soil formation then needs to be referenced to a former state of the art, e.g. before and after a forest operation. Different equations for soil loss have been developed like the Universal Soil Loss Equation (USLE) by Wischmeier and Smith (1965) or the Revised Universal Soil Loss Equation (RUSLE) by Renard et al. (1997).

4. Impact of forest management on SQ indicators

3.1. Regeneration and forest stand establishment

Forest operations that are performed in order to ensure successful regeneration and establishment of the new forest include, e.g. soil scarification carried out before stand regeneration, the choice of appropriate tree species, artificial drainage, prescribed burning and weed control in order to create a favourable environment that promotes fast and efficient establishment and good survival of seedlings.

Soil scarification

During heavy storms windthrow can cause large physical disturbances of soil profiles [162, 230]; however, most disturbances on forest land are manmade. Different types of mechanical scarification methods such as patch scarification, disc trenching, and ploughing have been introduced mainly to improve survival of seedlings during forest regeneration. The use of scarification almost 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. However, ploughing causes considerable disturbance to the soil profile since horizons are mixed and turned up and down in this way disrupting the pedogenic processes [230]. On arable land, ploughing has been performed regularly. [229] ranked different soil preparation techniques according to the volume of soil disturbed: ploughing ($350\text{-}850\text{ m}^3\text{ ha}^{-1}$) > mounding ($170\text{-}340\text{ m}^3\text{ ha}^{-1}$) > scarification using a disc trencher ($110\text{-}280\text{ m}^3\text{ ha}^{-1}$).

The amount of vegetation cover after soil scarification indicates the degree of soil disturbance. [127] 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%. [140] 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.

Other studies in Finland showed that scarification affected the vegetation cover for a long time after treatment. After five years after scarification the biomass and the nutrient pool of the vegetation was significantly higher in plough ridges than in plough furrows [156B]. In a Swedish study on the effect of soil scarification, the authors found a 16-19% lower nitrogen pool in the treated soils than in untreated control soils [233] 60 years after treatment. In another study by [37], scarification also caused significant losses in nutrient capital which was thought to impact negatively on future forest productivity.

Although soil preparation in the regeneration phase significantly improves growth and survival of seedlings a significant increase in leaching of nutrients may take place [81, 177B]. [31] also noted a compaction (see later) of the mineral soil as a consequence of scarification after two years.

Ploughing may cause erosion to increase since mineral soil gets exposed to wind and rain [230] and it is reasonable to recommend the use of lower impact physical manipulations on susceptible sites in order to reduce erosion.

Choice of tree species

Individual tree species vary in their soil-forming impact. 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 [132]. 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. Lately, [14] made a thorough 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 [157, 54, 16] as well as topographic position and the distance to the forest edge [54]. Several studies have shown a much larger throughfall deposition to coniferous stands than to deciduous stands [210, 157, 54, 179, 114, 178].

Nutrients that are not taken up can be leached from the soil. Studies comparing the output via water seepage below pure stands with different tree species observed 2-4 times higher output of nutrients from Norway spruce than from beech stands [117, 17, 64, 179].

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 [206]. The annual symbiotic N-fixation can be substantial and it was estimated to be between 50 and 200 kg N ha⁻¹ [19, 21, 25] 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 and where N deposition is large.

Only a few studies have compared the effect of different tree species on weathering rates and shown that Norway spruce promote weathering of soil minerals and has a weathering rate which was 2-3 times higher than under species like beech, oak and birch [117, 17, 64, 14]. The mineral weathering rate is mainly influenced by soil pH and soil concentrations of dissolved organic carbon (DOC) [55, 174] and such studies showed even 2-3 times more DOC under Norway spruce than under beech and oak [174, 12].

Different tree species have different effect on pH in soil. The difference in pH below different tree species could be as much as 1 pH unit but most often it was between 0.2-0.4 units [14]. A national Swedish review of the effect of tree species on pH showed a tendency to 0.5 units higher pH for beech or birch stands compared to spruce stands (Ring et al., 2008). A specific Danish study by [159] 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, [14] 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 [23, 32]. Differences in litterfall quantity are generally smaller (between 3.5 and 4.0 t ha⁻¹ yr⁻¹, reviewed by [14]) than differences in litter quality (nutrient concentrations, ratios among nutrients and specific components of varying recalcitrance) [20]. Foliage of deciduous tree species is generally richer in nutrients with higher concentrations of N, potassium (K), calcium (Ca) and magnesium (Mg) than coniferous tree species [14]. 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 [14].

Nutrients stored in SOM become available for the trees again through decomposition, exchange reactions and dissolution. Soil communities thus exert strong influence on the processing of organic matter and nutrients. Tree species have a strong effect on the composition of soil microbial communities. A study by [125] 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 [20]). Accumulation of nutrients in the forest floor also significantly differs among tree species indicating differences in the biodegradability of the litter [102, 213]. The decomposition activity of the micro flora and ultimately the turnover of nutrients are connected to the initial litter quality, light transmittance, air temperature, moisture and soil fauna. 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 deteriorate the site while beech is characterised as producing plenty of raw humus [23]. 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 [17, 64, 14]. 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 [14].

Artificial 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 artificial drainage have often increased substantially due to improved conditions for root growth and increased mineralisation of decomposing peat [225]. On peatland, artificial drainage therefore improves aeration and is often a pre-requisite for trees to be able to use the available stock of nutrients and SQ. However, often the increase in mineralisation rates may cause a surplus of available nutrients with significant leaching to ground water or water streams as a consequence [40, 80, 169, 225]. This may be problematic for water quality [81] and biodiversity [41]. The positive effect on growth may be temporary and a continuous effect may depend on ditch cleaning or secondary ditching [225, 232]. 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, [225] could, however, not find any signs of nutrient deficiency up to 75 years 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 effective and rooting depth increase permanently, the nutrient stock available for plant uptake will increase. However, we have found no evidence in scientific literature for this statement.

Drainage may furthermore cause erosion to increase [230]. Forest management operations that cause exposure of bare soil and provide obvious drainage channels will contribute most to erosion [230].

Prescribed burning

Prescribed burning can be an effective management tool in order to remove undesirable vegetation and slash [142] and renew the forest [185]. 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. Prescribed fires in forest stands are most common in the United States, but earlier on it was used for soil scarification in Scandinavia. Prescribed fires reduce much of the organic material to mineral-rich ash, thus, lowering the risk of wildfire and reducing woody competition [212, 42], releasing and recycling nutrients, creating openings where new forests can establish, controlling species composition and competing vegetation as well as reducing insect and disease infestations [221, 38, 185, 227].

Prescribed burning are generally low intensity and low severity fires performed under controlled conditions when soil moisture content is moderate to high for the purpose of achieving a clearly defined management goal [42]. This kind of fire is used prior to seeding or planting too. Approximately 50% of the available forest floor and understory is consumed in a typical prescribed burning. On the other hand, wildfire is a natural disturbance in some ecosystems which is never intentional and often occurs when it is warm and soil moisture is low. Hereby, wildfires become more severe.

The largest change of forest soils caused by fire is the removal of organic matter [43]. Bulk density increases as a result of the collapse of the organo-mineral aggregates [73, 43] and the clogging of soil pores by ash or freed clay minerals [56], which causes a decrease in the water holding capacity of soil [27, 22]. 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, the pH-value of the humus layer was increased by up to one pH unit after prescribed burning. The first wetting fronts after a fire are of extremely high pH [79]. 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 [142].

Fire can cause substantial losses of N, C and S through volatilisation even at low temperatures [3, 100, 228, 39, 142, 65, 43]. Estimates of N volatilisation loss during combustion of forest floor and other fuels range from ca. 50 to 100% of the N content [49, 61, 62, 121]. A slightly lower N contents in forest floor (31 to 51%) was observed after a prescribed burning [142]. Low-temperature fires may cause little initial loss in mineral soil N [222]. 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 [5, 182, Tiedemann 1987]. The long-term effects of fire are dependent upon fire intensity and time since the fire [93]. Other elements, such as P, Ca and Mg require higher temperatures to volatilise. [142] 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 [81].

Tree seedlings were observed to grow better on burned sites [15]. Wildfire as well as prescribed burning 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.

Viro P.J. (1974) Effects of forest fire on soil. In *Fire and ecosystems*. Edited by Kozlowski T.T. and Ahlgren C.E. Academic, New York. pp 7-45.

Raison R.J. (1979) Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. *Plant Soil* 51, 73-108.

Weed control

Weeds exert a strong competition on trees for water and nutrients in new cultures. Understory control in this period greatly improves tree growth in a number of species [44, 141, 203, 146]. 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 [183]. Weed control includes a number of removal methods: i) mechanical removal by patch scarification, disc trenching and ploughing (increase mineralisation) (see site preparation), ii) chemical removal by the use of herbicides (decrease biological uptake), iii) mulching and lastly iv) competitive weed control.

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. [161] 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, e.g. rye, to take over and suppress the weeds, yet allow the seedlings to get enough light and water to grow. Nursery trees, growing faster than the wanted tree species, may also help to create a faster forest climate and hereby repress weeds and prevent frost.

All weed removal methods disturb the soil to some extent. The most intensive mechanical disturbances increase net N mineralisation, nitrification, and nitrate losses to seepage water [216, 152, 153, 11, Gundersen et al. 2010]. 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 [217]. The mineralisation rates were increased for a relatively short period followed by a longer recovery period in which net mineralisation decreased.

3.2. Nutrient additions in forests

In forestry, nutrients are sometimes added via fertilisers, lime and wood ash to the soil in order to increase the wood production and to compensate and return nutrients taken out when harvesting.

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. The aims of fertilisation are: 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.

Stem-growth increases as a result of fertilisation with N, P and K has been observed [197, 193]. However, [197] furthermore observed that fertiliser additions caused a significant decrease in foliar concentrations of all nutrients except for N. After fertilisation, [15] 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 increase in the concentration of soil N in the A horizon [93]. However, [140] and [161] observed no consistent effect of fertilisation on the soil nutrient availability in White spruce and pine, except for Ca. If N is added alone or in too high amounts, 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 [80]. 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 [201, 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 is 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 eliminated.

Liming

Positive growth effects of liming in agriculture have for more than 100 years inspired to look for the same gain in forestry. Many forest experiments have thus been performed but 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 [51]. In Sweden, growth effects of liming included both positive and negative short-term response [165]. 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 [112].

In principle, liming should improve soil pH, the Ca storage, and the Mg storage if dolomitic lime is used [112]. 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.

Lundström U.S., Bain D.C., Taylor A.F.S. and van Hees P.A.W. (2003) Effects of acidification and its mitigation with lime and wood ash on forest soil processes: a review. *Water, Air, and Soil Pollution: Focus* 3, 5-28.

Löfgren S., Zetterberg T., Larsson P-E, Cory N., Klarqvist M., Kronnäs V., Lång L-O, 2008. Skogsmarkskalkningens effekter på kemin i mark, grundvatten och ytvatten i SKOKAL-områdena 16 år efter behandling. Skogsstyrelsens Rapport nr 16, 123 s.

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 than stemwood [164]. 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 [9].

The major components of wood ash are Ca, K, Mg, silicon (Si), Al, iron (Fe) and P as well as trace elements, some of which are toxic [149, 199, 86, 60, 107, 24]. Ash is generally low in N and S because it is vaporised during combustion. Due to different soil mobility of toxic elements like cadmium (Cd) and caesium (Cs), caution must be taken when wood ash is applied to forests. The chemistry of the wood ash is dependent on the tree species. In general, ash from deciduous tree species contains more K and P and higher proportions of macronutrients but less Ca and Si than ash from coniferous tree species and is therefore likely to be a more effective fertiliser [223, 164].

When wood ash is applied to a soil it will raise the pH of the upper soil. Untreated 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 in the XX layer 10-19 years after application [143, 138, 30, 181]. The transport of ash components down through the profile is however 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 [30, 181]. Hence, an increase in the pH of mineral soils is not usually found [177, 10, 66] except when high doses (>10 t ha⁻¹) have been applied [97].

The content of both K and P in wood ash seemed to be lower than from commercial fertilisers (K: 65-70%; P: 28-70%) [148, 164]. 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 [180, 10]. 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 [35]. 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.

As long as N remains the growth limiting nutrient [204], the addition of other nutrients through wood ash will not increase growth on mineral soils. On the other hand, wood ash addition in forest stands on nutrient rich peat soils has shown a significant positive effect on tree growth [63] and improved conditions for natural stand regeneration [88, 122, 123]. Peat soils deficient in K and P but with a good N status show the highest increase in tree growth [191] while tree growth on peat soils low in N (<1%) remains low [188, 189]. An increase in productivity of 3-4 m³ ha⁻¹ over 55 years was found after wood ash applications of 5 t ha⁻¹ to drained peatland [190, 109].

A future task is to decide for an optimum wood ash application. [6] recommended an application of wood ash to achieve a humus layer base saturation of 50% and a mineral soil base saturation of 20%. 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 seem to grow best with base saturation rates of 30% while hornbeam and sycamore are likely to prefer levels in excess of 50% [164]. A granulated form of ash has been tested 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 [164]. In Sweden, granules are not considered efficient in a larger perspective when also including the stabilization process of ash and its economy.

3.3. Stand harvesting

The last forest operation in a rotation is final felling and harvest which is by far the largest forest operation management in the forest life time changing the ecosystem drastically.

Harvesting

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) [163, 18, Johnston and Crossley 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 [104, 170]. As a result, ammonium is available as a substrate for nitrifying bacteria, and the produced nitrate is subsequently leached below the rooting zone [214, Gundersen et al. 2010] leading to soil acidification and loss of nutrients [90]. The effects are more pronounced when the share of removed biomass is increased as for example in whole tree harvesting (WTH) or when residues are harvested.

A significant amount of nutrients is exported from the ecosystem in harvesting [74, 14, 175] and studies have shown a negative effect of intensive biomass harvesting on productivity (e.g. [91, 197]). Mostly, the negative effect is explained as an induced N deficiency. If the effect continues in the long-term, such forest management does not satisfy criteria and indicators for sustainable forest management [175]. Fertilisation may neutralise the possible negative production effect, at least in the short-term [91]. However, other studies have shown no negative production effect of intensive biomass harvesting (e.g. [168], Nord-Larsen 2002) indicating that soil weathering plus atmospheric deposition have been able to supply the stand with sufficient amounts of nutrients, at least in the short-term.

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. Continuous cover forestry (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 [151]. The effect of such changes on productive and protective forest functions and SQ is, however, still not well studied.

Nutrient pools and fluxes: Compared to natural unmanaged systems, harvesting in managed forests reduces the long-term inputs of organic matter and nutrients to the soil. Significant amounts may already be removed in stem harvesting, while WTH or harvesting of residues e.g. for energy purposes causes further export of nutrients from the forest ecosystem. In an even-aged system, most nutrients are removed in clear-cut compared to individual thinnings. Some studies estimated stand nutrient stocks, corresponding to nutrient removals in clear-cutting (Table 2).

There is a substantial difference in the size of nutrient removals, depending on the size and age [172] of tree species and density of the trees at the time of cutting [46, 160, 74 13], site productivity, harvesting intensity, and nutrient concentration level in the biomass [202]. 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 [71], and thinning regime [198, 195].

Compared to harvesting of stems only, very intensive biomass removal, including also harvesting of stumps 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 [175]. The corresponding increase in removal of biomass was about 2 times (Table 3). 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, where selected model calculations show that the removal of biomass in this case 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, [75] found losses of K twice as high in Sitka spruce for WTH logging as for logging of stems only. 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.

The nutrient balance approach (section 2.1) has been used in several studies to quantify the effect of intensified biomass utilisation on the ecosystem nutrient status [28, 226, 29, 111, 155, 144, 4]. Other studies have shown a biomass export related decrease in soil stores of nutrients, often in Ca [120, 89]. [220, 168] found that concentrations of total soil C, N and P were reduced after harvest of all aboveground living vegetation, and other researchers have likewise found that WTH significantly reduces soil concentrations of N [131] and P [139, 209, 183]. 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) [93, 183].

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. Nitrogen related effects have been demonstrated, but also shortage of P, Ca and K may become a problem in areas with significant N deposition [175]. Magnesium deficiency may occur in some areas just like it was induced by acid rain in Central Europe in the 1980s [184]. A deficit of N may in some areas be alleviated by high atmospheric deposition of N due to pollution, 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 [112]. The use of nutrient balances

and soil nutrient pools as indicators of potential changes in SQ has been further described and exemplified by [175].

Soil acidity: Harvesting significantly influences all the acid producing processes (Section 2.2). In high deposition areas air pollution is considered the most important source of acidity whereas biomass harvesting may be more important when performed at an intensive level [2, 219], especially in low deposition areas. Clear-cutting may increase soil acidity and the loss of base cations as shown by [119, 124, 99, 192]. Using soil acidification as an indicator of long-term changes in SQ after harvesting furthermore requires site specific knowledge about natural fluctuations and of the sites ability to restore itself by weathering.

Bulk density and soil porosity: 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. Heavy machinery might cause compaction of soil [59B] if off-road driving is not performed correctly. This may lead to increased bulk density, loss of soil porosity, increased erosion, and decreased availability of water.

Compaction will affect tree height, diameter and volume growth through a deteriorated root environment where roots will have difficulties of extending during dry summers and during wet winters because of lack of oxygen [70, 224, 47, 218, 58]. [193] found that seedlings planted in compacted soils had greater mortality and slower growth than seedlings planted in uncompacted soil. [76] 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, [183] observed no effect on tree growth after compaction on a range of 62 sites in the United States. Yet another study proved compaction to generally improve both survival and growth on seedlings (Fliming et al. 2006; Brais 2001).

Soil disturbance by harvesting machinery is influenced by factors such as the amount of litter and slash on the forest floor, soil texture, soil moisture, weight of harvest machines along with the weight of the logs, wheel size and pressure, speed, operator skill, operation planning, the use of technical equipment such as mobile bridges etc., terrain and weather [194, 218]. Especially, moist and fine-textured soils are particularly susceptible [135, 1, 76, 134, 218, 96], whereas frozen soils are quite resistant to structural degradation induced by traffic [15].

[8] observed that approximately half of the observed forest area was affected by trafficking. [47, 187] observed no difference in compaction between moderate and severely trafficked areas inside forests and suggested that most compaction will occur during the first few passages of the machinery over the area. On skid trails, the greatest increases in soil bulk density (more than 50%) were indeed associated with the first trips over the ground [1, 69, 77, 187, 218], and even quite low ground pressures (e.g. 35-65 kPa) can result in substantial compaction [68]. Changes in pore size distribution may also occur, with large pores collapsing to form smaller ones [118]. The most severe compaction occurs within the upper 5-10 cm of the soil surface which is where most of the fine-roots are found [1].

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 [72, 220]. Typically, an applied force pushes together the mineral and organic particles effectively filling or compressing many of the existing soil pores [1]. [193] found that macro-pore volumes decreased from 13.5% to 5.7% in the topsoil after compaction. In a study by [205], soil porosity was reduced by 50%. A macropore volume <10% has been observed to restrict root growth [76, 171, 193]. 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 varies widely [193].

Eliasson L. (2005) Effects of forwarder tyre pressure on rut formation and soil compaction. *Silva Fennica* 39, 549-557.

Despite this, most studies of compaction have concentrated on the effect on bulk density. [218] compared bulk density in a forest area before and after harvest and observed no significant difference in pre-harvest and post-harvest soil bulk density. On the other hand, [96, 205, 168, 8] found increases of 16-27% in bulk density as a consequence of trafficking on different soil types down to a depth of 15-20 cm. Several other studies can be quoted showing increase in bulk density after compaction [76, 33, 193, 47]. Furthermore, [103] observed reductions in porosity immediately after felling both in the stand and at the skid trails. [168] 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 $1.40\text{-}1.55 \text{ Mg m}^{-3}$ plant roots have been found incapable of penetrating soils [110].

Compaction furthermore increased the available water holding capacity [6] and reduced soil moisture content by 11% after forest floor removal [205]. 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 [47].

Large machinery sometimes remove organic matter from the forest floor and along with the extra compaction, soil nutrient availability will change [34]. This might influence weathering rates and N mineralisation [231, 218], the latter since organic material might be physically protected against microbial attacks in small pores after compaction [33]. 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. Compaction therefore had a negative effect on N mineralisation, which was reduced by 18%, mostly at higher soil moisture contents. However, [205] found no change in net N mineralisation rate caused by compaction in neither forest floor nor mineral soil, but along with [84] they observed an increase in the net N mineralisation rate after the removal of the forest floor. 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 [205] 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.

Partial harvest methods, such as group selection, thinning or shelterwood harvest have been suggested as more sustainable methods of bole removal than clear-cutting [26, Johnston and Crossley 2002]. Trade-offs 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 [1, 218].

Coarse-textured soils tend to recover faster than fine-textured soils [1]. Recovery has been estimated to take everything from 5-15 years after logging to 40 years or more under skid trails and forest roads [76, 229, 103, 171]. [168] resampled soil 10 years after actual compaction 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.

Soil erosion and sediment transport: Harvesting cause erosion to increase, and as a result, sedimentation in water bodies and surface waters also increases [229]. Soil erosion from forest harvesting is a major environmental concern, especially when harvesting is performed on steep slopes [106, 87]. There is little literature on erosion after logging as well as research comparing the effects of silvicultural treatments on soil erosion. [167] found the erosion after logging in the US to be within the limits of the rate of soil formation while [129] estimated high erosion after logging. [87] 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. 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 [83]. The size of the tree canopy determines the erosive power of precipitation while the other factors protect the soil surface and prevents soil loss.

Residue management

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 (with 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 [45, 130]. 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. [130]). Residue burning will thus result in nutrient export through volatilisation and leaching (e.g. [130, 150]).

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 [50, 36, 200, 196]. 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 [85, 128, 105, 94, 93] as well as in Eucalypts [130]. 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 [45]. On the contrary, [154] found no such impact. [150] 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 [130].

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 2002) (See 3.1 on soil scarification).

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 2 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 [57]. They conclude that the amount of inorganic N probably increases after stump harvesting and probably there is an increased risk for erosion and sediment transport to adjacent surface waters. However, the effects of such removal to SQ are not well documented, and research is needed in this area.

4. Improved evaluations of SQ changes - conclusion and recommendations

We identified harvesting of biomass to cause the largest change in SQ with 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. In order of magnitude, harvesting is therefore the forest operation that exerts the largest change in the SQ indicators. In this case, the input-output nutrient budget was considered the best indicator to characterize the change. It has the advantage of predicting changes in SQ early, even before effects are observed on the vegetation. The information given is indeed interesting to forest managers; however, it is not a simple indicator for forest managers to interpret. The calculation of nutrient budgets is furthermore rather labor-intensive.

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 nutrient balances and acidification might theoretically be counteracted by wise fertilisation and liming but this is seldom done for economic as well as environmental reasons. Modern intensive forestry includes heavy machine trafficking with mainly negative influence on macro-porosity and erosion and large reductions in growth potential are observed hereafter. On steep slopes clear-cut or intensive harvesting might be problematic due to a potential increase in loss of soil material due to erosion.

Specific recommendations on the best SQ indicators for monitoring of SQ changes due to forest management are likely to be evaluated and given on a case to case basis since good indicators seem to differ for different soils. The relationship between a SQ indicator and the production response in a forest stand is difficult to interpret. The concept therefore needs to be further studied and developed.

Monitoring and prediction of changes in SQ is crucial in order to reveal impacts of forest management on SQ. Thus, monitoring is an essential component of environmental management. It provides us with information in order to make educated decisions about how we manage our resources. Regardless of the magnitude and focus of each monitoring programme, they all strive to increase the understanding of how human involvement and environmental interaction impacts SQ, ultimately leading to more sustainable use of the forest soil resource.

Several monitoring programmes on SQ have been undertaken worldwide during the last 3 decades. One of the largest programmes is the Long-Term Soil Productivity (LTSP) programme in the US (Powers 2006) which addresses both short- and long-term consequences of site and soil disturbance on fundamental forest productivity. In order to achieve an overall picture, Winder (2003) examined the international literature for the existence of soil monitoring programmes or environmental monitoring systems, which included monitoring of soil parameters and also included resampling of soil properties. Such repetition of measurements in existing networks is important in order to establish some of the SQ indicators. The review included only English language resources. Primary internet searches were performed between September 2001 and June 2003. Winder found 52 environmental monitoring programmes, the majority of programmes (80%) being situated in European countries. Three United Nations programs and five international/European networks were also found.

A large amount of SQ data is therefore available and efforts should be made to gather these data in common databases. Since resources often are limited and time required to observe measured changes in soil parameters is long, SQ models (i.e. crop growth models and soil degradation models) will be needed in order to generalize data trends and project SQ in the future. Here, monitoring networks obviously can provide data to test and validate the models.

Based on existing data, other steps may be taken to evaluate changes in SQ. Within the agricultural community, the developed SQ concepts and methods have as such been suggested to form the basis for soil-based evaluation of the impact of management operations on the long-term productivity. [101]

demonstrate a model – the Soil Quality Index Model, which integrates the components of SQ into one measure of management impacts on soils. They include five key growth-determining attributes of soils: 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 might be useful for evaluation of SQ changes due to management within forestry as well.

Forest managers need to know how forest operations and other site disturbances may be performed to sustain site productivity and forest protective functions. It is therefore essential to communicate evaluations of SQ changes to forest managers in a suitable way. They need to know if decisions like organic matter removal incl. harvest residue removal, tree species choices, fertilisation, compaction etc. will reduce SQ, if they can mitigate SQ reductions or ameliorate SQ profitably. Therefore, recommended best management practices and possibly other decision support tools should be developed. Forest and soil management recommendations already exist in several countries, e.g. [Johnston and Crossley 2002, 108, 147], but they should be improved and updated with regular intervals as new knowledge on SQ responses to management is created.

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Table 1. An overview of the different impacts of forest management operations on SQ, the kind of change it will cause in the forest ecosystem and the possible indicators which could be used to detect the impacts on SQ.

Impacts on SQ	SQ indicator	Change in	Impact caused by	Why is it important?
Reduced nutrient input and availability	Nutrient pools and fluxes of a range of chemical properties – input-output budgets	<ul style="list-style-type: none"> • Nutrient availability • Growth • Nutrient leaching 	<ul style="list-style-type: none"> • Harvesting • Prescribed burning • Tree species • Nutrient additions • Soil preparation 	An imbalanced nutrient supply where the output of nutrients is higher than the input will cause a decrease in soil stores of nutrients which can be interpreted as a warning for long-term negative productive effects.
Soil acidification	pH	<ul style="list-style-type: none"> • Mobility of heavy metals • Nutrient leaching 	<ul style="list-style-type: none"> • Harvesting • Prescribed burning • Tree species • Nutrient additions 	Most trees and soil organisms will have an optimum pH range for growth and the pH of the soil affects which species will grow best.
Compaction	Bulk density Macroporosity	<ul style="list-style-type: none"> • Pore size distribution • Soil moisture content • Root environment • Root and tree growth • Mortality 	<ul style="list-style-type: none"> • Off-road driving • Prescribed burning 	Compacted soils restrict air supply to tree roots, reduce root penetration, extension and germination and reduce capacity of the soil to store water.
Soil erosion	Soil formation rate Sediment yield	<ul style="list-style-type: none"> • Sedimentation in lakes and streams • Soil and nutrient transport 	<ul style="list-style-type: none"> • Harvesting • Off-road driving • Soil preparation • Tree species 	Loose soils may be subject to increased risk of erosion and dry out quickly.
Retainment of moisture and nutrients	Total C	<ul style="list-style-type: none"> • Organic matter content 	<ul style="list-style-type: none"> • Soil preparation • Prescribed burning • Tree species • Harvesting 	Total carbon is generally considered a good measure of organic matter in soil. Organic matter is important for soil quality because it helps soil retain moisture and nutrients, it contributes to a stable soil structure and it provides a source of food for soil microbes.

Table 2. A selection of case studies and model estimates of nutrient removals in single operations corresponding to clear-cutting.

Tree species	Stem diameter cm	Age years	Harvest	kg ha ⁻¹					Reference
				N	P	K	Ca	Mg	
<i>Case studies</i>									
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
<i>Model estimates^a</i>									
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

^a Note that estimates are theoretical figures of removal without taking into account whether it is technically possible to e.g. remove all above-ground biomass [202].

Table 3. 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
<i>Norway spruce</i>						
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
<i>Scots pine</i>						
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
<i>Birch</i>						
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

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4 The impact of forest management on water quality in Europe – *with a focus on nitrate.*

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Abstract

Water resources from forests are widely used for drinking water purposes in European regions and major cities. Strategies on forest management to protect or improve water quality are thus important. Here we review the impacts of major management operations (tree species choice, fertilisation, liming, clear-cut, thinning) on water quality with a main focus on nitrate leaching. The impacts are discussed for rotation scale, groundwater catchment or forest district scale as well as for alternative management strategies.

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. Callesen et al., 1999; 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 for 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 areas and less intensively managed land such as forests, although significant reductions in emissions have occurred (EEA, 2005). Forests function as receptors of airborne pollutants and are usually strong sinks for gasses and particles. The large canopy surface area in forests and the height of trees increase the filtering effect so that the deposition flux to forest is higher than for open land. Air pollution is thus an important external pressure on the protective function of forests on water quality. Furthermore, in the last century, 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 under discussion (Rothe et al., 1999; Andersson et al. 2000). 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. Sustainable forest management (SFM) aim to exploit the productive function and at the same time to preserve the protective functions of forest. To achieve this, more knowledge on the long-term and aggregated impacts of management (and from other pressures like air pollution) need to be incorporated into models and other decision tools that are available to the practical forestry.

Knowledge on the impact of forests management operations will be of increasing importance as the EU regulation related to water quality (2000/60/EC) 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 be relevant in relation to the Nitrate Directive (91/676/EEC) and the Drinking Water Directive (98/83/EC).

In this paper we focus on the potential threats to the protective function of forests on water quality. In the first sections we focus on the impact of a range of individual management operations (discussed in Raulund-Rasmussen et al., this issue) 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 compiled new data tables on the water quality response to management from original publications in the literature. In a discussion section we aggregate the effect of the individual operations to impacts at rotation scale and/or at the scale of a forest districts (where all periods of rotations will be represented). The purpose of this exercise is to qualitatively evaluate the impact of management alternatives with an increasing level of intervention as described in Raulund-Rasmussen et al. (this issue). The final section is a concluding summary, which also provides recommendations for future research.

Concepts and indicators

Water quality indicators

Water quality is an ambiguous concept. We therefore used a set of indicators of water quality that were previously identified (Raulund-Rasmussen et al., 2006) in the analysis. The identified indicators were pH, alkalinity, and the concentrations of nitrate, dissolved organic carbon (DOC), dissolved aluminium and heavy metals. For heavy metals the compounds need to be specified for a further analysis of the impact of forest management on these compounds. We included impacts on Cd, Pb and Hg in our review of the literature, although the information in relation to management impacts was dispersed. Where appropriate we also discuss management impacts on suspended solids in streams and lakes.

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.

The indicators cover a relative broad field of issues and environmental impacts. Our main emphasis in the analysis of impact from individual forest management operations are on nitrate leaching. First of all because nitrate leaching seems to be the most pressing European wide issue (EEA, 2005). 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 responses of the acidity and aluminium indicators (Dise et al., 2001) as well as of some heavy metals that are sensitive to acidification (section 2.2). The analysis of the acidity indicators is therefore incorporated with that of impacts on nitrate leaching. The impact of forest management on DOC and suspended solids are mainly of local importance and these indicators are thus only treated under the management operations where they are most relevant.

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)). Nitrate is highly mobile in soils and production in excess of plant and microbial uptake requirements will be transported through the soil profile (Qualls 2000). Gundersen et al. (2006) compiled survey data from the literature on nitrate leaching from mature undisturbed forests. Observed nitrate concentrations span a wide range ($0\text{--}50 \text{ mgN L}^{-1}$) in soil water and a lower and much narrower range ($0\text{--}3 \text{ mgN L}^{-1}$) 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.

Ammonium is absorbed on the soil cation exchange complex and is thus quite immobile in the soil profile. Consequently, NH_4^+ usually contributes less than 5% to the total dissolved N concentration in soil water (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 (Michalzik et al., 2001) and forest 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 $1\text{--}3 \text{ kg N ha}^{-1}\text{y}^{-1}$, since nitrate concentrations were very low in those streams (Campbell et al., 2000; Perakis and Hedin, 2002).

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 (and used in the European Community) is 11.3 mgN L^{-1} ($50 \text{ mg NO}_3 \text{ L}^{-1}$) (WHO, 1998). The potential effects of elevated N leaching in forest waters are not only related to this potential toxicity of nitrate when used for drinking water. It also involves numerous cascading effects in downstream ecosystems such as eutrophication and acidification (Galloway et al., 2003).

The sensitivity of waters 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 ($\text{pH}<4.5$) (an exception is snowmelt acidification pulses but they are not related to management) and only if the management activity lead to soil acidifying processes.

Indicator interactions

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 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 by Hansen et al. (this issue).

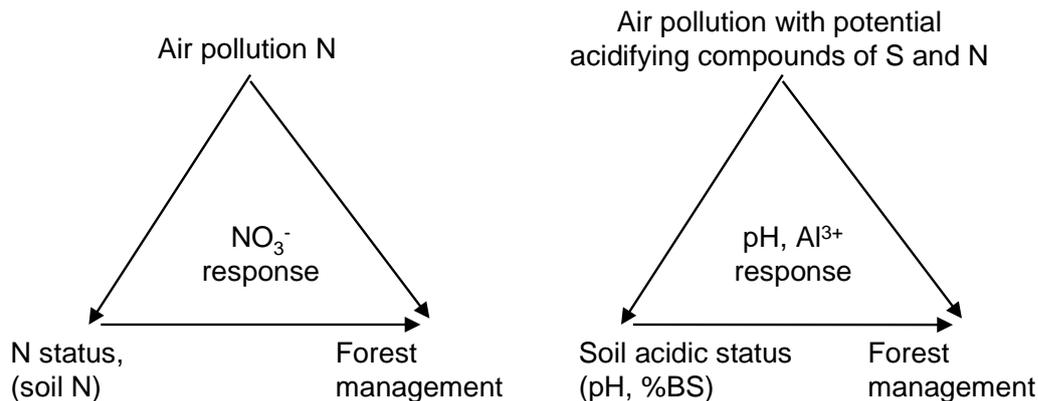


Figure 1. The water quality response (nitrate and acidity) to forest management operations is difficult to isolate from the impacts from the response due to air pollution. The status of the soil (which can be modified by air pollution) is also important in determining the response to management.

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).

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 causes alkalinity changes and fluctuations in Al³⁺ concentrations. Strong mobilisation of Al³⁺ related to nitrification and nitrate leaching has been shown in acid soils (e.g. Mulder et al., 1988; Wilpert et al., 2000). Figure 2 illustrates the interdependence of

nitrate, Al^{3+} and Cd in an acid soil. Soil water concentration of Al and Cd increase in parallel with nitrate when the pH is below 4.5.

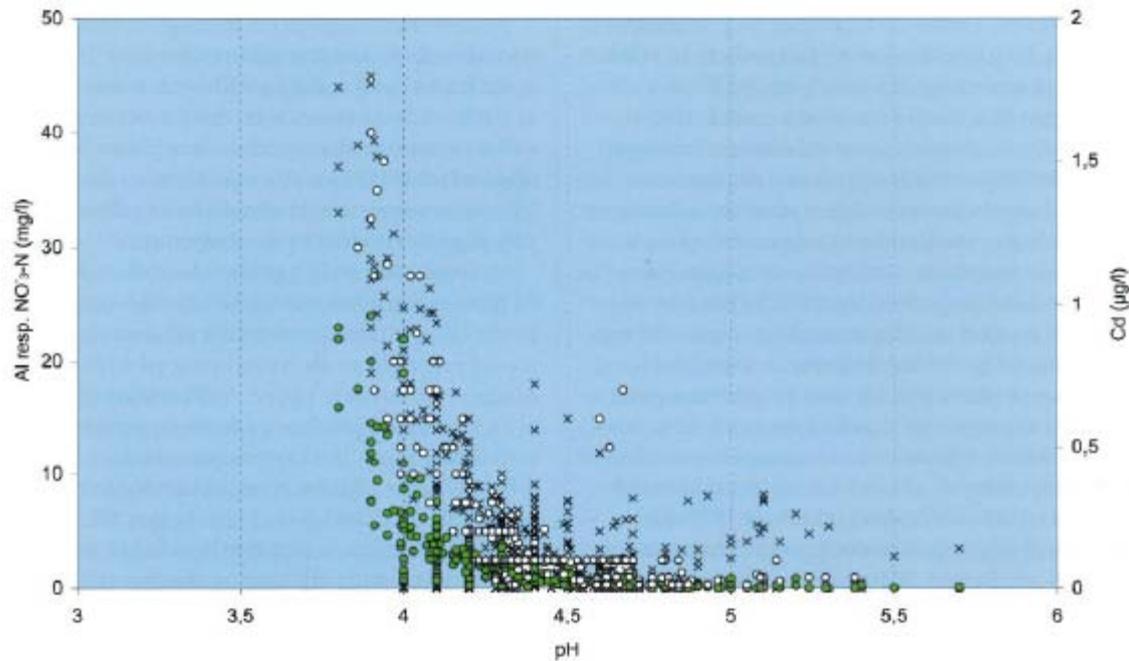


Figure 2. Soil water chemistry data from a clear-felling in Central Sweden where soil water was sampled at 50cm depth under plots representing different residue management strategies. Nitrate accumulation (x) and acidification occurred under residue piles mobilising Al (•) and Cd (o) (Ring et al. unpublished data).

Methods and approaches

As a starting point for the analysis of impacts on nitrate we used the 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 by Raulund-Rasmussen et al. (this issue) we have searched the literature for new data on nitrate leaching and for data on other impacts on water quality. For the other indicators than nitrate less information is available and often hidden within other issues. We include data on DOC, dissolved aluminium and heavy metals when relevant information have been identified.

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 concentrations of the substances is the relevant unit of measurements (e.g. nitrate in mg N L^{-1}), whereas for understanding the budgets and for quantifying the acidity produced in the system the fluxes from the forest ecosystems (e.g. leaching of N in $\text{kg N ha}^{-1}\text{y}^{-1}$) is the relevant unit. Where relevant and if the information in the compiled literature is available we will use both units.

The impact of potential management operations may in addition to air pollution and soil status be influenced 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 but for this paper we aggregate the information and only include the most important combinations of conditions and management operations.

Impacts of forest management operations

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 (Hansen et al. this issue). 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 (Kristensen et al., 2004; De Schrijver et al. 2008). This is shown by an enrichment of throughfall for N compounds in paired comparisons of coniferous and deciduous stands (e.g. Rothe et al., 2002 and references in Supplement Table 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.

The observed differences in N deposition between conifers and deciduous are reflected in higher output of nitrate from the soil under conifers (Figure 3, Supplement Table 1) when compared to deciduous species at the same site. Similar patterns were found by Rothe et al. (2002) at 12 other European spruce-beech pairs located directly adjacent to each other. They concluded that as the deposition level increases the species-related difference in nitrate leaching also increases. The 12 beech stands had 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 (Christiansen, 2006, 2010) show relative high nitrate leaching from beech (Supplement Table 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 (Supplement Table 1). Data from common garden experiments with broadleaf species (Vallø and Matrup sites in Supplement Table 1) reveal important differences in N retention among broadleaves (Christiansen et al., 2010) Maple and ash could retain all N inputs on a former agricultural field at high deposition (Matrup), but on an old forest soil (Vallø) oak and lime also had high N retention and no nitrate leaching (Table 1).

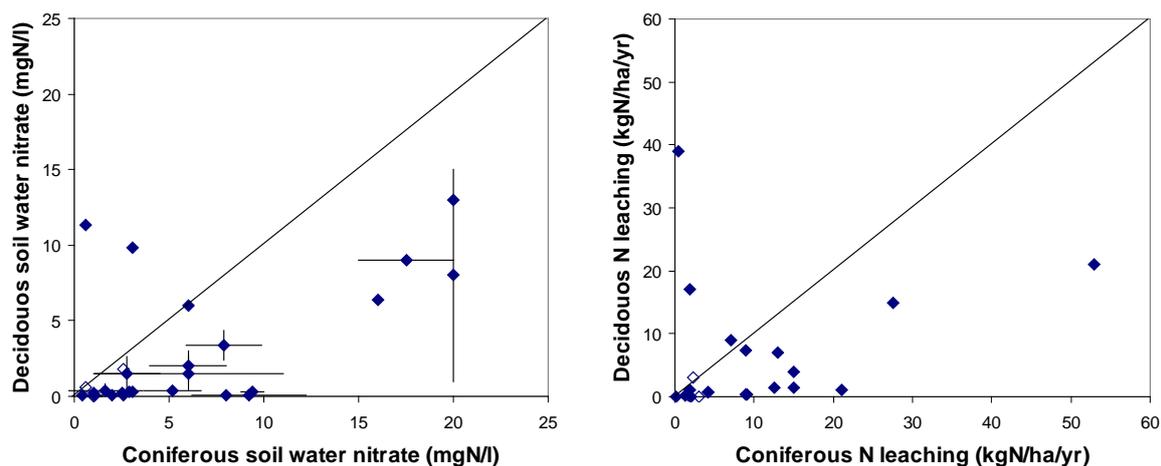


Figure 3. Comparison of deciduous and coniferous stands a) nitrate concentration in seepage water and b) N leaching fluxes from Norway spruce vs beech (filled symbols) and other species (open symbols) in adjacent stands on similar soils across Europe (Data from Supplement, Table 1)..

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 so far no documentation that this will actually 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 conifers to broadleaves 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, the 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. 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). Accordingly, high levels of nitrate in soil water (Binkley et al., 1992; Robertson et al., 2000) and in streams (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 the alder stand in UK ranged from approximately 3 to 6 mg N L⁻¹ (Robertson et al., 2000), which are only comparable to soil water concentrations found in high deposition regions in Europe (Gundersen et al., 2006). Cutting of alder stands and thereby drastically reducing the fixation N input decreased nitrate levels post cut (Robertson et al., 2000). Thus elevated N leaching is readily reversible when fixation input is reduced.

Site preparations

Physical manipulations

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 NO₃-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⁻¹y⁻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, Åström et al. (2002) found practically no effect of ditching on N export in a boreal catchment.

Chemical treatments

Fertilisation

Forest fertilisation with N is a common practice in intensively managed plantations. 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 NO₃-N L⁻¹ for at least a year in 34% of the studies. Some of the highest concentrations 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 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).

The highest concentrations in stream water (4 mg NO₃-N L⁻¹) are reported after fertilisation in N-saturated forests in Japan (Ohruai and Mitchell, 1998). 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.

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⁻¹yr⁻¹) probably have limited effect on tree growth and may accelerate the rate towards N-saturation.

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) 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). One recent concern about liming is the increased levels of nitrate in the soil solution as observed in a number of studies (Supplement Table 2; De Boer et al., 1993; 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 (Supplement Table 2; 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.

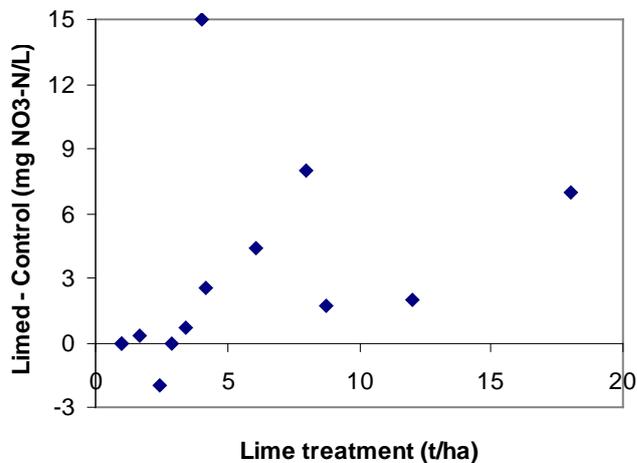


Figure 4. Nitrate concentration response (limed – control) versus dose of lime (or wood ash) applied. Data from Supplement Table 2 with wood ash doses converted to their equivalent lime dose.

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 (Figure 4), 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 (Supplement Table 2). 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.

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 factors for the duration and the extent of the N-cycle disruption (Gundersen et al. 2006).

Clear-cutting

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) 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 $150 \text{ kgN ha}^{-1}\text{yr}^{-1}$). Later, other studies of clear-cuts more close to commercial practice were performed at HBEF (Bormann and Likens, 1979) and numerous studies have been preformed in Europe (Supplement, Table 3).

In general, the nitrate concentration in soil and stream waters increase with peak nitrate concentrations within 2-3 years after clear-cut (Supplement, Table 3). 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). The response in concentration the seepage or runoff water is limited since at the same time the amount of seepage or runoff water increase due to lower evapotranspiration (Katzensteiner, this issue). Thus in terms of fluxes N leaching increase more than indicated by the increase in concentration (Supplement, Table 3).

A study by Vitousek et al., 1979 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).

Among the studies in Supplement, Table 3 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.

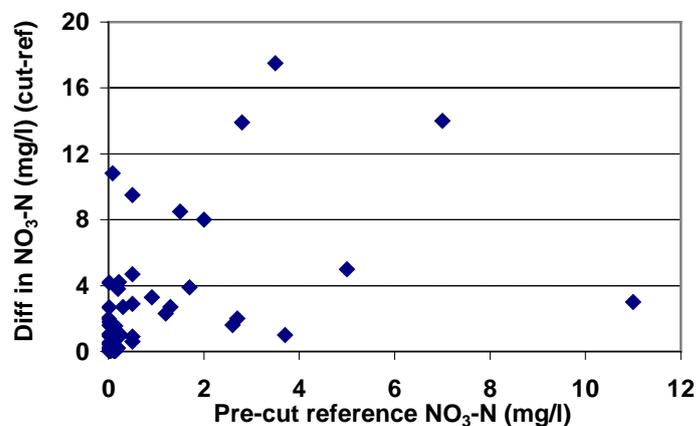


Figure 5. The increase in mean annual seepage water nitrate ($\text{mg NO}_3\text{-N L}^{-1}$) after clear-cut (nitrate response), i.e. the difference between cut stands and intact reference stands, plotted as a function of pre-cutting nitrate concentration. Data are calculated from Supplement Table 3.

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 Supplement, Table 3. 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 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 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 5 since pre-cut levels of nitrate were high (Supplement, Table 3) due to high N input from N fixation. In this forest, cutting decreased nitrate concentration levels (Robertson et al., 2000) probably due to the reduction in fixation N input and decreased evapotranspiration.

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 (Stupak, 2008). 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).

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). Decomposing logging residues may be an important sink for N due to the high C:N ratio in the material. 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. Across the studies, we found no general difference in nitrate leaching following CH and WTH. This indicates that responses in nitrate leaching to CH and WTH are a function of pre-existing site conditions; however the number of studies is small.

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). WTH 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. In large parts of Europe N will be replaced by deposition 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 (Hansen et al. this issue). 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.

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 (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. 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).

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 (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. 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), 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} .

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. 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).

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 3). 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 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 $1800 \text{ kg N ha}^{-1}$) (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. An extensive 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 $25 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Akselsson et al., 2004) 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, 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 $<5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ at low N deposition and $25\text{-}35 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ at high N deposition (Akselsson et al., 2004).

Other silvicultural regimes

Alternatives to clear-cut harvest include regeneration in gaps, selective cutting or thinning creating uneven 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; Emmett et al., 1993; Rothe and Mellert,

2004; Stevens et al., 1994). 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 10 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. 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.

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 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).

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.

Other management options

In mountainous or hilly areas building of infrastructure in particular roads both permanent roads and temporary trail for harvesting have impacts on hydrology and may lead to erosion. 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.

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 (Mayer et al. 2008 and references therein). The riparian zone in forests (the transition zone from upland to wetland) can effectively remove nitrate by denitrification and vegetation uptake as well as reduce sediment and DOC transport (Gundersen et al., 2010). 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. The complex issue of riparian management in forests was recently reviewed by Broadmeadow and Nisbet (2004) and Gundersen et al. (2010).

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, avalanches 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 6.1 and Table 2 we include the information from clear-cut plots that originated from windthrows and there were no signs of differences in water quality impact compared to commercial clear-cutting. However, an important difference is the scale of the impact, where commercial clear-cuts affect some hectares within a particular forest district hurricane windthrow may affect most of that district. If it is situated on a groundwater reservoir the whole reservoir will experience largely elevated nitrate leaching for 3-5 years or longer, which potentially may impair water quality.

Another example is the risk that monocultures enhanced 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 damaging 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).

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. 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. Prescribed burning has thus been suggested as a tool to manage soil N storage (e.g., Fenn et al., 1998). This option may be particularly relevant for the Mediterranean 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.

Impacts at rotation and forest district scales

The discussion of forest management impacts on water quality often have focused on the effects of clear-cut. In the short term (usually up to 5 years after the cut) increased decomposition lead to elevated DOC and nitrate in runoff, which may have strong negative consequences in local down stream fresh water. In acid systems this may lead to an acidification pulse with elevated dissolved Al. But on the large scale this effect may easily be diluted. As an example Lepistö et al. (1995) concluded from a study of 20 forested catchments in Finland, that logging must occur in a large fraction of a catchment before any effect on mean stream water nitrate is detectable.

At the rotation scale as well as at the landscape scale (i.e. an groundwater reservoir) the impact of harvest is of minor importance, especially since the peak in N leaching is followed by a period of rapid re-growth and high N retention (Klimo and Kulhavy, 1994; Rothe and Mellert, 2004) as illustrated in Figure 6. The response is usually most pronounced at high N status but the duration is longer at low N status (early rotation in Figure 7), both the extent of

the peak and the duration is dependent on the reestablishment of the plant N sink. This peak in N leaching may have a short local impact on freshwater.

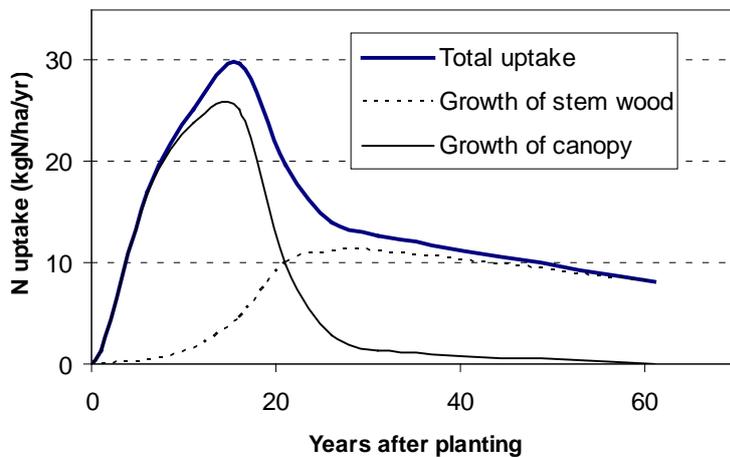


Figure 6. Nitrogen uptake in aboveground biomass over time in an aggrading oak forest on a fertile soil calculated from biomass accumulation (Gundersen et al. 2003).

In the period up to canopy closure the N cycle is dominated by a strong plant N sink, since the trees are building the N rich canopy components (twigs, foliage, bark) (Figure 6). Thus at the rotation scale (and landscape scale considering that stand ages are approximately equally distributed in a forest district) the N leaching impact is dominated by the conditions at the mature stage (Figure 7), i.e. the peak in nitrate after harvest is outweighed by the period with no leaching following. The N leaching in mature forests is in part determined by the soil C/N ratio i.e. the strength of the soil N sink (Gundersen et al. 1998, 2009). Soil C/N ratio usually refer to the organic layer. This works well for soils with mor or moder humus types, but for mull type soils mineral soil C/N ratios (0-5 cm) need to be used. Based on these considerations N leaching over the rotation scale can be predicted as follows (illustrated in Figure 7):

$C/N > 30$: The soil N sink dominate the N cycle. N is effectively retained. However at high N deposition (> 20 kgN/ha/yr) some leaching may occur as hydrologic driven leaching. N-leaching = 1-2 kgN/ha/yr

$C/N < 20$: The N cycling is dominated by the plant N sink. The soil can no longer retain significant amounts of N. Thus over the rotation N-leaching approximates Σ N-deposition – Σ N-removal in products. Forests planted on former arable soil are typical examples of this type (Hansen et al. 2007).

$20 < C/N < 30$: Here the soil N sink is less predictable (although it will still be related to and decreasing with soil C/N) and will depend on the level of N deposition as well (Figure 8). A combination of the two above scaled by soil C/N can be used as a reasonable estimate. Better would be to use measurements such as simply a KCl-extraction of soil mineral N below the root zone to improve prediction.

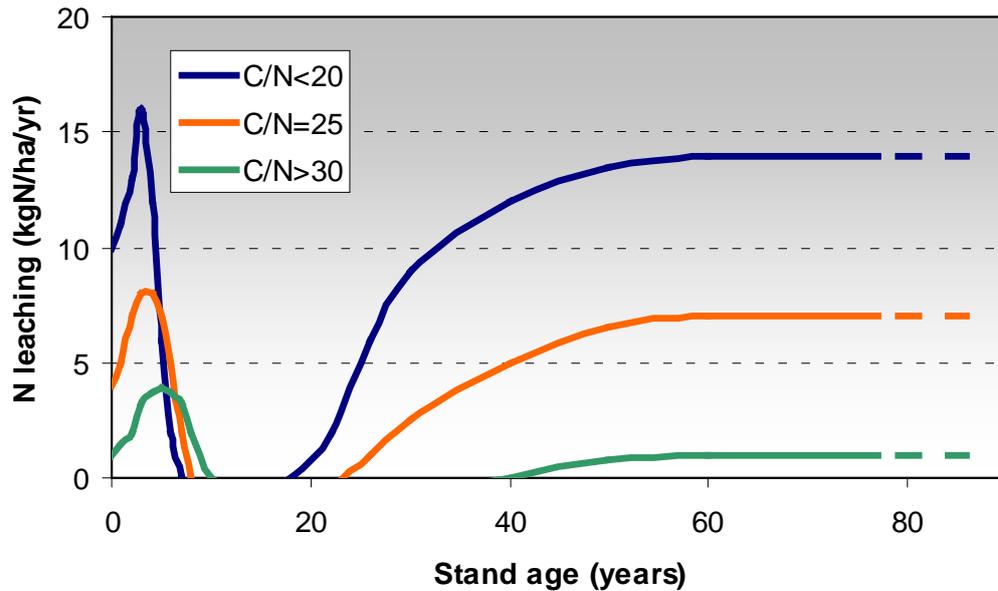


Figure 7. Temporal development of N leaching over a rotation after clear-cut for forests with different soil C/N ratio. Conceptual figure suitable for forests exposed to high N deposition ($>20 \text{ kg N ha}^{-1} \text{ yr}^{-1}$).

For further evaluation of that impact on nitrate leaching caused by management activities, the role of air pollution and site conditions needs to be accounted for as noted in section 2.2 and Figure 1. The interaction is summarised in the following and illustrated as a response surface in Figure 8. There is a threshold at $8\text{--}10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in throughfall input below which almost no nitrate is leached and above that N deposition explains approximately half of the variability in N leaching (Gundersen et al. 2006; Dise et al. 2009). 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 (see also previous slide). 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 main influence of management on nitrate response surface are exerted through: (i) *The tree species choice*. Tree species have different filtering capacity and conifers generally receive higher dry deposition than broadleaved species (section 4.1). This effect can probably be attributed to a leaf area (LAI, average over the year) effect and species specific effect on turbulence. As a rough approximation the following can be used: Conifer dry deposition N = $2 \times$ Broadleaf dry deposition N. Through tree species choice forest management thus can change N deposition to a forest area. (ii) *The effect of N removal in products*. As discussed previously the soil N sink is negligible at low C/N ratio soils (>20) and increasing as C/N increases, thus at lower C/N ratios the size of the plant N sink gets important. For forests with no or minimal management (protected forest reserves) the response surface in Figure 8 may be lifted (N leaching = N deposition, at low C/N), whereas increased N removal, e.g. by use of early thinnings and harvest residues for bioenergy, will reduce N leaching (lower the response surface in Figure 8).

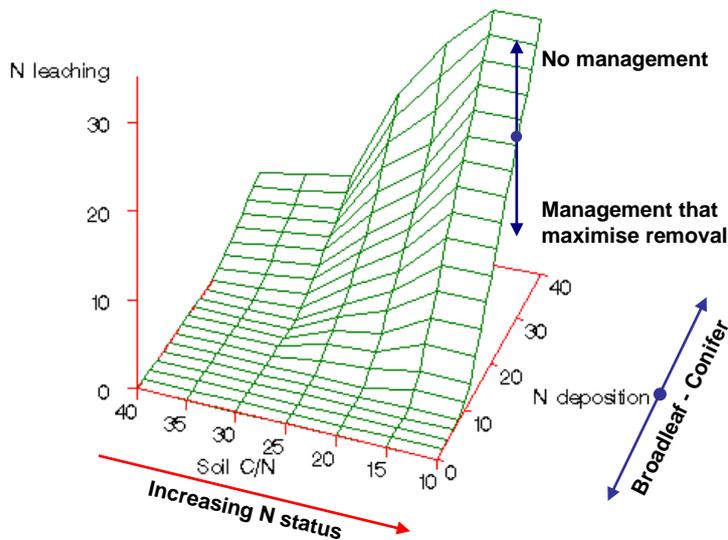


Figure 8. Nitrogen leaching response surface where N leaching is controlled by soil N status and affected by N deposition (conceptual figure based on thresholds and responses identified in Gundersen et al. (2006), see text). Change of tree species may reduce (or increase) N deposition to a site and thereby change N leaching. Increased N removal in harvest may lower the response surface.

In a N saturated forest district used for groundwater abstraction in Germany, Rothe et al. (1999) did a survey of seepage nitrate concentrations from differently managed stands. They used the observed variability in nitrate concentrations between these stands in an empirical model to predict the effects of potential management changes such as early regeneration of spruce, shelter wood regeneration and higher abundance of deciduous trees on the area mean nitrate concentration. The predictions showed that only scenarios with drastic changes to the forest management plan (such as complete conversion of conifer stands to deciduous stands) could reduce nitrate leaching noticeably. In 20 years the current mean nitrate concentration of mgN L^{-1} was only predicted to decrease from 5 to 4 mgN L^{-1} when the economically and practically feasible management changes were considered. This illustrates the difficulty in reducing nitrate leaching from an area already N saturated by management alone. A reduction in N deposition would directly lead to reductions under N saturated conditions (e.g. Gundersen et al. 1998b).

Synthesis

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.

Knowledge gaps

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 (Katzensteiner, this issue). Although differences between conifers (mainly Norway spruce) and broadleaves (mainly beech) are well documented there is a need for more comparisons of N cycling among tree species grown on the same soil. Differences among broadleaves, but also among conifers, need to be investigated. We found 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 (i.e. quantitative analysis of temporal developments hypothesised in Figure 7). 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.

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.

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Appendix (Supplementary material)

Table 1. Differences in seepage water nitrate concentrations (or leaching flux) between mature stands of coniferous (C) and deciduous (D) species at the same site. When more than one stand per tree species were investigated the number of replicates are noted in brackets for each species and the mean (and standard deviation) is given for the nitrate concentration. Data compiled from European literature.

Site	Species	Nitrate in drainage		Deposition	Obs. period	Reference
		mg NO ₃ -N L ⁻¹	kg N ha ⁻¹ y ⁻¹			
Höglwald, Germany	C Norway spruce (3)	9.4 (0.6)	21	30	1985-89, 94-97	Rothe et al. 2002
	D Beech (3)	0.3 (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 (2)		15	26		Wilpert et al. 2000
	D Beech and Beech/S.fir		4	17		
Nordrehein- Westfalen, Germany	C Pine		~3	11.2	1985-1987	Van Grinsven et al. 1991
	D Oak/Beech		~0	5.6		
Eurasburger Forst, Germany	C Norway spruce (6)	9.2 (3)			Spring 1998	Rothe and Mellert 2004
	D Beech (4)	0.1 (0.1)				
Munchner Schotterebene, Germany	C Norway spruce (10)	1.6 (2)			1998-1999	Rothe and Mellert 2004
	D Beech (4)	0.4 (0.4)				
Stadtwald Landsberg, Germany	C Norway spruce (6)	5.2 (1.5)			1999-2000	Rothe and Mellert 2004
	D Beech (3)	0.4 (0.2)				
Forellenbach, Germany	C Norway spruce	1-4		16	1992-1995	Kleemola and Forsius 2000
	D Beech	0.2		13		
Gisburn, UK	C Norway spruce	0.5			, prior to clear felling	Robertson et al.2000
	C Scots pine	1.5				
	D Oak	0				
de Driesprong, The Netherlands	C Norway spruce (4)	2.8 (1.7)			January 1999	Tietema 2004
	C Douglas fir (5)	11 (5)				
	C Scots pine (10)	5.8 (3.0)				
	- Larch (5)	4.8 (6.1)				
Hoogmoerhei de, Belgium	D Beech (5)	1.5 (1.1)			1998-2000	De Schrijver et al. 2004
	C Corsican pine	16	53	96		
Aubure, France	D Silver birch	6.4	21	27		
	C Spruce		15	19		
Mont-Lozere, France	D Beech		1.5	9	1993-95	Dambrine et al. 2000
	C Norway spruce		1.8	10.9		
France	D Beech		0.2	7.9	1981-1986	Durand et al. 1992
	C Norway spruce		1.8	10.9	1981-1986	

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		
Munkarp, Sweden	Norway spruce	2.6	8.9	18.6	1990-1991	Bergkvist and Folkeson 1995
	Beech	0.1	0.4	14.6		
	Birch	1.8	7.3	11.3		
Munkarp, Sweden	Norway spruce	1.0	4.2	27.5	1990-1991	Bergkvist and Folkeson 1995
	Beech	0.1	0.7	10.4		
	Birch	0.2	0.8	10.6		
Vestskoven, Denmark ^b	C Norway spruce (3)	8 (2)	13			Hansen et al. 2007
	D Oak (3)	3.4 (1.0)	7			
Stenholt Vang Denmark ^b	C Norway spruce	6	7	24	1994-97 (D. fir and Oak 1995-96)	Hansen (ed.) 2003
	C Sitka spruce	20	25	25		
	C Douglas fir	15	30	12		
	D Beech	6	9	9		
	D Oak	9	15	10		
Ulborg, Denmark	C Norway spruce	0.4	0.2	17	1994-97 (D. fir and Oak 1995-96)	Hansen (ed.) 2003
	C Sitka spruce	1.5	2.6	21		
	C Douglas fir	0.7	1.3	20		
	D Beech	0.05	<0.1	13		
	D Oak	0.05	<0.1	12		
Lindet, Denmark	C Norway spruce	2	1.3	24	1994-97 (D. fir and Oak 1995-96)	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, Denmark	C Norway spruce	0.6	0.5		2000	Christiansen et al. 2006
	D Beech	11.3	39			
Vallø, Denmark	C Norway spruce	3.1	1.9	23 ^c	2004-2006	Christiansen et al. 2010
	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		
Matstrup, Denmark ^b	C Norway spruce	19		37 ^c	2004-2006	Christiansen et al. 2010
	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 Afforestation on former agricultural field, 30-40 years since cultivation.

^c Significant canopy uptake may be the reason for the lower N input in deciduous than coniferous throughfall.

Table 2. 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 mm	Nitrogen deposition kg N ha ⁻¹ yr ⁻¹	Lime treatment Dose (and type) t/ha	Nitrate control stands mg NO ₃ -N L ⁻¹	Nitrate in limed stands (kg N ha ⁻¹ yr ⁻¹)	Treatment effect	Reference
Seepage										
Klosterhede, Denmark	56 years <i>Picea abies</i>	Haplic podzol	26	860	15	1.7 (CaCO ₃) 0.7 (MgSO ₄)	0.01	0.01-0.74	Yes	Ingerslev 1997
Uddevalla, Sweden	<i>P. abies, Pinus sylvestris</i>	Podzol	n.d.			1 (CaCO ₃)	0.2	0.2	No	Nohrstedt 1992
Vallsta, Sweden		Podzol				1 (CaCO ₃)	<0.03	<0.03	No	
Hasslöv, Sweden	<i>P. abies</i>	Haplic podzol	20-27	1100		3.45 (CaCO ₃)	0.27 (1)	0.98 (5)	Yes	Persson and Wiren 1996, Nilsson et al. 2001
						8.75 (CaCO ₃)	0.27 (1)	1.99 (10)		
Öringe, Sweden	<i>P. abies</i>					3.35 (CaCO ₃)	(3)	(10)	Yes	Persson and Wiren 1996
Åled, Sweden	<i>P. abies</i>	Dystric Regosol			14	4.2 (wood ash) ^a	0-1	0-6	Yes	Högbom et al. 2001
Grunewald, Germany	<i>Pinus sylvestris</i>	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	<i>P. abies</i>	Alfisol	23-27	850	30	4 (dolomite)	~15 (33)	~30 (55)	Yes	Kreutzer 1995, Kreutzer and Weiss 1998
Convent Forest, Germany	<i>P. abies</i>	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	<i>P. abies</i>	Dystric Cambisol	16	1076		8 (wood ash) ^d	~2 ^e	~10	Yes	Brunner et al. 2004
St. Anthonis, Netherlands	<i>Pinus sylvestris</i> <i>Pseudotsuga menziesii</i> <i>Quercus robur</i>	podzol	20-24	450		3 (dolomite)	(1.6) young pine (6.9) old pine (3.6) young fir (13.9) old fir (2.5) young oak (9.4) old oak	(10.8) (12.7) (12.9) (36.5) (1.1) (17.6)	Yes	De Boer et al. 1993

Site, stand type	Dominant tree species	Soil type	C:N ratio forest floor	Rainfall mm	Nitrogen deposition kg N ha ⁻¹ yr ⁻¹	Lime treatment Dose (and type) t/ha	Nitrate control stands mg NO ₃ -N L ⁻¹	Nitrate in limed stands (kg N ha ⁻¹ yr ⁻¹)	Treatment effect	Reference
Stream water										
Gjastad, Norway	<i>P.abies</i> <i>P.sylvestris</i>	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).

Table 3. 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 $\text{kg N ha}^{-1}\text{y}^{-1}$	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 $\text{kg N ha}^{-1}\text{y}^{-1}$	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 $\text{kg N ha}^{-1}\text{y}^{-1}$	Keersmaecker et al. 2000
Eiseneck, Austria	Seepage	Norway spruce	~0	3-6		Input 8 $\text{kg N ha}^{-1}\text{y}^{-1}$ Flux change 0.5 to 15-27 $\text{kg N ha}^{-1}\text{y}^{-1}$	Katzensteiner 2003
Convent Forest, Black Forest, SW Germany	Seepage	Norway spruce		15	2-3	Input 25 $\text{kg N ha}^{-1}\text{y}^{-1}$, details missing	Wilpert et al. 2000
Ebersberg Forest, Bavaria, Germany	Seepage	Norway Spruce	0-2	17	1-3	Input 25 $\text{kg N ha}^{-1}\text{y}^{-1}$, 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

Windthrown, site 10, Bavaria, Germany	Seepage	Norway Spruce	<5	10	4	for control. Windthrow	Mellert et al. 1998
Münchner Schotterebene, Germany	Seepage	Conifer, n=3	0-2 ^b	6-10		Survey in forest district. 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

5 The impacts of forest management on the carbon cycle

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1 Introduction, concepts and indicators

Fixation and accumulation of carbon in forest ecosystems has to be based on data of the forest stand inventory including annual and mean increment but also on data obtained by studies carried out in long-term field research stations. Particularly carbon fluxes between atmosphere, forest stand biomass and soil are affected by forest practices including methods of the stand establishment, soil preparation and fertilization, species composition, logging methods and using the wood material but also by factors such as fires, windbreaks and insect pests under concrete climatic conditions.

We considered in this review the forest ecosystem including the soil and vegetation under influence of forestry practices.

The carbon element is the major constituent of anthropogenic greenhouse gases molecules such as the carbon dioxide, CO₂, methane, CH₄, 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 N₂O 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 fertilisers production and application are rarely considered.

The terrestrial carbon cycle includes five major pools, the atmosphere, soil, biomass, harvested products and fossil fuel (Figure 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).

For example, carbon stock in the wood biomass of forests of EU countries is dependent both on the area of forests and the production level of forest stands reaching following values:

> 1000 mil. t (Sweden)

500 – 1000 mil. t (Germany, France, Finland, Austria, Poland)

200 - 500 mil. t (Romania, Italy, CR)

100 – 200 mil. t (Spain, Slovakia, Lithuania, UK, Hungary, Latvia, Estonia, Bulgaria)

50 – 100 mil. t (Netherlands, Denmark, Luxemburg, Cyprus, Malta) (according to Paschalis-Jakubowicz 2004).

The highest value of carbon accumulation in wood biomass is mentioned by Paschalis-Jakubowicz (2004) on the example of Austria (more than 70 mil. t carbon per 1 mil. ha forest) and the lowest value on the example of Cyprus and Spain (less than 20 mil. t per 1 mil. ha forests).

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 oxydised as CO₂ 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 CO₂ 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, CH₄, volatile organic compounds (isoprene etc.) contribute marginally to the exchange between the ecosystem and the atmosphere. Methane is produced under anerobic 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.

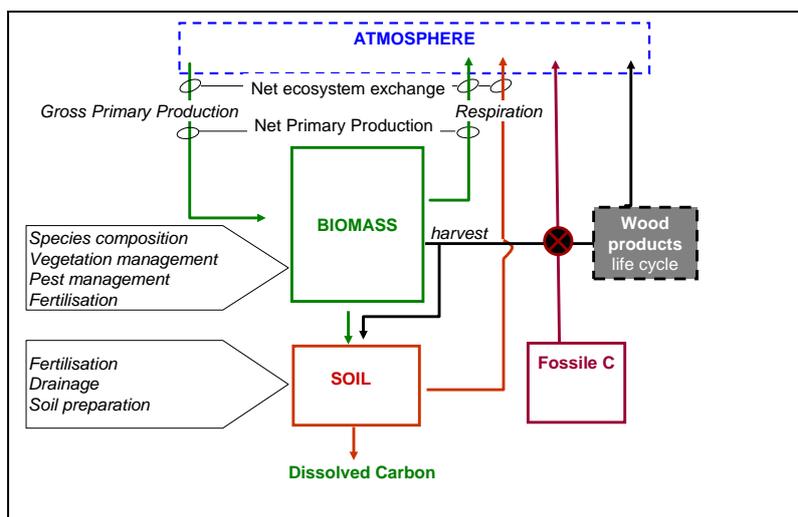


Figure 1. Carbon cycle in a managed forest ecosystem and main effects of management practices.

Soil appears to be the highly important sink of carbon. The ratio of soil: vegetation C density increases with latitude. Land use change particularly conversion to agricultural ecosystems, depletes the soil C stock. Consequently, afforestation of agricultural soils and management of forest plantations can enhance SOC stock through C sequestration. The rate of SOC sequestration, and the magnitude and quality of soil C stock depend on the complex interaction between climate, soils, tree species and management, and chemical composition of the litter as determined by the dominant tree species. Soil C sequestration in boreal and temperate forests may be an important strategy to ameliorate changes in atmospheric chemistry (K. Lal 2005).

Tab. 1 Carbon stock in selected biomass of the world (Lal 2005).

Biom	Area	C density Mg/ha		C stock (Pg)	
		vegetation	soil	vegetation	soil
Tundra	927	9	105	8	97
Boreal/Taiga	1372	64	343	88	471
Temperate	1038	57	96	59	100
Tropical	1755	121	123	212	216
Wetlands	280	20	723	6	202
Total	5672	Mean 54	Mean 189	373	1086

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. 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 CO₂ or CH₄ affects differentially the climate (Table 2). It is therefore recommended to distinguish each greenhouse gases as far as possible. Table 3 lists the main operational

indicators derived from this approach and proposed within the framework of the EFORWOOD project.

Table 2 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 ₂ O	150	270	290	190

Table 3. 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 ₂ O pr ha pr year	Eforwood WCI (22)

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.

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.

2. Effect of management operations

2.1 Changes in land use

Conversion of arable land to forest or arable land and pasture is converted to forest, additional carbon dioxide will be removed from the atmosphere and stored in tree biomass. The carbon stock on that land increases. Carbon sequestration rates vary by tree species, soil type, regional climate, topography and management practice.

Stevens and al. (2008) quoted Gifford (2002) calculation an average increase soil organic carbon stock after afforestation of cropland to coniferous of 53%.

Table 4 Initial soil organic carbon stocks and standard deviation for each land use calculated from the National Soil Survey (Stevens et al. 2008)

Land use	Mean (t C.ha ⁻¹)	S D (t C.ha ⁻¹)
Coniferous	75.9	31.6
Grassland	68.6	25.9
Cropland	52.5	21.1
Heathland	48.6	30.2
Deciduous	66.9	33.9

2.2. Site preparation – Physical treatments

2.2.1 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; Örländer, 1996). Soil preparation is the second most expensive forest operation in fossil carbon after harvesting operations.

2.2.2. 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 (Minkinen 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 peatlands are 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 peatlands is uncertain (Minkinen 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-CO₂ . 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 N₂O 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 peatlands have a net cooling effect which suggests that drainage and afforestation should mainly target recent peatlands with low carbon stocks.

Drainage can increase the loss (output) of soluble organic carbon to watercourses in soils with the high accumulation of surface humus in waterlogged peat areas.

2.3. Site preparation – Chemical treatments

2.3.1. 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).

2.3.2. 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 moderate 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, in neither studies was an assessment of the complete carbon balance of the ecosystem achieved, making it impossible to realise how far the enhanced biomass growth could compensate the carbon loss by faster organic matter decomposition, lime decarbonation and associated fossil carbon loss due to lime extraction, transport and application.

2.4 Tree species

2.4.1. 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). For example, data from long-term projects in Central Europe mention carbon accumulation in forest floor in spruce stands about 25 000 kg.ha⁻¹, in beech stands 10 000 kg.ha⁻¹ and in oak mixed stands 2 700 kg.ha⁻¹ (Reichle 1981, Klimo, Maršálek 1992, Penka et al. 1985). In addition to species composition, these values can be also affected by site conditions. 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 the 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 5).

Table 5. 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]	ΣC [t/ha]
<i>Pinus sylvestris</i> (Scots pine)	490	60	62	122
<i>Picea abies</i> (Norway spruce)	430	74	140	214
<i>Abies alba</i> (Silver fir)	410	100	128	228
<i>Fagus sylvatica</i> (beech)	680	119	147	266
<i>Quercus</i> sp. (oak)	660	83	102	185

Sequestration of carbon in the humus layer of Swedish forests is described by Berg et al. (2009): The mean sequestration rate was 251 kg C.ha⁻¹.year⁻¹, which is higher than theoretical values. The sequestration rate was positively related to temperature sum, albeit including effects of forest management. The pine – dominated forest type had a mean rate of 238 kg C.ha⁻¹.year⁻¹, and spruce – dominated had a mean rate of 239 kg C.ha⁻¹.year⁻¹. Under similar site conditions, pine sequestrated more C than spruce, showing the importance of this type of ecosystem for C sequestration.

2.4.2. 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 impacts of the understorey vegetation on the carbon storage is therefore very much depending on the local site and management conditions.

2.5. 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 CO₂ 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 6 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 or 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 6. Impact of three management scenarios on carbon stocks (a), and relative global climate impacts (b) partitionned among biogeochemical, 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 (tC.ha ⁻¹)			Mean annual increment (m ³ .ha ⁻¹ .an ⁻¹)	Fossil C replaced (tC.ha ⁻¹ .an ⁻¹)
	Biomass		Soil		
	trees	other			
Management intensity					
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)			
	Carbon storage	Fossil carbon	Albedo	Total
integration time				
20 y				
igh	0.16	-0.11	-0.01	0.04
ow	-0.22	0.11	0.01	-0.10
50 y				
igh	0.16	-0.27	-0.01	-0.12
ow	-0.22	0.27	0.01	0.06

2.5.1. 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.

The effects of forest management practices was described by Nabuurs et al. (2008):

Forest can act as a carbon source or sink, depending on the balance between uptake of carbon through photosynthesis and release of carbon through respiration, decomposition, fires or removal by harvest activities.

After final harvest or thinning the rate of decomposition of slash on the ground is higher than accumulation of carbon in the vegetation and soil (Mäkipää et al. 1999 in Nabuurs et al. 2008).

Rotation length:

Most European studies indicate an increased total carbon accumulation in biomass and soil if rotation lengths are increased (Lash et al. 2005) in Nabuurs et al. 2008).

Regeneration regim:

Regeneration regimes can be characterized by the degree of canopy cover in one cut.

This can range from single – tree selection system to clearcut. The higher the share of standing stock that is cut, the higher is the input to forest floor C pools from residues (Bergh et al. 2003 in Nabuurs et al. 2008).

Tending (weed control):

Trees and weeds cut in tending operations are not usually removed from the site. The decomposition of their foliage, stems, and roots increases soil C content (Paul et al. 2002 in Nabuurs et al. 2008).

Thinning:

Thinning is an active reduction in stem number during the rotation of a stand. The aims of thinning include enhancing the growth of the remaining trees. Kramer 1988 in Nabuurs 2008 found that light to medium thinning from below can increase overall production by 3-11% compared to un-thinned Norway spruce stands in central Europe. Soil carbon may (temporarily) be enhanced due to increased litter input, but change in microclimate could also lead to increased decomposition. Furthermore, decreased litter input afterwards or removal of thinning residues could lead to a decrease in soil carbon stock (Bergh et al. 2003 in Nabuurs et al. 2008).

2.6. 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 vulnerability 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, lignoculture 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.

3. 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 ($15.6 \text{ m}^3 \text{ ha}^{-1} \text{ year}^{-1}$). Notably, annual removal of 10 or 20% of woody biomass per year gave both a high timber yield ($25 \text{ m}^3 \text{ ha}^{-1} \text{ year}^{-1}$) 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 very 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 Luysaert (pers. com.) exemplifies this result for 229 forest sites (Luysaert 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.

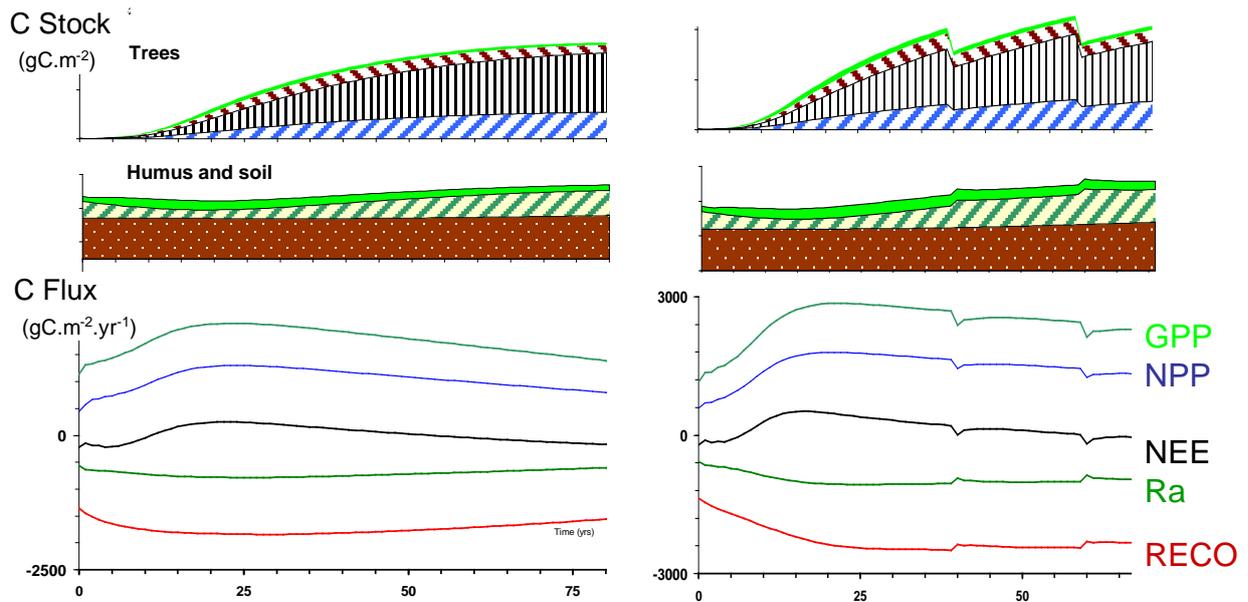


Figure 2. Course of carbon stock (upper diagrams) and flux (Lower diagram) in a forest ecosystem according to two management class, unmanaged (left) and managed (right).

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 leads generally to net carbon loss in the atmosphere¹ 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.

4. 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 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.

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.

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

Table 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			
<i>Prescribed burning</i> ¹	Decreasing	Decreasing, neutral or increasing	Decreasing, neutral or increasing
<i>Drainage of peatlands</i> ²	Decreasing	Increasing	Decreasing, neutral or increasing
<i>Site preparation method</i> ³			
Low-intensive	Neutral	Increasing	Increasing
Intensive	Decreasing	Increasing	Decreasing, neutral or increasing
<i>Tree species change</i> ⁴			
To conifers from broadleaves	Increasing	Increasing	Increasing
To broadleaves from conifers			
To mixed conifers and broadleaves from mono-specific coniferous	Decreasing	Decreasing	Decreasing
	Neutral or decreasing	Neutral or decreasing	Neutral or decreasing
Stem exclusion phase			
<i>Thinning method</i> ⁵	Neutral or decreasing	Decreasing	Decreasing
<i>Fertilisation</i> ⁶	Increasing	Increasing	Increasing
<i>Increased rotation length</i> ⁷	Decreasing, neutral or increasing	Increasing	Increasing
<i>Harvesting method</i> 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.

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 magnitudes.

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 favored in the future to provide a complete picture of the climate impacts of management scenarios.

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