13.05.02

METSÄNTUTKIMUSLAITOKSEN TIEDONANTOJA 859, 2002
THE FINNISH FOREST RESEARCH INSTITUTE, RESEARCH PAPERS 859, 2002

# Nutrient fluxes through a boreal coniferous forest and the effects of clear-cutting

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18.09.02

METSÄNTUTKIMUSLAITOKSEN TIEDONANTOJA 859, 2002
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*Academic dissertation*
*Faculty of Forestry*
*University of Joensuu*

To be presented, with the permission of the Faculty of Forestry
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of the University, Yliopistokatu 7, Joensuu,
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Joensuu, 2002

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FIN-01301 Vantaa
Accepted by Kari Mielikäinen, Research Director, 20.05.2002

ISBN 951-40-1839-7
ISSN 0358-4283

Gummerus Kirjapaino Oy
Saarijärvi 2002

# Abstract

Piirainen, S., 2002. Nutrient fluxes through a boreal coniferous forest and the effects of clear-cutting. The Finnish Forest Research Institute, Research Papers 859. 50 p. ISBN 951-40-1839-7, ISSN 0358-4283.

The objective of this thesis was to study the impact of practical clear-cutting (stem only) on the fluxes of the major plant nutrients: C, N, P, S, Ca, Mg and K, through a podzolic soil in a Norway spruce dominated mixed boreal forest. Changes in deposition, litterfall and, after clear-cutting, logging residue inputs of nutrients to the soil were also included. Data presented were collected from three sample plots during 1993 - 1999 situated in eastern Finland. The clear-cutting was carried out in September 1996 at two of the plots. Litterfall, bulk deposition, throughfall, stemflow and soil percolate from below different soil horizons before and after clear-cutting were collected at the sample plots.

Increased fluxes of dissolved organic carbon (DOC) from below the organic (O) horizon were observed after cutting. However, the flux (168 kg C ha$^{-1}$ a$^{-1}$) was small compared to the amount in the logging residues. The flux of DOC from below the illuvial (B) horizon (3 kg ha$^{-1}$ a$^{-1}$) was not affected by cutting. The deposition of total N (DTN) was small (4 kg ha$^{-1}$ a$^{-1}$), and the flux of dissolved inorganic N compounds through the soil was also small. After cutting some increased formation of $NH_4^+$ and $NO_3^-$ was observed. However, the DTN flux from below the B-horizon (0.2 kg ha$^{-1}$ a$^{-1}$) did not increase markedly after clear-cutting and the main fraction was organic N, as also before cutting. Increased fluxes of P and K from below the O-horizon were observed after cutting and the fluxes were the highest in the first year. The P retention capacity of the mineral soil was considerable and no significant increased fluxes of P from below the B-horizon (0.01 kg ha$^{-1}$ a$^{-1}$) were observed. Increased fluxes of K and Mg from below the B-horizon (1.6 kg K ha$^{-1}$ a$^{-1}$; 0.1 kg Mg ha$^{-1}$ a$^{-1}$) were occasionally observed. The deposition of S (2.0 kg ha$^{-1}$ a$^{-1}$) due to clear-cutting (loss of canopy interaction) decreased the flux of S from below the O-horizon, as also from below the B-horizon (0.7 kg ha$^{-1}$ a$^{-1}$).

Keywords: base cations, carbon, deposition, harvesting, nitrogen, phosphorus, soil water

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# Preface

This work has been carried out at the Finnish Forest Research Institute, Joensuu Research Centre and it has been a part of an ongoing project evaluating the effects of clear-cutting and site preparation on nutrients cycling, soil properties, soil water and stream water quality and yield in forested headwater catchments. The VALU project, has had project numbers 3087-05 in 1990-1996, 3218-04 in 1997-2000 and 3310-01 in 2001-2003. The VALU project was established as a continuation of the Nurmes study (e.g. Ahtiainen 1992), which was criticized for using atypical forestry practises the experimental treatments. Importantly, the forestry operations done in the VALU project were in accordance with the current silvicultural guidelines of the Finnish Forest and Park Service. My primary part in the VALU project was to study the changes in the quality and amount of deposition and soil water in upland sites.

I wish to thank my supervisors, Profs. Leena Finér and Hannu Mannerkoski from the University of Joensuu, Faculty of Forestry, for their support, guidance and endless patience. Special thanks are also due to Dos. Michael Starr from the Finnish Forest Research Institute, Vantaa Research Centre, who is a real expert on mineral soils, for rewarding co-operation and language revisions. I am also grateful to several persons who have been involved with their ideas and expertise to this cooperative VALU-project: Ms. Marketta Ahtiainen from the North Karelia Regional Environment Centre, Ms. Tiina Heinonen, Mr. Pekka Järviluoto, Mr. Urho Kettunen, Mr. Hannu Koivunen, Mr. Raino Lievonen, Dr. Anne-Marie Kurka, Prof. Eino Mälkönen, Mr. Ron Store and Mr. Markku Tiainen from the Finnish Forest Research Institute, Dr. Ahti Lepistö, Dos. Pertti Seuna and Dr. Pekka Vanhala from the Finnish Environment Institute, Mr. Juhani Karjalainen and Mr. Timo Impiö from the Finnish Forest and Park Service, and Mr. Jukka Höytämö from the North Karelia Regional Environment Centre, and Prof. Seppo Kellomäki, and Mr. Veikko Möttönen from University of Joensuu, Faculty of Forestry.

To the numerous field workers at the Nurmes field station of the Finnish Forest Research Institute I am grateful for the sampling and the pleasant atmosphere during the field trips. Special thanks to Mr. Timo Pelkonen and Mr. Esko Kokkonen, foremen of the field sampling crew.

Warm thanks also to Ms. Seija Repo, Ms. Anita Pussinen and Ms. Anki Geddala and to the whole staff of the laboratory of the Finnish Forest Research Institute, Joensuu Research Centre for analysis of the samples. With Seija in charge, the analysis of some 10 000 water samples was of the highest quality, reliable and always in time. I also wish to thank the staff of the Central Laboratory, Vantaa, and the laboratory of Muhos Research Station of the Finnish Forest Research Institute and the Department of Geography and the Faculty of Forestry of the University of

Joensuu for additional analysis. I am also grateful to Ms. Leena Karvinen for preparing the Figures and Tables and for assistance in the office work, and to Mr. Jaakko Heinonen for statistical advice. Special thanks also to our “coffee group” at Joensuu Research Centre for the enjoyable and interesting discussions during the coffee breaks, which brightened also those grey days.

I want to express my gratitude to my parents, Terttu and Jouko, and to my siblings, Pirjo, Jarmo and Virpi and their families. Thank you Mum and Dad for your support, encouragement and for passing on your love of nature. I am especially grateful to my husband Timo, and to my sons Henri and Rasmus for their support and love. Without you maybe I would have finished this work a little sooner, but it would not have reimbursed the great gratification I feel for being a very important person for you.

Finally I wish to thank Matti Heikurainen and The Ministry of Agriculture and Forestry for partly funding of VALU-project (project 4782/64/94) and the Finnish Ministry of Labour for supporting this work. Finally, I thank the Finnish Forest and Park Service for cooperation and making the site available and performing the forestry operations.

Pyhäselkä, April 2002 Sirpa Piirainen

## List of original publications

The thesis is based on the following papers, which in the text will be referred to by their Roman numerals.

**I** Piirainen, S., Finér, L. and Starr, M. 1998. Canopy and soil retention of nitrogen deposition in a mixed boreal forest in eastern Finland. Water, Air, and Soil Pollution 105: 165-174.

**II** Piirainen, S., Finér, L. and Starr, M. 2002. Deposition and leaching of sulphate and base cations in a mixed boreal forest in eastern Finland. Water, Air, and Soil Pollution 131: 185-204.

**III** Piirainen, S., Finér, L., Mannerkoski, H. and Starr, M. 2002. Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant and Soil 239: 301-311.

**IV** Piirainen, S., Finér, L., Mannerkoski, H. and Starr, M. 2002. Effects of forest clear-cutting on the sulphur, phosphorus and base cation fluxes through podzolic soil horizons (submitted to Biogeochemistry).

**V** Finér, L., Mannerkoski, H., Piirainen, S. and Starr, M. 2002 Carbon and nitrogen pools in an old-growth, Norway spruce-mixed forest in eastern Finland and changes associated with clear-cutting. Forest Ecology and Management (in press).

## The author's contribution

I-II The papers were initiated and planned jointly with L. Finér and M. Starr. Piirainen was responsible for the preparing of the first draft of the manuscript.

III-IV The papers were initiated and planned jointly with L. Finér, H. Mannerkoski and M. Starr. Piirainen was responsible for the preparing of the first draft of the manuscript.

V Sirpa Piirainen has performed part of the experimental work. She has also participated in the data analysis and preparation of the manuscript.

# Contents

# 1 INTRODUCTION

## 1.1 Background

The history of nutrient cycling studies in forests is long, since the first study was published in the end of 19th century (reviewed by Attiwill and Adams 1993). In the 1960s nutrient cycling studies were extended from the stand level to the catchment level so that input-output balances for the whole ecosystem could also be studied. However, much of the present knowledge about nutrient cycling and fluxes in upland forest ecosystems is based on studies done in temperate forests. Also much of the knowledge about the effects of forest cuttings comes from temperate forests. One of the oldest, most versatile and still continuing long-term study, the Hubbard-Brook Ecosystem Study (HBES), was established in North-America in 1963 (Likens et al. 1977). The aim of the HBES was to investigate the effects of natural and human disturbances on nutrient cycles and fluxes through a forested catchment. The Coweeta research catchment, also in the US and started in the 1940's (Swank & Crossley Jr 1987), showed increased nutrients concentrations in streams after forestry operations. The Walker Branch Watershed Project, again in the US and initiated in 1967, focused on the processes controlling water quality as it moved through the forested catchment (Johnson & Van Hook 1989).

In the boreal zone most nutrient cycling studies have been done at the stand level (Nihlgård 1972, Mälkönen 1975, Persson 1980, Kubin 1983, Finér 1989, Helmisaari 1995, Ukonmaanaho 2001) and catchment-scale studies are more rare and were done in Canada (Hendrickson et al. 1989, Watmough & Dillon 2001) and in Fennoscandia (Rosén 1982, Kallio & Kauppi 1990). The first studies on the influence of forest clear-cutting on nutrient leaching at the catchment-level in boreal zone were carried out in Sweden, in the Kloten project in 1969-1977 (Grip 1982) and in Finland, in the Nurmes study, which started in 1978 (e.g. Ahtiainen & Huttunen 1999). More recently nutrient studies have been carried out in the context of air pollution, particularly on the effects of acidic deposition and the imputs of sulphur (S) or nitrogen (N) deposition on nutrient imbalances and leaching (e.g. Hultberg 1985, Johnson & Lindberg 1992, Löfgren & Kvarnäs 1995, Nilsson & Wiklund 1995, Wright & Tietema 1995, Mälkönen 2000). More recently the emphasis has changed to the impacts of climate change on nutrient cycling (e.g. Roos 1996, Schulze 2000). This study focuses on the fluxes and interactions of nutrients as they flow through a mature forest and the subsequent changes that occur following clear-cutting at the stand level.

## 1.2 Nutrient fluxes in boreal forest

The succession of boreal forest communities usually starts from those dominated by broad-leaved species and ends in a coniferous forest community. Typically unmanaged, mature forests are dominated by coniferous species but mixed with a broad-leaved comonent. In Finland, where the forests are mainly managed for forestry, Scots pine (*Pinus sylvestris* L.) is the main tree species on dry or nutrient poor sites while Norway spruce (*Picea abies* Karsten) dominates on more moist and fertile sites. The area of pine dominated forest stands in Finland, 13 Mha, is over two-times that of spruce, 5 Mha (Tomppo 2000a). The majority (78 %) of these conifer-dominated stands grow on upland soils and the distribution between pine and spruce dominance is almost the same as for the whole forest land (Tomppo 2000b).

Compartments of the forest ecosystem (vegetation and soil) interact with each other by releasing (output) or receiving (input) substances. The rates of these transfers vary so that there can be an increase or reduction in the size of the pools of substances in each of these two compartments. On the same site type, the fluxes of nutrients differ between forest stands depending on the stage of development (e.g. Helmisaari 1995) and structure and composition of a stand (e.g. Starr & Ukonmaanaho 1994).

The major input flux to the ecosystem is the *uptake* of carbon (C) from the atmosphere by the vegetation (Fig. 1). For other nutrients, *deposition* from the atmosphere, consisting of wet and dry components, is the most more external source in nonfertilized systems. Dry deposition can be divided into gaseous and solid fractions. The sum of dry and wet deposition measured above a forest canopy or in an open place is called *bulk deposition* and the amount of water is called *bulk precipitation*. In forested areas, deposition is greatly influenced by the canopy and understorey vegetation before reaching the forest floor. The canopy can act as a sink if nutrients are intercepted or as a source if there is *foliar leaching* from foliar tissues or *washoff* of previously deposited material on foliar surfaces. Coniferous tree canopies are particularly effective in capturing suspended dry deposition because of the large surface area ehich is present throughout year (e.g. Hyvärinen 1990). The deposition measured beneath a canopy is called *throughfall deposition*. The composition of water flowing down tree stems, *stemflow*, often differs from that of throughfall and can be a notable source of nutrients around the base of the stem. The term *total throughfall* refers to the sum of throughfall and stemflow. Throughfall under understorey vegetation is difficult to measure and rarely done so and the effect of the ground vegetation is incorporated in the quality of water collected from under the humus layer.

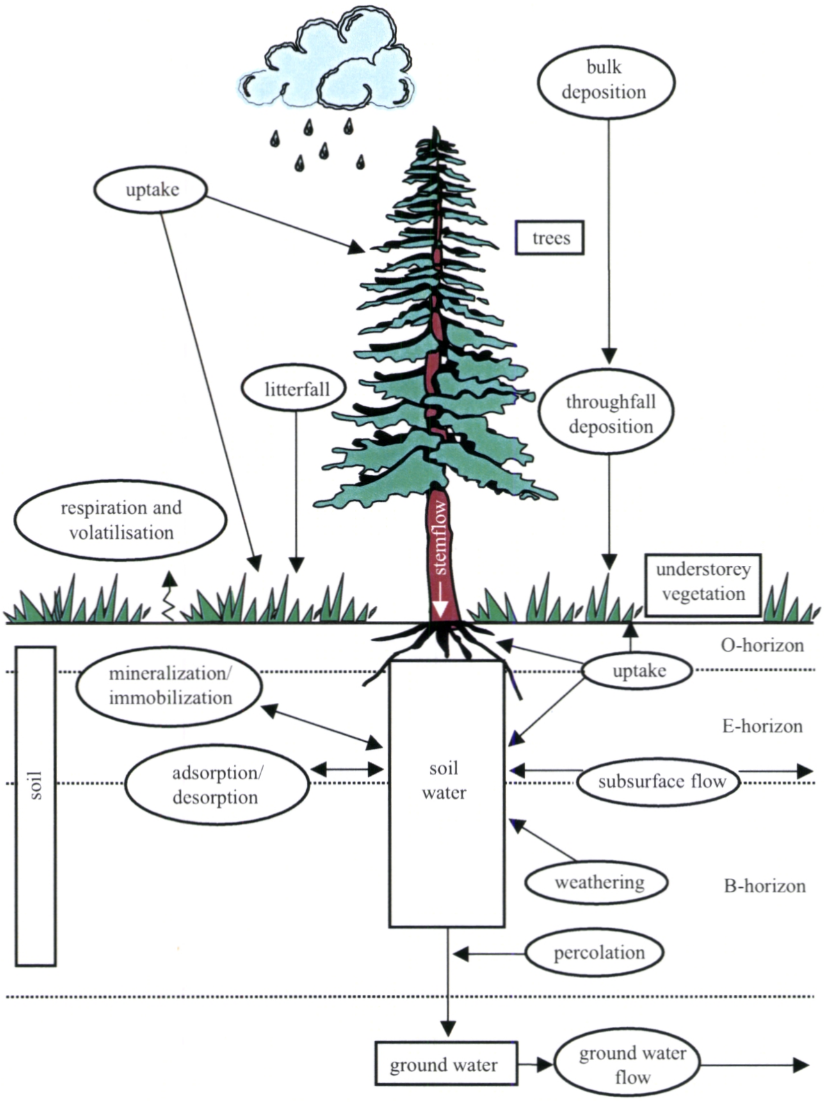

**Figure 1. Schematic presentation of the nutrient fluxes and release and retention processes in boreal forest.**

The largest ecosystem flux of nutrients is usually that of above- and belowground *litterfall* (Fig. 1). In conifer-dominated stands, litterfall is much lower than in broad-leaved stands (e.g. Mälkönen 1975, 1978). The nutrients bound in this solid flux are released through mechanical leaching, decomposition and *mineralization*, which depends on the activity of fungi, microbes and soil animals. Because of the cold, humid climate the decomposition of organic matter is relatively slow in boreal coniferous forests (e.g. Berg et al. 1993). However, the annual requirement of major nutrients [nitrogen (N), potassium (K) and phosphorus (P)] can often be met by the *retranslocation* of nutrients from the foliage back into the tree during leaf senescense and back into the developing foliage (e.g. Helmisaari 1995). Furthermore, *exudates* from living roots and microbes are an additional source of nutrients in a forest ecosystem. The *weathering* of minerals is an important source of mineral nutrients (base cations) (Fig. 1). Weathering is a slow, but continuous process in the boreal zone (e.g. Olsson & Melkerud 2000). In remote areas, it may be as important source of mineral nutrients as deposition (Starr et al. 1998).

The most important output fluxes from ecosystems are gaseous losses back to the atmosphere and *leaching* deeper into the subsoil and groundwater (Fig. 1). Some losses can also occur in solid form as erosion, but this flux is probably negligible in undisturbed, boreal forest ecosystems. Carbon is lost to atmosphere as $CO_2$ as a result of soil *respiration*. Nitrogen and S can also be released from the soil in gaseous forms. *Denitrification* is the main process releasing gaseous N to the atmosphere, but is negligible in acid, boreal forest soils (e.g. Paavolainen & Smolander 1998). The *volatilisation* of S only occurs under anaerobic conditions, such as in wetlands (Stevenson & Cole 1999) and is negligible in upland aerated forest soils. The most important dissolved output flux of nutrients from the forest ecosystem occurs via leaching losses as *deep percolation* or *subsurface flow*. The annual leaching of nutrients from undisturbed forested catchments is generally small in Finland (Kortelainen et al. 1999). However, leaching losses of nutrients can vary by day, season and year, and peak concentrations are important for biological effects in recipient waters. Such peaks occur during snowmelt, when relatively large amounts of water rapidly flush through the soil and biological retention by the forest is low (e.g. Andersson & Lepistö 1998). Small leaching losses of nutrients indicate balanced nutrient cycling or effective retention mechanisms in the soil.

## 1.3 Nutrient retention mechanisms in podzolic soils

The amount of nutrients leaching from forest soils depends on several factors. Nutrients added to the soil via precipitation, released by the decomposition of organic matter or weathering of minerals are susceptible to leaching if not taken-up by plant roots or microbes, or chemically retained in soil through a variety of mainly reversible *adsorption* processes. These adsorption processes include: ion exchange, complexation with inorganic and organic substances of low solubility, and precipitation and formation of secondary minerals.

The organic (O) horizon of podzolic soils consists of a litter and humus (mor) layer. Below the O-horizon is a weathered and eluviated (E) horizon, which has less base cations [calcium (Ca), magnesium (Mg) and potassium (K)], aluminum (Al) and iron (Fe) than the parent material as a result of podzolization (Lundström et al. 2000a). In this process organic acids are leached from the decomposing litter and exudated by roots and mycorrhizae, and attach silicate minerals. The base cations released are transported downward in ionic form with the percolating water while the Fe and Al are mainly transported as soluble organic complexes. In the illuvial (B) horizon, these organic complexes are precipitated as a result of over saturation, increasing pH conditions, or by microbial degradation of the organic complexes (Lundström et al. 2000a). The accumulation of clay, Fe and Al sesquioxides and humic substances in the B-horizon endow the soil with an ion (cations and anions) exchange capacity, much of which is pH dependent.

The effective cation exchange capacity (CEC; i.e. that at ambient pH) of the upper mineral soil (0-20 cm) in Finnish forests varies from 100 over to 500 $\mathrm{mmol_c\ kg^{-1}}$ in Finland (Tamminen 2000), and is strongly dependent on the organic matter and clay contents of the soil (Tamminen & Starr 1990). The effective CEC is pH dependent, its size increasing with increasing pH. Base cations released by decomposition and weathering are absorbed from the percolate by the cation exchange sites and $H^+$ released in exchange, thus helping to maintain nutrient cycling and soil fertility.

In the B-horizon, there are also a pH dependent positively charged sites that can adsorb phosphate ($PO_4^{3-}$) (Schnitzer 1969, Wood et al. 1984), sulphate ($SO_4^{2-}$) (Singh et al. 1980, Gustafsson & Jacks 1993, Gobran et al. 1998) and, to a lesser extent, nitrate ($NO_3^-$) anions (Stevenson & Cole 1999). The size of the anion exchange capacity (AEC) is considerably smaller than the CEC, and its pH dependency is negative. Phosphate ions under acid conditions precipitate with Al- and Fe-oxides to form insoluble Al-Fe-phosphate complexes. Sulphate adsorption, however, is at least partly pH dependent and therefore reversible (Harrison & Johnson 1992, Karltun 1995, Gobran et al. 1998).

The above soil processes affect the quality of the percolating water and, together with the factors affecting the percolation flux (evapotranspiration, soil permeability, moisture content etc.), determine the leaching losses of nutrients from the soil. The quality of soil water varies with the tension which it is held by the soil and therefore with sampling technique (e.g. Giesler et al. 1996). The soil water that is tightly adsorbed near the surface of soil colloids needs a considerable amount of energy to be released and can only be sampled in the laboratory using centrifugation. Suction (tension) lysimeters collect water from the macro- and micropores while gravity (zero-tension) lysimeter collects only water moving downwards by the gravity to groundwater. The reaction time between soil water and soil matrix is the shortest for the percolate water. If soil percolates are collected from below different pedogenic soil horizons in undisturbed ecosystems, they should reflect soil pedogenetic processes.

## 1.4 Effects of disturbance on nutrient fluxes

Abiotic and biotic disturbances to the forest affect the structure and functioning of forest ecosystems, including nutrient cycling and the leaching fluxes through the soil. In managed Fennoscandian forests, insect and fungal attacks seldom cause extensive damage and usually forests recover from such attacks. Damage caused by abiotic agents, such as snowfall, wind and wildfires are also local, and such damaged areas have remained small. For example, the area affected by wildfires in Finland is less than 1000 ha $a^{-1}$ (Sevola 2000). After a natural disturbance of the forest, the biomass of living vegetation rapidly decreases and a significant amount of dead organic matter is left behind as snags or fallen trees. The leaching and decomposition of this matter starts immediately, and the nutrients released move into the soil. The capacity of the mineral soil and surviving vegetation to retain these nutrients controls the leaching loss to ground and surface waters.

From the point of view of nutrient cycling and leaching losses, forest cuttings can be considered as an abiotic disturbance to the system. Clear-cutting and thinnings take place on 0.40-0.55 M ha every year in Finland (Sevola 2000). This is 2-3 % of the total forest area. Cutting, particularly at clear-cutting, or intense forest fires, represent an intense disturbance with considerable risks for the increased leaching loss of nutrients and decline in soil fertility. Timber removal instantly reduces the pools of nutrients in the ecosystem, especially if whole tree harvesting is carried out, and dramatically alters the nutrient flux pathways. More of the precipitation reaches the ground and evapotranspiration and nutrient uptake by the trees are greatly reduced, all increasing the risk of excessive nutrient leaching. Furthermore, the decomposition of the logging residue left on the site and the humus layer may be stimulated by the changes in moisture, light or temperature conditions and favouring the release of nutrients. The leaching and volatilisation losses of nutrients are, however, usually much smaller than that represent by the timber removal (e.g. Lepistö et al. 1995, Ahtiainen & Huttunen 1999), although they can be of similar magnitude in the case of wildfires (e.g. Lamontagne et al. 2000).

Increased mineralization of organic matter after clear-cutting of coniferous forests has been observed (e.g. Smolander et al. 1998). On fertile sites, with high potential for nitrification, clear-cutting can increase $NO_3^-$ leaching losses from below the rooting zone (Staaf & Olsson 1994), to ground water (Wiklander 1981, Kubin 1995) or to streams (Grip 1982, Wiklander et al. 1991, Ahtiainen & Huttunen 1999). Clear-cutting can also increase the leaching losses of $NH_4^+$ (Grip 1982, Ahtiainen & Huttunen 1999). Site preparation carried out after clear-cutting in order to promote regeneration can further promote mineralization. For example, Kubin (1995) showed increased $NH_4^+$ leaching after site preparation (ploughing) in a site in northern Finland.

Increased decomposition of organic matter also releases P. Increased $PO_4^{3-}$ leaching from below the logging residues and O-horizon after the clear-cutting of a deciduous forest has been observed (Qualls et al. 2000). Increased streamwater concentrations of $PO_4^{3-}$ after clear-cutting have also been observed in a number of studies (Grip 1982, Ahtiainen & Huttunen 1999, Lamontagne et al. 2000). The reason for that has probably been the changed hydrological conditions in soil e.g. increased soil moisture and high groundwater table that established anaerobic conditions preventing P retention due to reduction of Fe (Armstrong 1982). Increases in base cation leaching after clear-cutting have also been observed, particularly concerning K (e.g. Grip 1982, Kubin 1995, Lamontagne et al. 2000). This is to be expected since $K^+$ is more easily leached from logging residues (Titus & Malcolm 1992, Staaf & Olsson 1994, Stevens et al. 1995) and less strongly held by the soil exchange sites than the divalent cations $Ca^{2+}$ and $Mg^{2+}$.

Although clear-cutting has been shown to increase nutrient concentrations in streamwater, the treatments in these studies have been experimental in design and optimised to show clear-cutting effects. Thus, whole catchments or large continuous areas have been harvested and harvesting has been taken right up to the stream bank. In addition, the studies have often been carried out in the temperate zone, often under conditions of relatively high N and S deposition. Few studies have been carried out in the boreal zone, particularly in areas of low N and S deposition. More knowledge is needed on the effects of clear-cutting of boreal upland forests on nutrient leaching through the soil also in remote areas with low deposition loads. This information will be valuable to determine the possible prejudicial effects of forest final cuttings to water ecosystems and for the forestry planning to minimize the risks of nutrient leaching.

# 2 AIMS OF THE STUDY

The primary objective of this thesis was to study the impact of clear-cutting (stem only) on the fluxes of the major plant nutrients: C, N, P, S, Ca, Mg and K, through a podzolic soil in a Norway spruce dominated mixed boreal forest. By making detailed input-output budgets for the three main soil horizons in the profile both before and after clear-cutting, it was possible to evaluate the soil processes affecting the retention and release of nutrients, to determine the changes in nutrient pools in the soil, and to assess the risk of nutrient leaching to watercourses. The changes in deposition, litterfall and, after clear-cutting, logging residue inputs of nutrients to the soil were also included.

# 3 MATERIAL AND METHODS

## 3.1 Study area

The data presented in this thesis was collected from three 50 m x 50 m plots during 1993-1999. The plots are located in the Kangasvaara catchment in eastern Finland (63° 51'N, 28° 58'E, 220 m a.s.l.) (Finér et al. 1997). The forest was an old-growth mixed coniferous forest stand dominated by Norway spruce (*Picea abies* Karsten), with Scots pine (*Pinus sylvestris L.*), white and silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth) and European aspen (*Populus tremula* L.) present (Table 1 in papers III and V). In September 1996 clear-cutting was done manually at two of plots and one plot was left uncut to serve as a control. Only stems (with bark) were removed, the logging residues (branches, leaves and tree tops < 8 cm diameter) being left evenly distributed on the cut plots.

The site was classified as a *Vaccinium-Myrtillus* site type according to Finnish classification (Cajander 1949, Mikola 1982). The field layer vegetation was dominated by dwarf shrubs (*Vaccinium vitis-idaea* L. and *V. myrtillus* L.) and the bottom layer by feather mosses (*Pleurozium schreberi Brid.* and *Hylocomium splendens (Hedw.) B. S. & G.)*. The forest floor consisted of a litter and a mor humus layer with an average thickness of 3 cm. The soil, derived from sandy till, was classified as a Haplic Podzol according to the classification of FAO (1989) with a clay content of <2 %. The stone content ($m^3/m^3$) of the upper 30 cm soil layer, determined by the rod penetration method of Viro (see Tamminen & Starr 1994), was 28 %. Further characteristics of the soil are presented in Table 1 in paper II and Table 2 in papers III and IV.

The mean annual air temperature recorded at Kangasvaara during the study period averaged +0.7°C, the annual temperature sum (sum of daily mean temperature exceeding +5°C) 903°C and the annual bulk precipitation 662 mm. The long-term (1960-1990) mean annual temperature for the area is +1.4°C, temperature sum 1055°C and precipitation 540 mm (Climatological statistics… 1991). The precipitation was the highest in 1995 (759 mm) and the lowest in 1997 (517 mm). Precipitation fell mainly as snow from November to April, accounting for 39 % of the annual bulk precipitation. A detailed description of the study area is presented in Finér et al. (1997).

## 3.2 Sampling

### *3.2.1 Fluxes*

Bulk precipitation (5 collectors in summer and 3 in winter) in an open area nearby the study plots and throughfall (16 collectors in summer and 8 in winter) on the plots were collected during a four-year period before clear-cutting (1993-1996) and during a three-year period afterwards (1997-1999) using permanently open collectors (Fig. 2). Throughfall was collected by placing the collectors systematically at equal distances along each side of the plots. After clear-cutting, throughfall was collected only at the uncut plot. Stemflow was collected during 1993-1995 at two of the sample plots. For the following years, stemflow fluxes were estimated as the average of the measured years. The bulk precipitation and throughfall collectors were emptied once a week during the snow-free period and once a month during winter. On each sampling occasion the volume collected was recorded. The samples were then pooled by plot for analyses. Stemflow was collected weekly and only during the snow-free period. Full details of the bulk precipitation, throughfall and stemflow measurements are given in papers I-IV.

Soil water (percolate) was sampled at each sample plot using 27 systematically placed, zero-tension (gravity) lysimeters (Fig. 2). Nine lysimeters were installed at each of the following depths: below the organic (O) horizon, below the eluvial (E) horizon and below the upper part of the illuvial (B) horizon, correspondingly to average depths of 3, 12 and 35 cm, respectively (Fig. 2). The lysimeters below the O-horizon were of the Jordan type (Jordan 1968); a plexiglass gutter, closed at the ends with a collecting area of 420 $cm^2$. The other soil water collectors were made of a polythene plastic funnel, filled with quartz sand, having a collecting area of 299 $cm^2$ and fitted to a 2 L sample collection bottle. The lysimeters were emptied on the same day as the precipitation and throughfall collectors when the soil was not frozen (generally from April to October), and they were analysed separately. Care was taken during the clear-cutting operation not to disturb the lysimeter installations. A more detailed description of the soil water sampling is given in papers I-IV.

Canopy litterfall (paper V) was collected from September 1992 to September 1996 at each plot using 16 funnel traps (Fig. 2) during the snow-free period and 8 traps during winter (collecting area 0.5 $m^2$; collecting height 1.3 m). The traps were emptied once a week during summer and once a month during winter. The collected litterfall was separated into three fractions: branches and twigs, leaves and the rest. The dry weight of each fraction was recorded before being analyzed for total elemental concentrations.

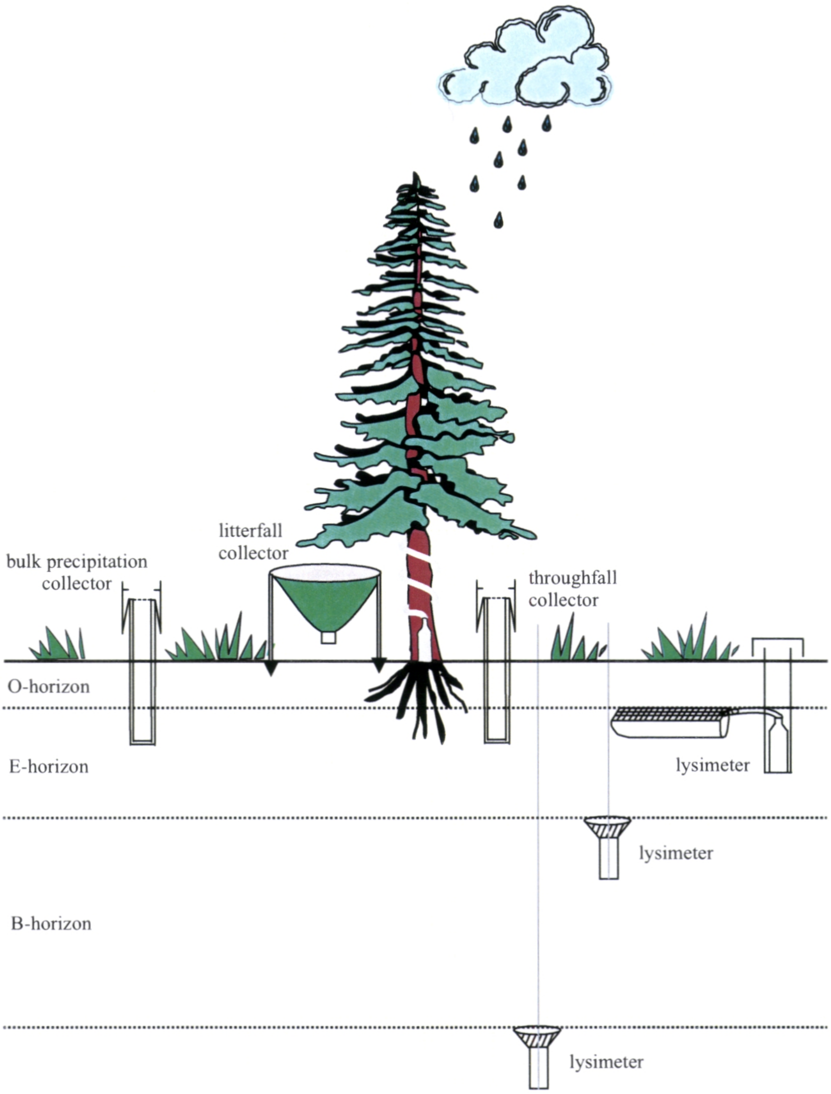

**Figure 2. Schematic presentation of the collectors at Kangasvaara.**

### *3.2.2 Pools in soil and tree biomass*

Volumetric samples of soil from the O-, E- and B-horizons were taken in July 1993. Twenty rectangular cores (10 cm x 10 cm) of the O-horizon and cylindrical cores (diameter 35 mm) of the E- and B-horizons were systematically taken from each plot. The thicknesses of the O-, E- and B-horizons averaged over the three plots 3.1, 6.8 and 13.9 cm, respectively (Table 1 in paper II).

Breast height diameter (dbh), tree high and crown length of all living trees (height > 1.3 m) on the plots were measured in September 1992 and again in July-

August 1996. Harvesting of sample trees on the plots for derivation of allometric biomass functions (described in detail by Finér 1989) for stemwood, stem bark, foliage and brances, was carried out in August-September 1996 in connection with the clear-cutting. Nine Scots pine, nine Norway spruce and seven birch trees were selected, based on the distribution on dbh values, and carefully felled. Both deciduous tree species (birch and aspen) were represented by birch. A detailed description of the tree sampling for the biomass functions is presented in paper V.

## 3.3 Laboratory analyses and calculations

### *3.3.1 Water samples*

All water samples were kept cold and transported to laboratory the day after sampling. First, pH was measured (Radiometer PHM92 pH-meter), and thereafter the samples were filtered through a Schleiche & Schuell no. 589[1] filter; a Schleiche & Schuell no. GF 52 glass wool filter in the case of soil percolate samples. All the filtrates were stored in a freezer (-18 °C) until further analysis. Nitrate, $SO_4^{2-}$ and $PO_4^{3-}$ concentrations were determined by ion chromatography (Dionex series 100 or 300 or 500), and $NH_4^+$ and dissolved total N (DTN) were determined by flow injection analysis (Tecator FIA-Star 5020). Dissolved organic N (DON) was calculated by subtracting dissolved inorganic N (DIN; sum of $NH_4^+$-N and $NO_3^-$-N) concentrations from DTN concentrations. Dissolved organic carbon (DOC) was measured with a Shimadzu TOC-5000 analyser. Concentrations of cations, $Ca^{2+}$, $Mg^{2+}$, $K^+$ and sodium ($Na^+$), were determined by flame atomic absorption spectrophotometry (FAAS; Perkin-Elmer 5000). Total S and P concentrations in a representative number of samples were measured with an ICP emission spectrometer (ARL 3580 in 1993-1996 and TJA Iris Advantage thereafter). These data were used to derive transfer functions to estimate total S and P concentrations from $SO_4^{2-}$ and $PO_4^{3-}$ concentrations (Table 3 in paper IV).

The additional dry deposition of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ captured by the canopy was estimated using the ratio of $Na^+$ in bulk deposition to $Na^+$ in total throughfall minus bulk deposition. This is based on the assumption that $Na^+$ is not exchanged in canopy (Ulrich 1983, Draaijers et al. 1997), and that the dry deposition characteristics of $Na^+$ are the same for $Ca^{2+}$, $Mg^{2+}$ and $K^+$.

Monthly and annual deposition and fluxes in the soil for each plot were calculated from mean concentrations (weighted by the volume contribution of individual collectors), the volume of water collected and the total collection area of all collectors. The difference in solute concentrations between the uncut and cut plots was tested by the MULTTEST procedure with bootstrap p-value adjustment

[SAS for Windows 6.12 (SAS® 1997)]. The statistical testing was done on annual and seasonal (spring, summer, autumn) values before and after clear-cutting. Monthly, logarithm transformed concentration values were used. The periods before and after clear-cutting were analysed separately and if no differences were found between plots before clear-cutting, differences thereafter are considered to be the result of clear-cutting. The spring season is considered to be from snowmelt (first sampling in March in 1994) through to the end of May, summer June through August, and autumn September through October (November in 1994 and 1999 and December in 1996). Correlation analyses were carried out on monthly plot data. More detailed descriptions of numerical handling and statistical analysis are presented in the papers I-IV.

### *3.3.2 Tree biomass and soil samples*

The soil, biomass and litterfall samples were first dried at 60°C. Subsequently, the mineral soil samples were sieved to retain the < 2 mm fraction while the organic (humus layer, biomass and litterfall) samples were milled into a fine powder. Soil total C and N concentrations were determined using a LECO CNH-600 analyser. Nitrogen concentrations in the tree biomass and above-ground litterfall samples were determined using the Kjehldahl method (Halonen et al. 1983) and C calculated assuming it to be 52 % of dry weight. Total P, S, Ca, Mg and K concentrations in the litterfall and O-horizon samples were determined from a wet digestion ($HNO_3+H_2O_2$) and in the mineral soil samples from an *Aqua regia* ($HNO_3+HCl$) digestion by ICP emission spectrometry (TJA Iris Advantage). Effective cation exchange capacity (CEC) of the soil samples was calculated as the sum of 0.1 M $BaCl_2$ extractable cations and the concentrations of plant available nutrients from a $NH_4Ac$ extraction (pH 4.65) by FAAS (Perkin Elmer 5000). Iron and Al fractionations into organic, amorphous and crystalline forms were according to van Reeuwijk (1995).

The pools of nutrients in each soil horizon and plot were calculated from the concentrations, bulk density and the horizon thickness values. Mineral soil values were subsequently corrected for gravel (2-20 mm) and stone (> 20 mm) contents. The C and N pools in living roots (sampling and calculations are described in detail in paper V) were subtracted from soil pools of these elements.

For the nutrient pools in the various tree biomass compartments, the samples taken for biomass and wood density determinations were also analysed for nutrient concentrations. These samples were analysed in the same way as described for the litterfall and O-horizon samples. The dry weight biomass of branches and foliage stripped from the harvested stems and left on site were estimated using relationships established between these biomass fractions and stemwood in the 1996 data described above. The mean content in branches and foliage were then used to calculate logging residue inputs to the forest floor for each nutrient. The calculations are described in more detail in paper V.

# 4 RESULTS AND DISCUSSION

## 4.1 Fluxes prior to clear-cutting

### *4.1.1 Water*

The annual bulk precipitation during 1993-1999 averaged 493 mm of which 92 % reached the forest floor as total throughfall (papers II and IV). The bulk precipitation value was, on average, only 75 % of the summer precipitation value recorded using an automatic recording weather station located in the same open area as bulk precipitation collectors at Kangasvaara (Finér et al. 1997). Evaporation losses from the bulk precipitation collectors most probably accounted for thee lower precipitation values. Throughfall was probably closer to the true amount of throughfall since the collectors were shaded by canopy and wind speeds lower and thus evaporative losses were proportionally smaller than for bulk precipitation. The amount of stemflow was small, accounting for only 1.5 % of total throughfall (paper II), typical for Norway spruce dominated forests (Seppänen 1964, Päivänen 1966, Nihlgård 1970, Starr 1995).

Before clear-cutting 61, 33 and 11 % of the total annual throughfall passed through the O-, E- and B-horizons on average (Fig. 1 in paper I). The water flux was the greatest at the time of snowmelt. During the growing season, percolation from below the B-horizon only occurred after heavy rainfall events, and then was small.

The measurement of soil water percolation is difficult and there are uncertainties using zero-tension lysimeters (e.g. Giesler et al. 1996). Installation can disturb roots and pore connectivity, thus affecting the flow of water. Furthermore, the boundary between the intact soil and quartz sand in the funnel of the lysimeter may influence the movement of percolating water. Sand accounted for 41 and 39 % of the < 20 mm soil in the E- and B-horizons, respectively. By using quartz sand with a particle size of 0.8 to 1.2 mm to fill the funnel, it was expected to minimise the effect of a change in soil texture upon the movement of water (discussed further in paper II). The influence of cutting the roots in the soil core above the funnel lysimeters installed under the E- and B-horizons was probably short lived, as the roots quickly grew back and the vegetation above the lysimeters was indistinguishable from that elsewhere. Nevertheless, there was large variation in the weekly amounts of water collected by the lysimeters (cv < 175 %). This is to be expected because of spatial heterogeneity of vegetation and soil physical properties. The variation between lysimeters was the greatest during snowmelt when there is the added variability of frozen and unfrozen soil. It is probable that some of the melting water flowed laterally out of the upper soil before reaching the B-horizon. Because of the variability and uncertainty in the lysimeter fluxes, physically based modelled soil water fluxes could have been used. Both SOIL (Jansson 1991) and WATBAL (Starr 1999) have been applied but non-measured values for some of the required parameters were used and the results were not in agreement or always logical. Therefore, the zero-tension lysimeter values have been used and are considered reliable enough in the comparison of the effects of the clear-cutting treatment on the amounts of water percolating through the soil.

### *4.1.2 Carbon*

The total (i.e. above- and below-ground) tree biomass annual increment was 2084 kg C ha$^{-1}$ and that of the understorey vegetation 466 kg C ha$^{-1}$ (paper V). The annual flux of C to the forest floor associated with the above-ground tree litterfall was more than 13-times larger than the DOC flux in total throughfall (Fig. 3). These amounts are similar to the litterfall C fluxes reported by Nihlgård (1972) and throughfall C fluxes reported by Currie et al. (1996) and Michalzik & Matzner (1999) for coniferous forests. The flux of DOC leaving the O-horizon was relatively small; < 13 % of litterfall and throughfall C inputs (Fig. 3). Assuming the O-horizon is in steady-state, much of the litterfall flux of C must be mineralised to $CO_2$ and returned to atmosphere. The mineral soil effectively retained the percolating DOC flux from the O-horizon and the annual flux from below the B-horizon averaged 3 kg ha$^{-1}$a$^{-1}$ (Fig. 3). The magnitude of this flux was similar to that recorded for stream water from catchments with podzolic soils reported by Tate & Meyer (1983) and Guggenberger & Zech (1993).

The reduced flux of DOC from the mineral soil could be due to biological and chemical processes. The biodegradability of DOC is in soil water, however, usually quite low due to the abundance of hydrophobic and hydrophilic acids that are resistant to microbial degradation (Qualls & Haines 1992). Humic substances are known to be strongly adsorbed by Fe and Al sesquioxides (McDowell & Wood 1984, Jardine et al. 1989, Qualls & Haines 1992), especially amorphous Al hydroxides (Kaiser & Zech 1998), leading to organic matter enrichment of the upper B-horizon (Lundström et al. 2000b). Amorphous Fe and Al hydroxide concentrations were enriched in the upper part of the B-horizon (Table 2 in paper III), as is typical for podzolic soils, and the greatest amount of retention could therefore have been expected to occur there. However, retention of DOC by the E-horizon was also evident (Fig. 1 in paper III). Biological immobilization to microbial metabolism or root uptake of DOC could occur, although they had minor importance in previous studies (Neff & Asner 2001). Nevertheless, the pool of C was higher in the B-horizon than in the E-horizon (Fig. 3).

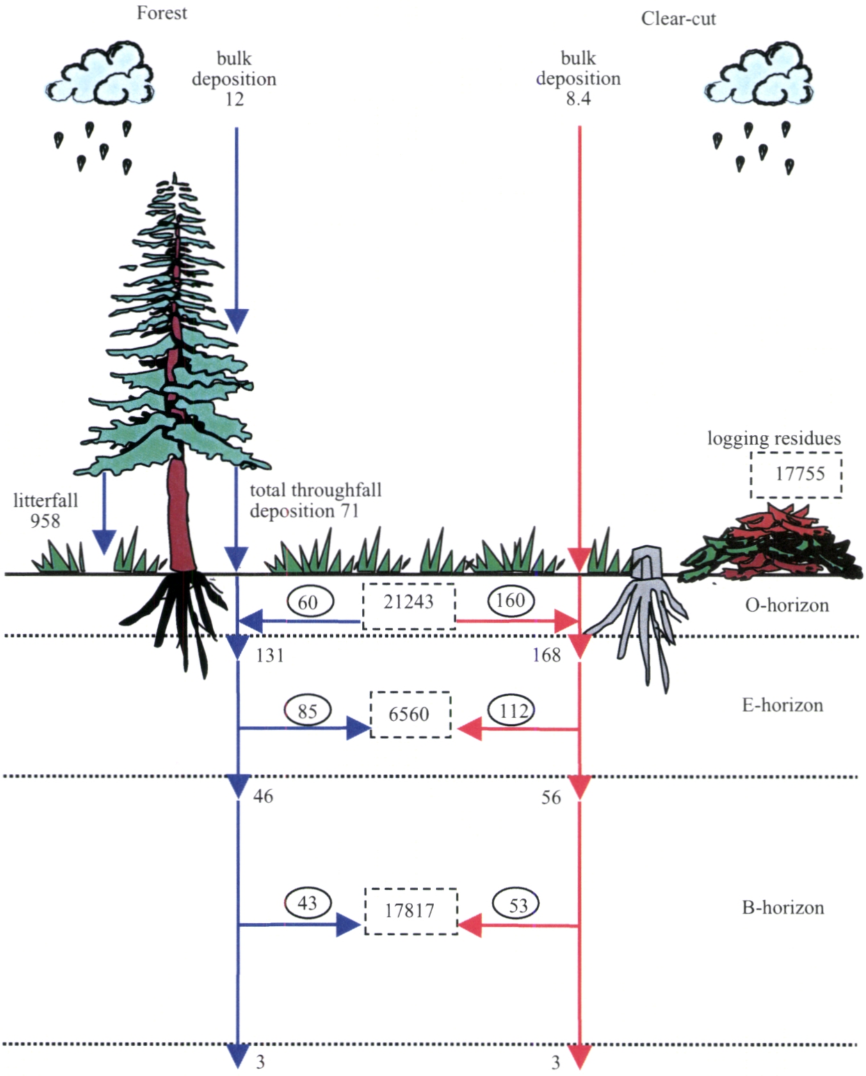

**Figure 3. Fluxes of C (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. The dashed boxes represent pools (kg ha$^{-1}$) in logging residues and soil horizons. The circled values represent leaching from or retention to each soil horizon.**

The total amount of C in the 0-60 cm mineral soil layer, 31 614 kg ha$^{-1}$, was higher than that in the O-horizon, 21 243 kg ha$^{-1}$ (paper V). The upper part of the B-horizon (7-21 cm depth) accounted for 56 % of the 0-60 cm mineral soil C pool (Fig. 3). According to other Finnish studies the largest amount, 68 %, of soil C is estimated to be in the mineral soil to 1 m depth and only 28 % is in the organic layer, the rest (4 %) is below 1 m (Liski & Westman 1997). Liski & Westman (1997) consider that there is no current accumulation of C in forest soils and that the C pool is in steady-state. At Kangasvaara, the main pool of C in the combined stand-soil system was in the living vegetation (> 60 %; Fig. 1 in paper V), as has been also shown in *Vaccinium, Myrtillus* and *Oxalis-Myrtillus* type sites in southern Finland (Liski & Westman 1995).

### *4.1.3 Nitrogen*

The annual uptake of N by the trees at Kangasvaara was 16.1 kg ha$^{-1}$ a$^{-1}$ (paper V), which falls within the range for coniferous boreal forests reported by Nihlgård (1972), Mälkönen (1975), Finér (1991) and Helmisaari (1995). Uptake by the understorey vegetation was not measured in this study, but it has been estimated to be as much as 8.9 kg ha$^{-1}$ a$^{-1}$ (paper V). The annual N flux to the forest floor via above-ground tree litterfall (Fig. 4) could supply 75 % of the tree uptake, which was in the upper end of the range of values reported by Nihlgård (1972), Mälkönen (1975) and Helmisaari (1995). The annual bulk deposition input of total N (Fig. 4) was small in comparison to levels elsewhere in Europe (Hjellbrekke 2000). Some of the inorganic N input was taken up in the canopy and a smaller amount of DON was released to throughfall (paper I). The net retention by the tree canopy was 2.4 kg ha$^{-1}$ a$^{-1}$ (paper I), which corresponds to 15 % of the annual tree uptake of N. Likens et al. (1977) and Helmisaari (1995) concluded that N deposition supplies about 10 % of the annual N demand.

In boreal coniferous forests receiving low N deposition loads, the uptake of N is principally controlled by the rate of organic matter decomposition and the release of inorganic N compounds. Nitrogen fixation of atmospheric $NO_2$ is negligible, except where there is a significant proportion of N fixing species such as *Alnus* in the stand. However, recent research indicates that roots and mycorrhizas are able to take up N in organic forms (Näsholm et al. 1998). In the study by Persson & Wirén (1995), estimated annual N mineralization rates in boreal coniferous soils ranged from 35 to 105 kg ha$^{-1}$ and averaged 1.4 % of the soil N pool in the top 50 cm layer. Assuming 1.4 % of the soil pool of N at Kangasvaara is mineralised annually, this would produce 32 kg N ha$^{-1}$, which exceeds the uptake by trees and understorey vegetation.

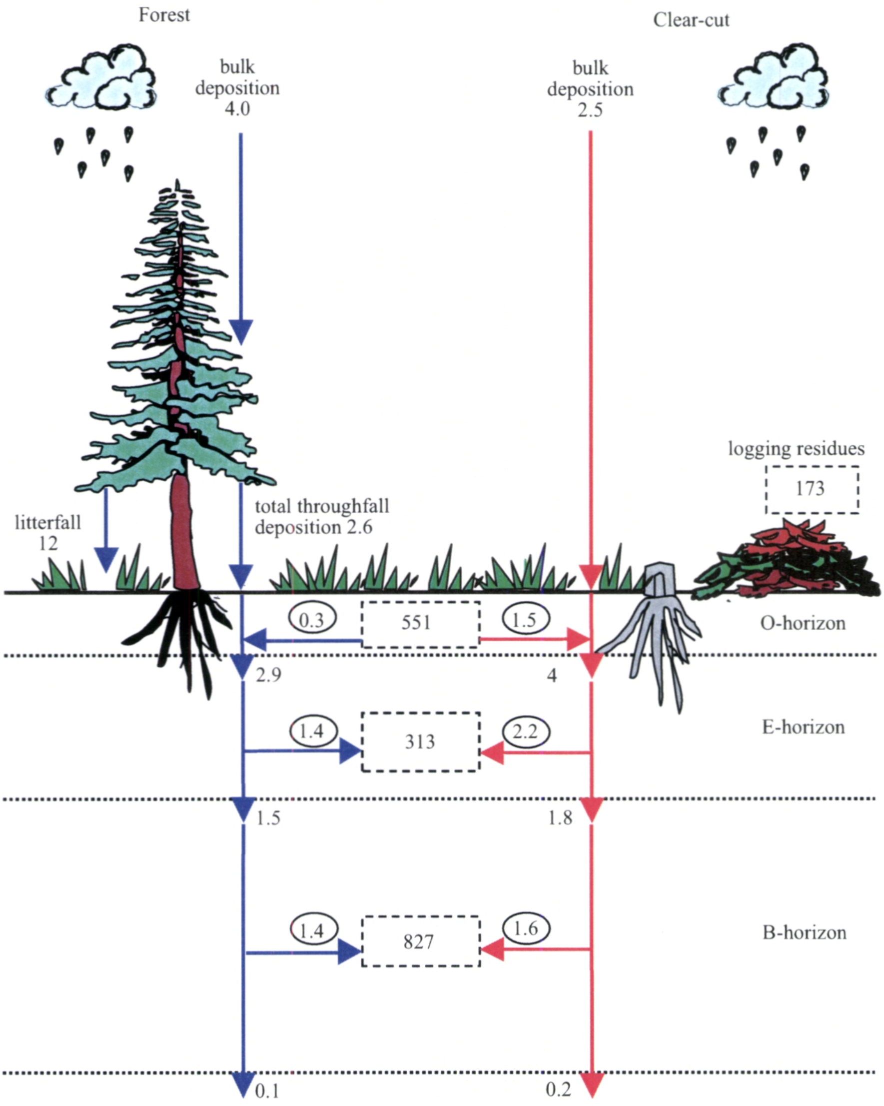

**Figure 4. Fluxes of total N (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. See explanation in Fig. 3.**

The tight cycling of N in boreal coniferous forest is reflected in the minor leaching losses of inorganic N (Tamm 1991). This was also the case at Kangasvaara. The annual average flux of DTN from below the O-horizon was small (Fig. 4) and mainly organic (> 80%; paper I), which is typical for both coniferous (Sollins & McCorison 1981, Fahey et al. 1985, Smolander et al. 2001) and deciduous (Qualls et al. 1991) forests. Most (84 %) of the total annual throughfall deposition of $NO_3^-$ was retained in the O-horizon (Fig. 1 in paper I). The figure was less for $NH_4^+$ and there was an increase in the leaching of organic N. The C/N ratio in the O-horizon was > 35 (paper III), which indicates a low nitrification capacity (McNulty et al. 1991, Gundersen et al. 1998). The leaching of DIN from the O-horizon occurred mainly during snowmelt, when the biological uptake is small and the water flux high (Fig. 3 in paper I). This snowmelt peak in N leaching fluxes from the organic layer of forest soils has been shown in other studies (e.g. Helmisaari & Mälkönen 1989, Pardo et al. 1995). However, much of the percolating DIN and DON was subsequently retained in the mineral soil (Fig. 4), presumably due to uptake by roots and micro-organisms, and in the case of $NH_4^+$, also by cation exchange processes.

Most of leaching of N from the O-horizon at Kangasvaara was in the form of organic N (Fig. 1 in paper I). The importance of organic N leaching has been shown in other studies (e.g. Fahey et al. 1985, Qualls et al. 1991, Currie et al. 1996, Smolander et al. 2001). The low DTN flux from below the B-horizon (Fig. 4), particularly of DIN (Fig. 1 in paper I), clearly shows that N deposition at Kangasvaara is not resulting in the N saturation of the system. In areas with the high N deposition and leaching of $NO_3^-$, ecosystems are described as being N saturated (Tietema et al. 1993, Bredemeier et al. 1998, Gundersen et al. 1998). The amount of DTN in stream runoff from catchments similar to Kangasvaara is also typically small (< 3 kg $ha^{-1}$ $a^{-1}$; Grip 1982, Rosén 1982, Ahtiainen 1992). Nitrogen losses through volatilisation and denitrification are likely to be very small in such freely draining, acid soils, although they can be induced by increased inputs of inorganic N through deposition or fertilization (e.g. Martikainen 1985).

The N in biomass and soil is initially derived from the atmosphere. At Kangasvaara the N pool in the living vegetation compartment was 476 kg $ha^{-1}$ (Table 2 in V) and was more in the soil compartment (1691 kg $ha^{-1}$ down to 25 cm), a third of which was in the O-horizon (Fig. 4). These values fall within the range of values given by Tamminen (1991) for southern Finland. Virtually all soil N is associated with organic matter, even in the mineral soil, and the amount in inorganic forms is < 0.1 % (e.g. Rosén 1982, Kubin 1983).

### *4.1.4 Phosphorus*

The annual deposition load of $PO_4^{3-}$ is also small at Kangasvaara (paper IV), as is generally the case in the boreal zone (e.g. Nihlgård 1972, Helmisaari 1995). Organic and inorganic P was leached from the canopy, but fluxes of P in total throughfall were very small (Fig. 5). Litterfall was a much greater source of P to the forest floor, the above-ground tree litterfall flux of P being almost six-times greater than that of the throughfall flux (Fig. 5). Tree uptake of P (2-9 kg $ha^{-1}$ $a^{-1}$; Nihlgård 1972, Mälkönen 1975, Finér 1991, Helmisaari 1995) is therefore controlled by the mineralization of organic matter (Stevenson & Cole 1999) and studies have shown that litterfall can satisfy up to 60 % of the annual uptake of P. The decomposition of organic matter at Kangasvaara was probably not limited by the availability of P as the C/P ratio of the O-horizon was < 2000 (Table 2 in paper IV), a value which has been found to be the threshold value for P mineralization (Berg & Staaf 1980).

Some leaching of organic and inorganic P from the O-horizon at Kangasvaara took place (paper IV). The phosphate is likely to be dominated by $H_2PO_4^-$ ions under such acid conditions (Stevenson & Cole 1999). Just over half of the

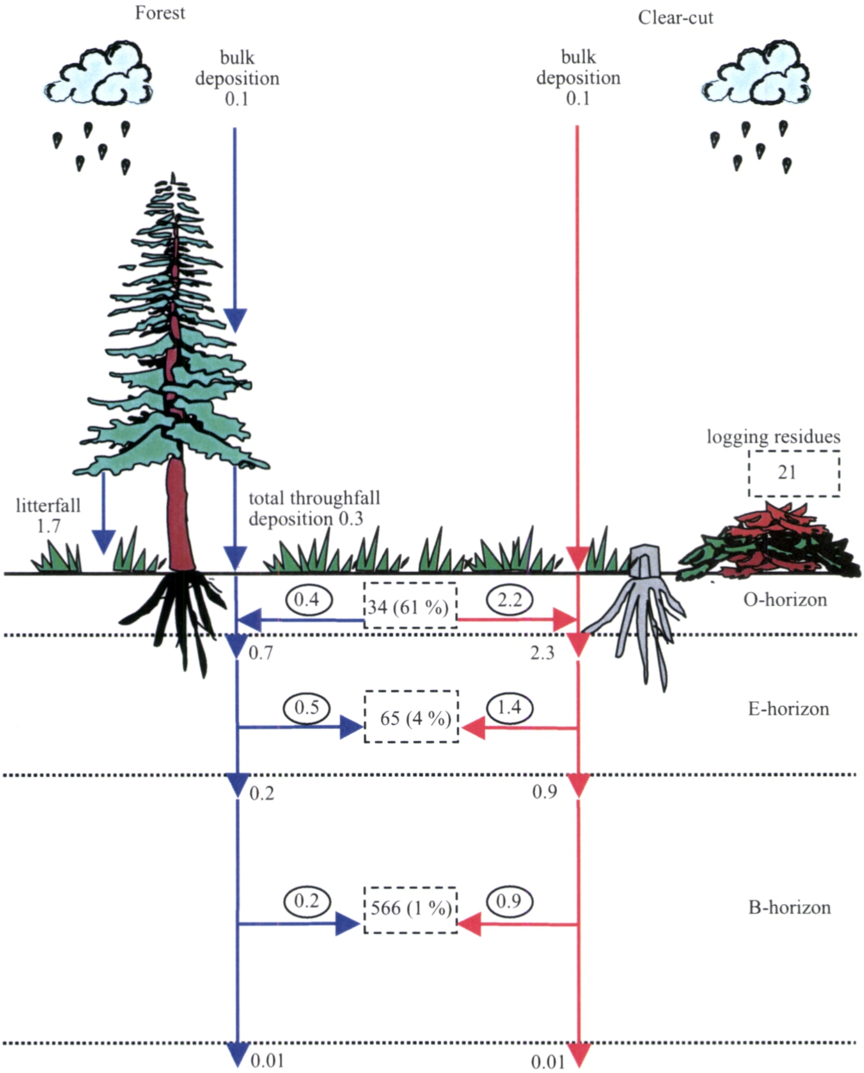

**Figure 5. Fluxes of total P (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. Percentages represent available pool of P in soil horizons. See further explanation in Fig. 3.**

annual flux of total P to the mineral soil was inorganic P (paper IV). Other studies have found the flux of P from the O-horizon to be dominated by organic P (Yavitt & Fahey 1986, Qualls et al. 2000). The retention of P by the mineral soil was effective (Fig. 5), which is typical for podzolic soils, and the pool of P in the soil is usually larger than that in the biomass pool (Kubin 1983, Yanai 1992). Phosphate ions, as well as the organic P compounds, co-precipitate with soluble Al and Fe oxides to form Al-Fe-phosphate complexes, which are highly insoluble and poor sources of available P for plants (Stevenson & Cole 1999), or strongly adsorb to oxide surfaces (Schnitzer 1969, Wood et al. 1984, Stevenson & Cole 1999). Precipitation is the dominant process in many soils (Johnson & Cole 1980), although the adsorption of phosphate ions is important in podzolic soils because of the enrichment of Al and Fe oxides in the B-horizon (Singh et al. 1980, Gustafsson & Jacks 1993, Gobran et al. 1998). The P adsorption capacity of soils depends inter alia on the amount of Al and Fe oxides, which were abundant in the B-horizon at Kangasvaara (Table 2 in paper III). The soil pool of total P was therefore much higher in the B-horizon than in the E-horizon and the flux of P from below the B-horizon small (Fig. 5). There was little leaching of organic P (paper IV). The efficient retention of P in podzolic soils explains why P concentrations are low in runoff from boreal, forested catchments with runoff losses < 1 kg ha$^{-1}$ a$^{-1}$ (Grip 1982, Rosén 1982).

### *4.1.5 Sulphur*

The main origin of S in soil and vegetation is the weathering of pyrite ($FeS_2$) found in igneous rocks (Stevenson & Cole 1999). Sulphur is an essential plant nutrient, being primarily taken up from soil in the form of $SO_4^{2-}$ ions. Some uptake of S occurs also in gaseous form, $SO_2$, through the stomata and by sorption to leaf and bark surfaces (Murphy & Sigmon 1990), but after oxidation to $SO_4^{2-}$, it is mainly washed-off with rain (Lindberg et al. 1986, Lindberg & Lovett 1992, Draaijers et al. 1997). The bulk deposition input of $SO_4^{2-}$ at Kangasvaara was 2.2 kg S ha$^{-1}$ annually, the total throughfall deposition being some two-fold (Fig. 1 in paper II). These values are at the lower end of the range of values reported for coniferous forests in North America and north-western Europe (Helmisaari & Mälkönen 1989, Hultberg & Grennfelt 1992, Lindberg & Lovett 1992, Lindroos et al. 2000, Ukonmaanaho 2001). Sulphur is not leached from cell tissues (Lindberg et al. 1986), and the addition of sulphate to throughfall is the washoff of dry deposition collected by the canopy between rainfall events. The amount of $SO_4^{2-}$ enrichment in throughfall at Kangasvaara was within the range reported for Norway spruce dominated stands (Bredemeier 1988, Hultberg & Grennfelt 1992).

The O-horizon retained over 50 % of the annual total throughfall deposition of $SO_4^{2-}$ (Table 3 in paper II). Plant uptake and microbial immobilization probably account for most of this retention (Fitzgerald et al. 1988, David et al. 1991, Strickland et al. 1996). The C/S ratio in the O-horizon was > 1500 (paper IV), which is greater the threshold value reported necessary for net mineralization and leaching of S (Berg & Staaf 1980). Studies indicate that the annual uptake of S by coniferous forests varies from 1.5 to 6.0 kg ha$^{-1}$ (Nihlgård 1972, Finér 1991). On the basis of these values, the annual S uptake at Kangasvaara could be largely satisfied by deposition. The annual S flux as above-ground tree litterfall was small, 17-67 % of uptake (Fig. 6). Previous studies have shown that over 60 % of the annual S uptake by trees can be met by litterfall (Nihlgård 1972, Finér 1991).

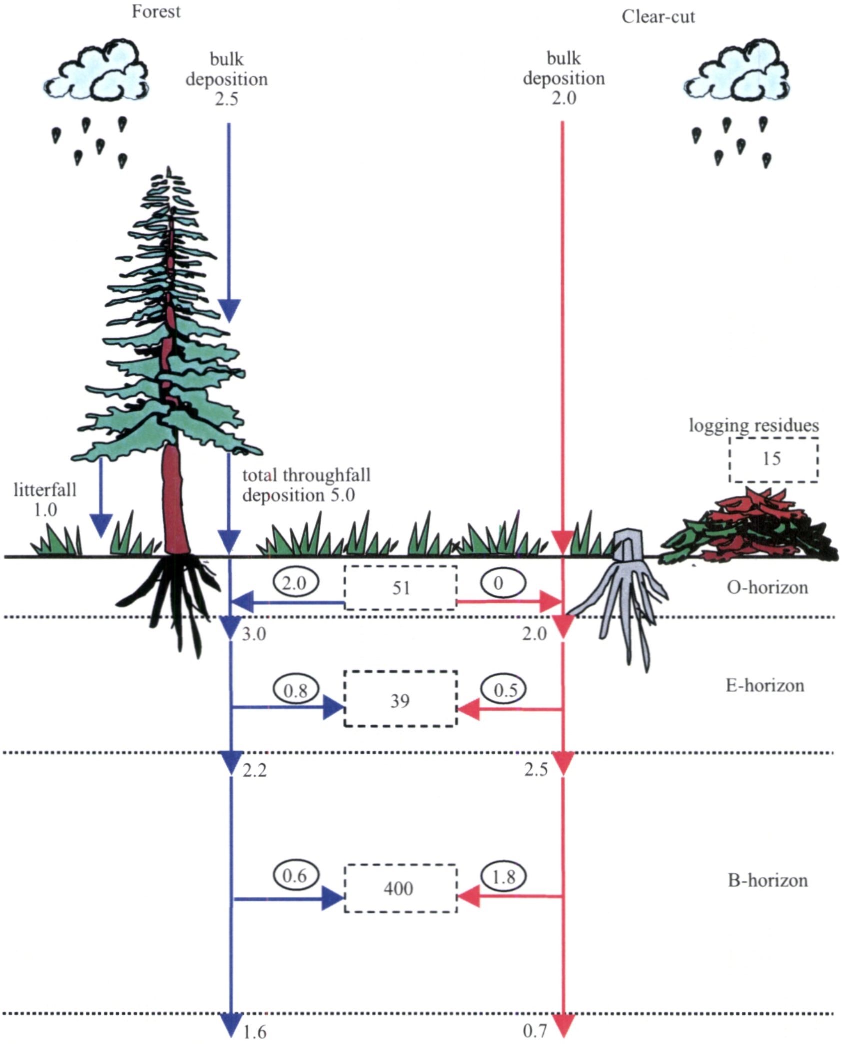

**Figure 6. Fluxes of total S (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. See further explanation in Fig. 3.**

Most (70 %) of the annual soil water flux of total S to the mineral soil was inorganic ($SO_4^{2-}$-S, paper IV) and effectively retained by the B-horizon (Table 3 in paper II). The pool of S in the B-horizon was much larger than that in the E-horizon (Fig. 6). The adsorption of $SO_4^{2-}$ by soil, as with $PO_4^{3-}$, is strongly related to the content of Fe and Al oxides in the B-horizon of podzolic soils and is an important acid neutralizing process (Johnson 1980, Singh et al. 1980, Johnson & Todd 1983, Singh 1984, Gustafsson & Jacks 1993, Karltun & Gustafsson 1993). The S flux from below the B-horizon was smaller than that of the total throughfall flux, indicating accumulation of S in the soil (Fig. 6). Some 48 % of the annual S percolating flux from below the B-horizon was organic (paper IV). In areas receiving low levels of S deposition, such as Kangasvaara, the leaching of $SO_4^{2-}$ to streams is < 5 kg ha$^{-1}$ a$^{-1}$ (e.g. Rosén 1982).

### *4.1.6 Base cations*

The input of base cations in bulk precipitation (Fig. 7-9) was in the middle of the range of values reported for Finland by Hjellbrekke (2000). Significant amounts of base cations were leached from the canopy (Fig. 7-9), as shown in many other studies e.g. Nihlgård (1972) and Helmisaari (1995). The small contribution of estimated dry deposition (see chapter 3.3.1) to total throughfall, indicates that canopy leaching was the main source of $Ca^{2+}$ (56 %), $Mg^{2+}$ (75 %) and $K^+$ (79 %) in throughfall, the rest originating from wet and dry deposition (Fig. 1 in paper II). However, there was a close 1:1 relationship between the sum of base cations leached from the canopy and the flux of $SO_4^{2-}$ in total throughfall (Fig. 2 in paper II). This indicates that the leaching of base cations from the canopy is strongly controlled by the amount of $SO_4^{2-}$ deposition (Westling et al. 1995). The $H^+$ ions associated with $SO_4^{2-}$ in deposition are exchanged for the base cations in the foliage and it is therefore an important acid neutralizing process (Ulrich 1983, Bredemeier 1988, Westling et al. 1995, Draaijers et al. 1997). Total throughfall was the main pathway by which $K^+$ was returned to the forest floor (Fig. 9) and litterfall the nain pathway for $Ca^{2+}$ and $Mg^{2+}$ (Fig. 7 and 8). However, the litterfall flux of $K^+$ may be underestimated since the litter collected in the litterfall traps is subject to leaching by the passage of throughfall through the collector (Ukonmaanaho & Starr 2001).

Mineral weathering is an important process in the soil supplying base cations for uptake or leaching. Studies in Finland give estimates of $Ca^{2+}$+$Mg^{2+}$ weathering release in forest soils from 3.5 to 5 kg ha$^{-1}$ a$^{-1}$ (Johansson & Tarvainen 1997, Starr et al. 1998). The ratios of base cations to N in annual uptake values for coniferous trees have been shown to be 0.34-0.73, 0.061-0.19 and 0.11-0.64 for $Ca^{2+}$, $Mg^{2+}$ and $K^+$, respectively (Nihlgård 1972, Mälkönen 1975, Finér 1989, Helmisaari 1995).

Applying these ratios to Kangasvaara give annual base cations uptake values of 5.5 - 11.7 kg ha$^{-1}$ for $Ca^{2+}$, 1.0 - 3.1 for $Mg^{2+}$ and 1.7 - 10.3 kg ha$^{-1}$ for $K^+$. On this basis, litterfall could supply all of the annual $Ca^{2+}$ uptake at Kangasvaara and more than 65 % and 39 % of that for $Mg^{2+}$ and $K^+$, respectively. Those figures are in accordance with previous studies (Nihlgård 1972, Finér 1989, Helmisaari 1995). Throughfall deposition of $K^+$, which was mainly derived from the canopy (foliar leaching), accounted for more than 61 % of the annual tree uptake value. Corresponding percentages for $Ca^{2+}$ and $Mg^{2+}$ were 23-49 and 23-70 %, respectively. Previous studies indicate that throughfall supplies over 30 % of the annual uptake of $Ca^{2+}$ or $K^+$ and over 15 % of that for $Mg^{2+}$ (Nihlgård 1972, Helmisaari 1995).

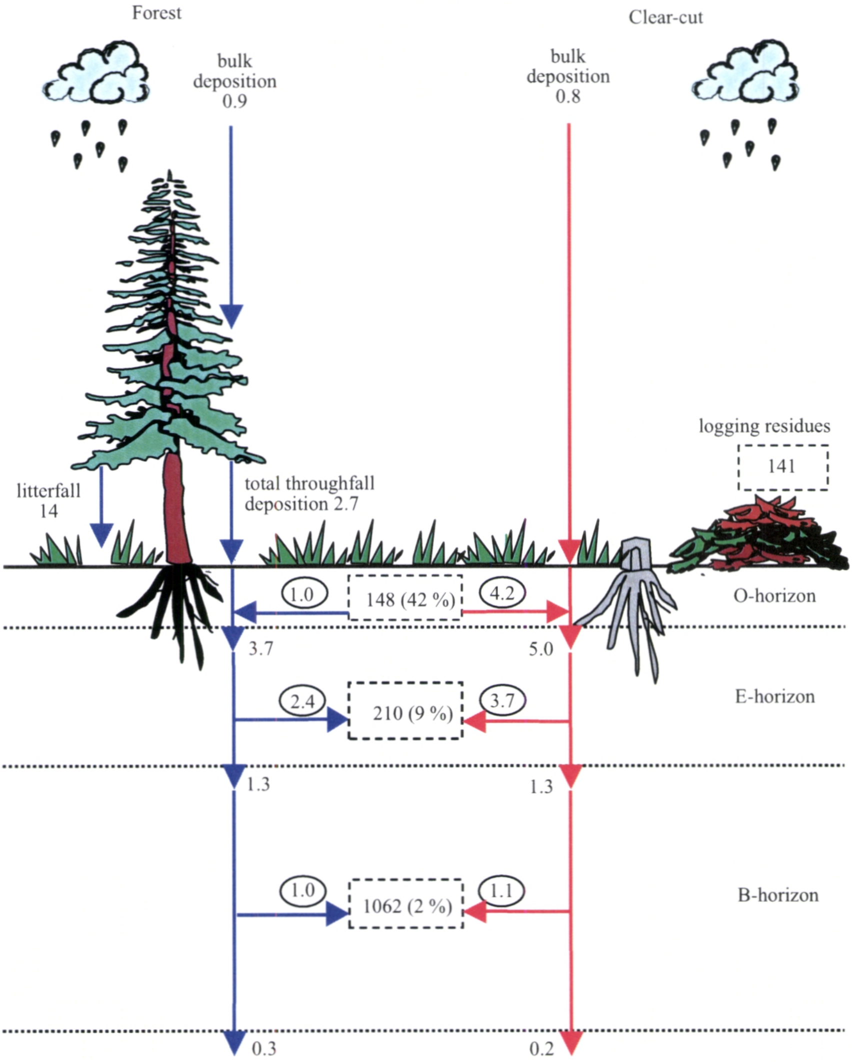

**Figure 7. Fluxes of Ca (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. See further explanation in Fig. 5.**

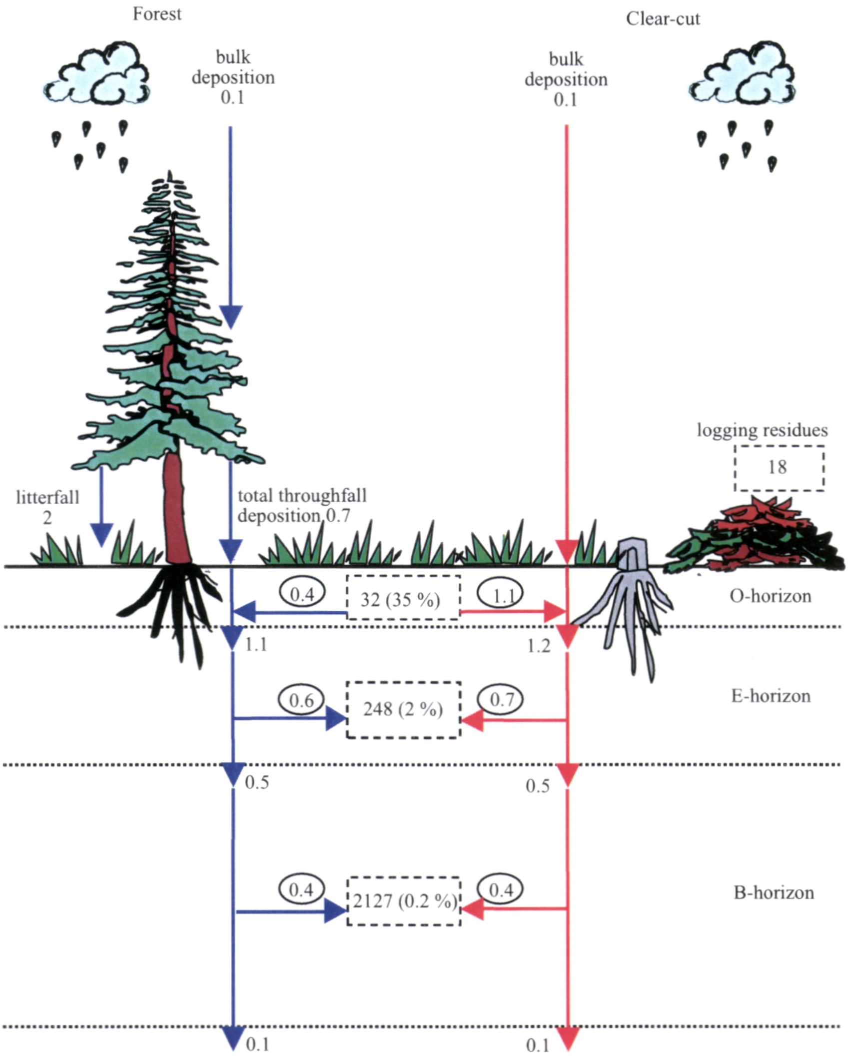

**Figure 8. Fluxes of Mg (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. See further explanation in Fig. 5.**

Leaching outputs of $Ca^{2+}$, $Mg^{2+}$ and $K^{+}$ from the O-horizon were greater than the total throughfall inputs (Fig. 7-9), with the $K^{+}$ flux being the highest (6.5 kg ha$^{-1}$a$^{-1}$). In the percolate from below the B-horizon, the sum of base cation flux was nearly balanced by the flux of $SO_4^{2-}$ anions, but not in the case of the percolate from below the O- and E-horizons (Fig. 4 in paper II). Much of the base cation percolate flux was retained in the mineral soil (Fig. 7-9), presumably as a consequence of uptake or adsorbtion onto exchange sites. The annual flux of base cations from below the B-horizon was small, the soil behaving as a sink for atmospheric inputs of $Ca^{2+}$ and $K^{+}$ while outputs of $Mg^{2+}$ were equal to deposition

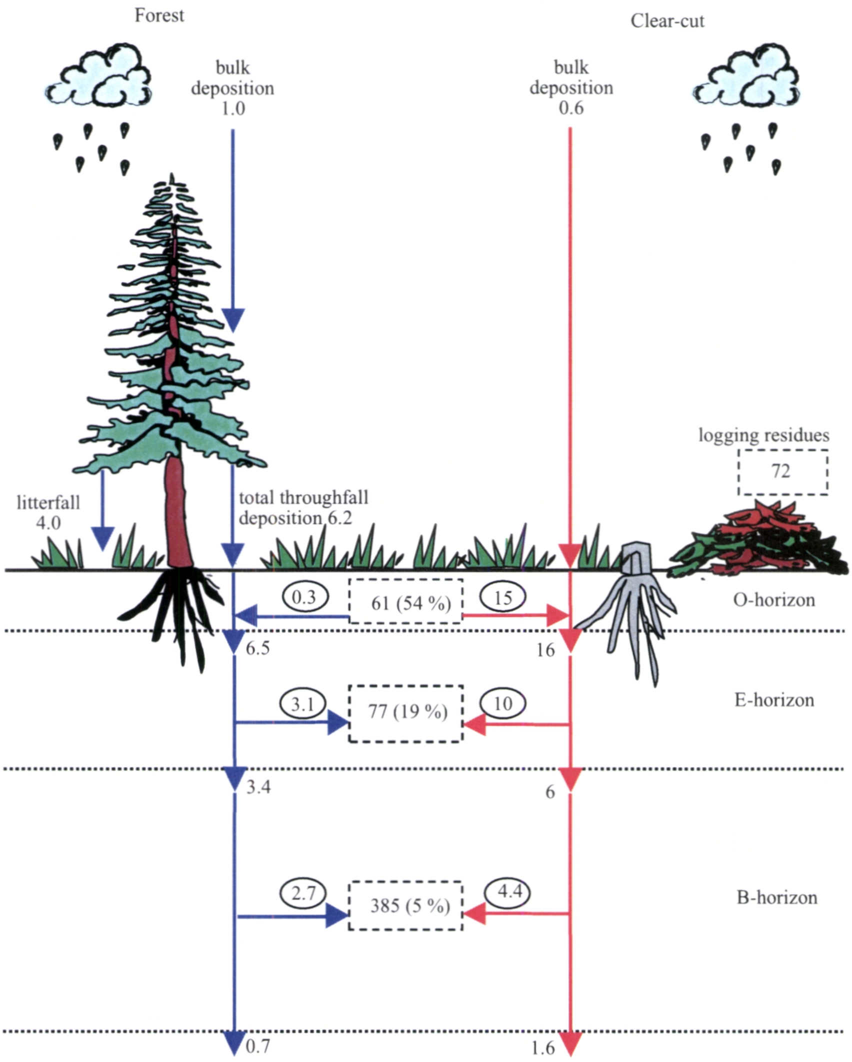

**Figure 9. Fluxes of K (kg ha$^{-1}$ a$^{-1}$) through the vegetation and soil before and after clear-cutting. See further explanation in Fig. 5.**

inputs. The annual output of base cations from forested boreal catchments to streamwaters have been shown to be < 8 kg ha$^{-1}$ for $Ca^{2+}$, < 3 kg ha$^{-1}$ for $Mg^{2+}$ and < 2 kg ha$^{-1}$ for $K^+$ (Grip 1982, Rosén 1982, Watmough & Dillon 2001).

The pool of total base cations in the soil was large, with only a small fraction being plant available (Fig. 7-9). Most of the bioavailable pool of $Ca^{2+}$ and $Mg^{2+}$ in upland forest ecosystems is to be found in the soil, whereas most of the $K^+$ pool bound into the vegetation (e.g. Nykvist 1971, Mälkönen 1975, Likens & Bormann 1999).

## 4.2 Effects of clear-cutting on fluxes

### *4.2.1 Water*

Interception at Kangasvaara was small, and the amount of precipitation reaching the forest floor after clear-cutting increased by only 8 %. The bulk precipitation was probably underestimated, however, as discussed earlier (chapter 4.1.1). The proportion of incident precipitation percolating from below the O-horizon was unchanged after clear-cutting, indicating that the loss of canopy interception was replaced by the interception of logging residues. Only at the cut plot 1 in 1999, was there a clear decrease in the fraction of precipitation passing through the O-horizon; 67 % compared to 84 % before cutting (papers III and IV). However, from below the B-horizon, the proportion of precipitation passing through increased to 7 % from 4 % at plot 1 in 1999. At plot 2, changes were observed only in the proportion of precipitation passing through the B-horizon; an increase in 1997 and 1998 to 9 and 6 % from an average of 5 % before cutting (papers III and IV). Streamwater runoff generally increases after clear-cutting in similar climate conditions (e.g. Seuna 1988). The reasons for the observed minor changes in soil water percolate fluxes were probably complicated. For example, the infiltration capacity of the soil could have been weakened due to the dryness and contraction of the O-horizon, and the amount of soil water moving horizontally as subsurface flow could have increased as a result of increased soil water content.

### *4.2.2 Carbon*

Clear-cutting eliminated the C fluxes to the forest floor from both total throughfall and litterfall, but a large amount of C was added in the form of logging residues (Fig. 3). The elevated leaching of DOC from the O-horizon reflects the decomposition of these logging residues and possibly also increased decomposition of the organic layer itself. DOC leaching from the O-horizon was almost three times larger after cutting than before (paper III). The decomposition of logging residues starts immediately after cutting. According to a study by Hyvönen et al. (2000), the mass loss of logging residue needles after 3 years was as large as 82 %. Much of the C in the logging residues must therefore have been lost to the atmosphere as $CO_2$ through microbial respiration, as also observed by Mattson et al. (1987).

In the mineral soil DOC was effectively retained, through accumulation of humic Al and Fe sesquioxide complexes and by biological immobilization and mineralization (paper III). The relative retention of DOC by the E- and B-horizon was unaffected by clear-cutting so that the flux from below the B-horizon did not change (Fig. 3). In a clear-cutting study in a temperate deciduous forest by Qualls et al. (2000), elevated DOC fluxes were observed from below the soil C-horizon.

### *4.2.3 Nitrogen*

After clear-cutting, the deposition input of DIN to the forest floor increased and that of DON decreased (Fig. 1 in paper III). The net result of this was that the deposition of total N was unchanged (Fig. 4). The annual flux of N to the ground as litterfall ended at clear-cutting, but there was a large and single addition in the form of logging residues (Fig. 4). The soil water flux of DTN from below the O-horizon increased after cutting, and over 70 % was organic N (paper III). But the loss over the three-year period was small compared to the total N pool in logging residues and O-horizon (Fig. 4). The proportions of DIN and DON in the percolate collected from below the O-horizon was unchanged, indicating that N mineralization was not accelerated after clear-cutting (paper III). The mean C/N ratio of the logging residues was 104 (Table 1 in paper III). The threshold value for net mineralization is reported to be 60-110 (Berg & Staaf 1980, Berg & Ekbohm 1983). However, the C/N ratio varied among different biomass fractions, being much higher in the branches than in foliage (paper V). Previous studies have shown a marked increase in DIN leaching from the O-horizon after clear-cutting, often dominated by $NH_4^+$ in the case of coniferous forests in the boreal zone (Rosén & Lundmark-Thelin 1987) and by $NO_3^-$ on richer sites in the temperate zone (Sollins & McCorison 1981). At Kangasvaara, the proportion of $NH_4^+$ was greater than that of $NO_3^-$ (Fig. 3 in paper III).

The mineral soil efficiently retained the increased input of $NH_4^+$, probably due to cation exchange but a part may also have been nitrified to $NO_3^-$. Elevated fluxes of $NO_3^-$ from below the B-horizon were observed on occasions after clear-cutting (Fig. 1 in paper III), indicating net nitrification or reduced root uptake. Nitrification is generally low in acidic boreal coniferous forest soils but after clear-cutting, changing moisture, temperature and pH conditions and availability of $NH_4^+$ may promote nitrification (Matson & Vitousek 1981, Martikainen 1984) and leaching of $NO_3^-$ can occur (Tamm et al. 1974, Staaf & Olsson 1994, Kubin 1998, Ahtiainen & Huttunen 1999).

Increased DIN leaching may have occurred if deposition loads of N had been higher (Adamson et al. 1987, Wiklander et al. 1991) or if the site was more fertile, with a higher potential for nitrification (Stevens & Hornung 1988). However, under the conditions at Kangasvaara, the increase in DTN flux from below the B-horizon after clear-cutting was negligible (Fig. 4).

### *4.2.4 Phosphorus*

The deposition inputs of P to the forest floor were not altered much after clear-cutting (Fig. 5). The logging residues added a considerable amount of P to the forest floor and the leaching of P from the O-horizon increased immediately after clear-cutting (Fig. 2 in paper IV). Over the three-year period after clear-cutting, 6.9 kg P ha$^{-1}$ leached from below the O-horizon (Fig. 5), mainly (75 %) as $PO_4^{3-}$-P (paper IV). The increased P flux most probably originated from the logging residues, since the C/P ratio of the logging residues averaged 866 (Table 1 in paper IV). The threshold value for net mineralization has been shown to be 2000 (Berg & Staaf 1980). Qualls et al. (2000) also reported increased $PO_4^{3-}$ concentrations in soil percolate collected from below the logging residue and organic layer after the clear-cutting of a deciduous forest.

The mineral soil effectively retained both the inorganic and organic P leaching from the O-horizon (paper IV). This resulted in unchanged fluxes from below the B-horizon after the clear-cutting (Fig. 5). The retention of P in the mineral soil was very effective and can be explained by the Al- and Fe-oxide content of the B-horizon (Table 1 in paper II). Thus, cutting increased the pool of P in the mineral soil. The adsorption of P by mineral soil also explains why P concentrations in watercourses has generally not been affected by catchment clear-cutting (Stevens et al. 1995, Yanai 1998, Briggs et al. 2000). Some studies, however, have shown increased total P concentrations in stream water after cutting (Adamson et al. 1987, Ahtiainen & Huttunen 1999). Runoff in these cases probably occurs through the organic horizon, at least during some parts of the year such as during snowmelt or when ground-water table levels are raised, resulting in anaerobic conditions under which P becomes soluble (Armstrong 1982).

### *4.2.5 Sulphur*

The annual deposition of $SO_4^{2-}$ to the forest floor decreased by more than 50 % after clear-cutting (Fig. 1 in paper IV) and that of organic S by 43 % (paper IV). The throughfall deposition of total S averaged 5 kg ha$^{-1}$ a$^{-1}$ before cutting (Fig. 6). The annual flux of S via litterfall was small, but the logging residues after cutting resulted in a large input of S to the forest floor. However, the flux of total S from below the O-horizon decreased after cutting and the proportion as organic S increased a little (paper IV). The C/S ratio of the logging residues averaged 1184 (Table 1 in paper IV), which is less than the threshold value for net mineralization of 1500 reported by Berg & Staaf (1980). In the first year after clear-cutting, the retention of $SO_4^{2-}$ by the O-horizon decreased (Fig. 2 in paper IV), probably due to diminished nutrient uptake and increased mineralization of the logging residues. Later on, $SO_4^{2-}$ retention by the O-horizon returned to the level before cutting, indicating increased uptake and/or decreased mineralization. Sulphate retention in the mineral soil decreased after clear-cutting (Fig. 2 in paper IV). The reduction may also be the result of decreased root uptake but it also may be due to reduced adsorption as a result of lowered soil solution $SO_4^{2-}$ concentrations and increased pH (Fig. 1 and 3 in paper IV). The adsorption of $SO_4^{2-}$ by soil is controlled by the amount and degree of saturation of amorphous Al and Fe compounds (Nodvin et al. 1986, Nodvin et al. 1988, Harrison & Johnson 1992). The reaction is known to be partly reversible, depending on the soil solution $SO_4^{2-}$ concentrations and pH (Harrison & Johnson 1992, Karltun 1995, Gobran et al. 1998). Increased mineralization of organic S might also explain the observed increases in $SO_4^{2-}$ fluxes from below the B-horizon (paper IV). However, the flux of total S from below the B-horizon did not increase after clear-cutting (Fig. 6), which is in agreement with findings from temperate forest studies (Likens et al. 1970, Likens et al. 1977, Fuller et al. 1987).

### *4.2.6 Base cations*

The throughfall deposition of base cations decreased after clear-cutting; that of $K^+$ the most, some 90 % (Fig. 7, 8 and 9). However, the amount of base cations coming to the forest floor increased after clear-cutting due to the logging residues. Base cation leaching from the O-horizon also increased, particularly that of $K^+$ (Fig.4 in IV). These increased fluxes of base cations can probably be attributed to the decomposition and mineralization of the logging residues (Titus & Malcolm 1992, Staaf & Olsson 1994, Stevens et al. 1995, Robertson et al. 2000), but leaching of cations, especially that of $K^+$, can occur directly from dead needles (Tukey Jr 1970, Ukonmaanaho & Starr 2001). In the case of $Ca^{2+}$, the increased leaching is probably also due to the leaching from the organic matter in the O-horizon itself (Hendrickson et al. 1989).

Before clear-cutting the flux of the sum of base cations from below the O-horizon was correlated with the flux of $SO_4^{2-}$ anions (paper II). However, cutting decreased the $SO_4^{2-}$ flux and increased that of $PO_4^{3-}$ anions and soluble organic matter measured as DOC. Thus, after clear-cutting, the $K^+$ flux was most strongly correlated with that of $PO_4^{3-}$ anions (paper IV) and that of $Ca^{2+}$ and $Mg^{2+}$ with the DOC flux ($Ca^{2+}$: $r=0.86$, $n=40$; $Mg^{2+}$: $r=0.74$, $n=40$), indicating that organic anions were the main accompanying counter anions. The soil water percolate flux of $K^+$ from below the E-horizon was also most strongly correlated with the $PO_4^{3-}$ ions flux (paper IV) and those of $Ca^{2+}$ ($r=0.63$, $n=38$) and $Mg^{2+}$ ($r=0.55$, $n=38$) with

DOC flux. In the percolate from below the B-horizon, the flux of $K^+$ became more strongly correlated with that of $SO_4^{2-}$ anions (paper IV) while that of $Ca^{2+}$ remained the most strongly correlated with DOC (r=0.51, n=18) and that of $Mg^{2+}$ was most strongly correlated with the $NO_3^-$ flux (r=0.58, n=20).

Nevertheless, the increased input fluxes of base cations to the mineral soil were efficiently retained, probably due to adsorption onto soil exchange sites. This would result in a theoretical increase in the soil pool of cations after clear-cutting. But this increase is likely to be small compared to the total (*Aqua Regia*) pools (Fig. 7-9). Although mean annual balances showed retention of base cations occasional increased fluxes of $Mg^{2+}$ and $K^+$ were observed from below the B-horizon (Fig. 4 in paper IV). However, the leaching of base cations at Kangasvaara is likely to remain low since the leaching of anions is also low and because of the effectiveness of cation exchange reactions in the soil.

# 5 CONCLUSIONS

The results from this study showed that, in spite of a large change in the input pathways to the soil, nutrient fluxes from below the B-horizon remained small and little affected by clear-cutting. Nutrients reaching the soil in deposition and in the form of logging residues were effectively retained in the soil. The retention could be attributed to immobilization by soil microbes, uptake by ground vegetation, and adsorption by the soil. The Al- and Fe-oxides that characterise the B-horizon of the podzolic soil were implicated in the adsorption of $SO_4^{2-}$ and particulary $PO_4^{2-}$ anions. There was also little effect of clear-cutting on $NO_3^-$ production, which remained low. The lack of an increase in accompanying anions and the adsorption of the base cations by the soil explain why there was little effect also on the fluxes of $Ca^{2+}$, $Mg^{2+}$ and $K^+$. There was evidence that DOC fluxes and that of the associated organic anions replaced inorganic anions in accompanying the leaching of base cations.

The deposition loads of S and N and associated acidity at Kangasvaara were small, which restrict the generalization of the results to the areas with higher deposition inputs. Also the measurement of the amount of soil percolates in mineral soil and especially the soil water movement need more accurate methods. The percolates collected in this research represent probably only one component of the soil water reaching watercources, but it is unclear which proportion.

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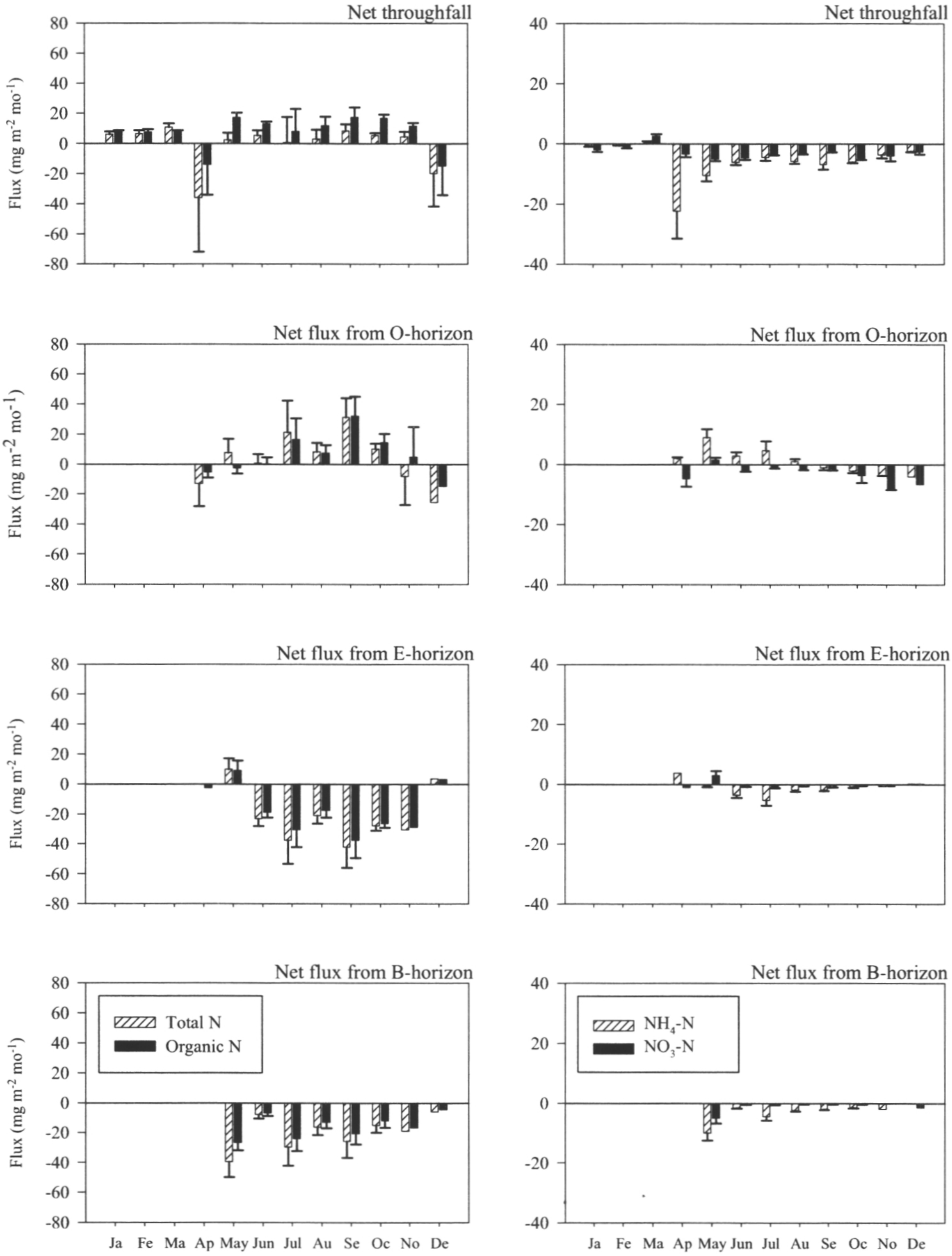
# Corrections

**1**. Figure 3 is incorrect in paper I, the correct figure follows:



# PAPER I

Piirainen, S., Finér, L. & Starr, M. 1998. Canopy and soil retention of nitrogen deposition in a mixed boreal forest in Eastern Finland. Water, Air, and Soil Pollution, 105: 165-174.

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# CANOPY AND SOIL RETENTION OF NITROGEN DEPOSITION IN A MIXED BOREAL FOREST IN EASTERN FINLAND

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**Abstract.** Nitrogen deposition, leaching, and retention were monitored in a mature spruce (*Picea abies* Karsten) dominated mixed boreal forest in eastern Finland. Bulk precipitation, throughfall, stemflow, and percolation through the podzolic soil profile were monitored from 1993 to 1996. Mean annual bulk deposition of total N was 3.83 kg ha$^{-1}$, of which 33 % was $NH_4^+$-N, 26 % was $NO_3^-$-N, and 41 % was organic N. Throughfall+stemflow flux of total N was 2.93 kg ha$^{-1}$ yr$^{-1}$. Sixty-four % of $NH_4^+$-N and 38 % of $NO_3^-$-N in bulk precipitation was retained by the tree canopy. Organic N was released (0.27 kg ha$^{-1}$ yr$^{-1}$) from the tree canopy. Nitrate-N was retained and organic N was leached as the water passed through the ground vegetation and soil O-horizon. Ammonium-N and organic N were retained mainly in the E-horizon. The output of total N from the E-horizon was only 5 % of the total N deposition in the forest stand during the study period and it was mainly as organic N. The output of inorganic N forms from under B-horizon was seasonal and occurred mainly at spring snowmelt.

**Keywords:** canopy, forest, nitrogen leaching, nitrogen retention

## 1. Introduction

Nitrogen (N), often the growth limiting nutrient in boreal coniferous forests, is tightly cycled with minor leaching (Tamm, 1991). The deposition of N may result in the acidification and eutrophication of N-limited freshwater ecosystems because the critical load for the most sensitive ecosystems (e.g., softwater lakes) is 5-10 kg N ha$^{-1}$ yr$^{-1}$ (Bobbink and Roelofs, 1995). In Finland, the annual N deposition ranges from 10.6 kg ha$^{-1}$ in the south to 0.9 kg ha$^{-1}$ in the north (Järvinen and Vänni, 1994). Because of retention in the canopy, amounts of throughfall N below boreal coniferous forest canopies are typically less than in bulk precipitation. Studies in Finnish boreal forests have shown that canopy retention of N may be 10-40 % of total N deposition (Hyvärinen,1990; Helmisaari, 1995; Starr, 1995). In areas with high N deposition (>10 kg ha$^{-1}$ yr$^{-1}$), $NO_3^-$ can leach in considerable amounts to groundwater or surface water (Gundersen, 1995). In areas with low N deposition (< 3 kg ha$^{-1}$ yr$^{-1}$), < 5-10 % of N deposition leaches below the surface layers in podzolic soils, and then mainly during snowmelt when there is little or no uptake (Helmisaari, 1995). Disturbances to forest ecosystems, such as forest harvesting, decrease canopy and root uptake and can increase the leaching of N to surface water (Lepistö *et al.*, 1995).

In this paper, we present data on the fate of $NH_4^+$-N, $NO_3^-$-N, and organic N in bulk deposition as it passes through the canopy and soil of a coniferous stand. The extent of N retention by the forest canopy and soil and the amount of N leaching from the soil are determined and evaluated. These baseline data are a part of an ongoing project that evaluates the effects of forest harvesting on the leaching of nutrients to surface waters (Finér *et al.*, 1997).

*Water, Air, and Soil Pollution* **105:** 165–174, 1998.
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## 2. Material and Methods

### 2.1. STUDY AREA

Data were collected from three 50 m × 50 m sample plots located in a mature mixed boreal forest in eastern Finland (63° 51' N, 28° 58' E, 220 m a.s.l.). Norway spruce (*Picea abies* Karsten) dominates the stand, but Scots pine (*Pinus sylvestris* L.), white and silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth.) and European aspen (*Populus tremula* L.) also are present. Norway spruce accounts for 50 %, Scots pine 34 %, and the deciduous tree species 16 % of the total volume of 266 $m^3$ $ha^{-1}$ in the tree stand. The canopy coverage in the stand is 54 %. The forest is uneven-aged; the oldest trees are 170 years old.

The site type is *Vaccinium-Myrtillus* type according to the classification of Cajander (1949). The field layer is dominated by dwarf shrubs (*Vaccinium vitis-ideaea* L. and *V. myrtillus* L.) and the bottom layer by feather mosses (*Pleurozium schreberi* Brid. and *Hylocomnium splendens* (Hedw.) B.S.&G.).

The soil, derived from sandy till, is a weakly developed iron-podzol (according to the classification of Kubiëna, 1953), with a clay content of <2 %. The bedrock is composed of granodiorite. The average thickness of the mor humus layer is 3 cm. Mean annual air temperature recorded at 2 m for the period 1993-1995 averaged +1.0 °C and the temperature sum was 908 °C (sum of daily mean temperatures >+5 °C). A more detailed description of the study area is presented by Finér *et al.* (1997).

### 2.2. DEPOSITION AND SOIL WATER SAMPLING

Bulk precipitation and throughfall were monitored from 1993 to 1996 and stemflow from 1993 to 1995. Bulk precipitation was collected in an open, treeless site within 500 m of the sample plots. Five plastic collectors (each 130.7 $cm^2$) were used during the snow-free period and three plastic bucket snow collectors were used during the winter (each 298.7 $cm^2$ for winter 1992/1993 and 1146.1 $cm^2$ thereafter). The collectors were placed 0.6 m above the ground during the snow-free period and 1.5 m above the ground in winter. Throughfall was measured in each sample plot using 16 collectors during the snow-free period and eight snow collectors during winter. The collectors were of the same type as used for bulk precipitation and they were placed systematically around the sides of the plots. Stemflow was collected at two of the sample plots. At each plot, five white or silver birches, Scots pines and Norway spruces representing different diameter classes (>6 cm) were fitted with spiral type stemflow collectors made of silicon tubing. The bulk precipitation and throughfall collectors were emptied once a week in the snow-free period and once a month in winter. On each sampling occasion, the samples from all collectors were bulked for analysis, by plot in the case of bulk deposition. Stemflow was collected weekly and only during the snow-free period. The samples were combined by species and plot for chemical analyses.

Soil water was sampled in each sample plot with nine systematically placed zero-tension lysimeters installed at three depths: under the humus layer (O-horizon), under the eluvial horizon (E-horizon), and 23 cm below the boundary of eluvial and illuvial horizons (B-horizon). The lysimeters under the O-horizon were of the Jordan-type (Jordan, 1968) and made of a plexiglass with a collecting area of 420 $cm^2$. The lysimeters installed under the E- and B-horizons were made of a polythene plastic funnel with a collecting area of 299 cm$^2$ and fitted to a 2 L bottle. The lysimeters were emptied on the same day as the precipitation collectors and the volume collected was recorded. Soil water samples were taken irregularly from October to April. The soil water samples from each lysimeter were analyzed separately.

### 2.3. LABORATORY ANALYSES AND CALCULATIONS

Samples were kept cold after sampling and were transported to laboratory the day after sampling. Bulk precipitation, throughfall, and stemflow samples were filtered through a Schleiche & Schuell no. 5891 filter and the soil percolate samples through a Schleiche & Schuell no. GF 52 glass wool filter; filtrates were stored in a freezer before analyzing. Ammonium and total N were determined by flow injection analysis (Tecator FIA-Star 5020) and $NO_3^-$ by ion chromatography (Dionex series 100 or 300 or 500). Organic N was calculated by subtracting the sum of $NH_4^+$-N and $NO_3^-$-N concentrations from the total N concentration. Deposition and leaching values were calculated by multiplying the concentrations by the amount of water collected. Pearson product-moment correlations and the paired *t*-test with a probability of Bonferroni´s inequality were used to evaluate the data statistically (Snedecor and Cochran, 1980) using SAS for Windows 6.11.

## 3. Results

### 3.1. BULK PRECIPITATION, THROUGHFALL, AND STEMFLOW

The mean annual bulk precipitation for 1993-1996, was 502 mm (±52 mm); monthly bulk precipitation ranged from 11 to 116 mm. Precipitation fell mainly as snow from November to April, and accounted for 45 % of the annual bulk precipitation. Throughfall was 86-100 % of annual bulk precipitation and stemflow was almost negligible, only 1.3-1.5 % of annual bulk precipitation. Monthly total throughfall (throughfall+stemflow) ranged from 9 to 114 mm.

The N in bulk precipitation was dominated by inorganic forms, whereas in throughfall and stemflow, organic N concentrations were higher (Table I). The $NH_4^+$-N and $NO_3^-$-N concentrations in throughfall were significantly lower than those in bulk precipitation, and organic N concentration in stemflow was significantly higher than that in throughfall and bulk deposition.

Over the study period, deposition of total N in bulk precipitation averaged 3.83 kg ha$^{-1}$ yr$^{-1}$ (Figure 1), 59 % being inorganic N. The annual bulk deposition of $NH_4^+$-N (1.27 kg ha$^{-1}$) was slightly higher than that of $NO_3^-$-N (1.01 kg ha$^{-1}$), contributing 33 % and 26 % to the annual total N deposition, respectively. Deposition of $NO_3^-$-N was greater during the dormant season than during the growing season; the opposite pattern was observed for $NH_4^+$-N (Figure 2).

TABLE I

Water volume-weighted mean annual concentrations (mg $L^{-1}$; SD in parentheses) of different N compounds during 1993-1996. Values within columns followed by the same letter do not differ significantly ($p = 0.0125$) from each other.

| | Total N | $NH_4^+$-N | $NO_3^-$-N | Organic N |
|---|---|---|---|---|
| Bulk precipitation | 0.76 (0.22) ef | 0.25 (0.04) a | 0.21 (0.02) a | 0.32 (0.20) a |
| Throughfall | 0.60 (0.05) abe | 0.11 (0.03) b | 0.15 (0.02) b | 0.35 (0.03) a |
| Stemflow | 1.35 (0.12) cdf | 0.07 (0.06) ab | 0.04 (0.01) bc | 1.21 (0.11) b |
| O-horizon | 1.06 (0.34) acef | 0.15 (0.09) ab | 0.04 (0.02) abc | 0.85 (0.23) b |
| E-horizon | 0.94 (0.26) ac | 0.17 (0.08) ab | 0.06 (0.06) abc | 0.71 (0.18) b |
| B-horizon | 0.35 (0.20) bd | 0.06 (0.03) b | 0.04 (0.04) c | 0.25 (0.15) a |

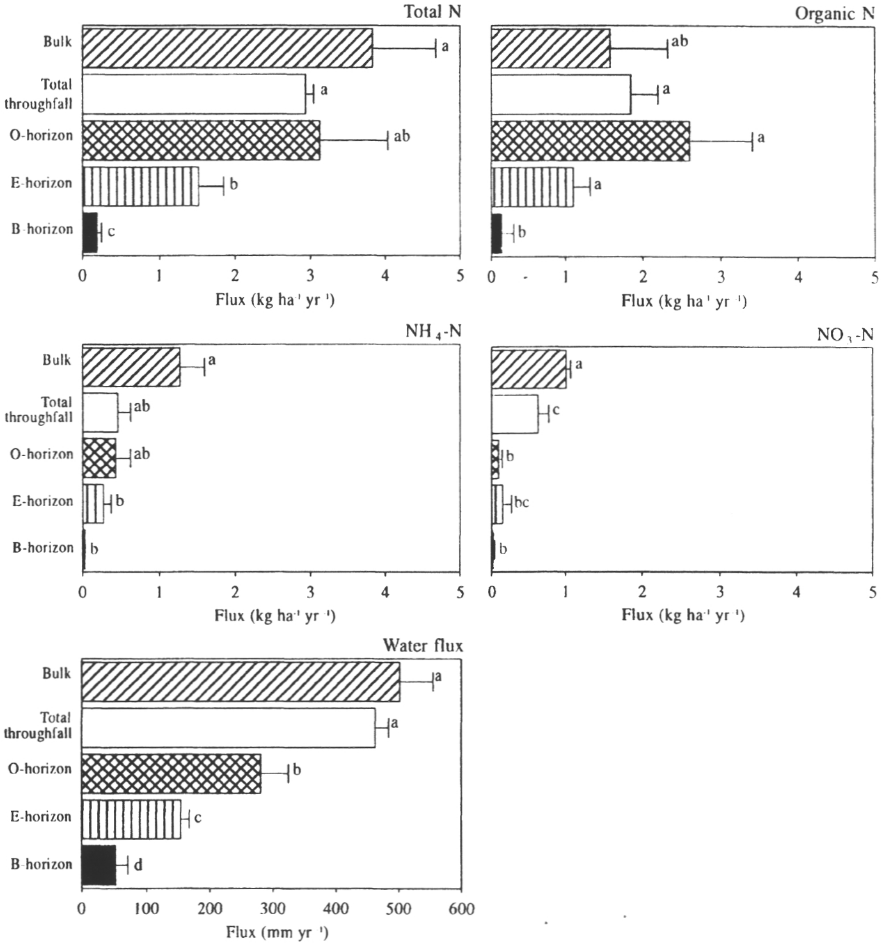

Fig. 1. Mean annual fluxes (±SD) of different N compounds and water in bulk deposition, total throughfall, and percolates from below the O-, E- and B-horizons; 1993-1996. Values with the same letter do not differ significantly ($p = 0.0125$) from each other.

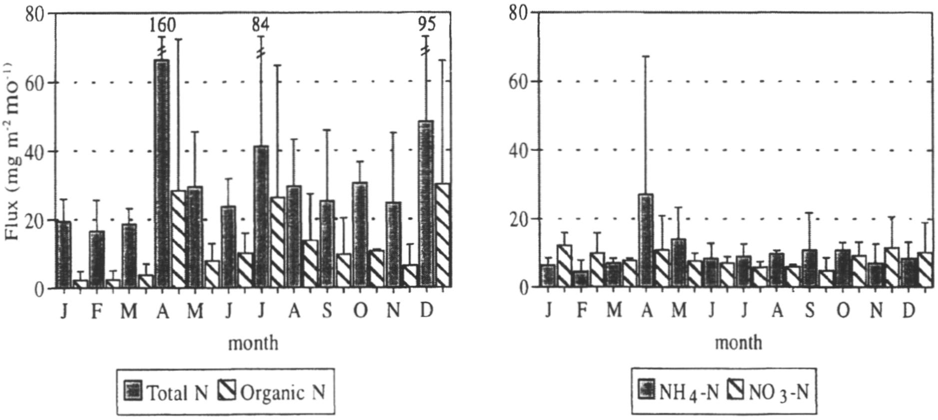

Fig. 2. Monthly mean bulk deposition (±SD) of different N compounds; 1993-1996.

Annual N deposition in total throughfall averaged 2.93 kg ha$^{-1}$, i.e. 77 % of that in bulk deposition (Figure 1). Total N that reached the forest floor via stemflow was only 2-3 % of total N deposition in total throughfall during the study period. Annual deposition of $NH_4^+$-N and $NO_3^-$-N in total throughfall was 0.46 and 0.63 kg ha$^{-1}$, respectively (Figure 1). However, the difference between bulk precipitation and total throughfall (net throughfall) shows that the forest canopy was a sink for inorganic N (Figure 3). The forest canopy retained 38 % of the $NO_3^-$-N bulk deposition and 64 % of the $NH_4^+$-N bulk deposition, corresponding to a net uptake of 1.19 kg N ha$^{-1}$ yr$^{-1}$. Canopy retention of $NH_4^+$-N averaged 0.81 kg ha$^{-1}$ yr$^{-1}$ and of $NO_3^-$-N, 0.38 kg ha$^{-1}$ yr$^{-1}$ and was the least during the dormant season. During the growing season, 76 % of $NH_4^+$-N and 61 % of $NO_3^-$-N deposition were retained by the canopy (Figure 3). Monthly canopy uptake of $NH_4^+$-N and $NO_3^-$-N was correlated with the monthly bulk deposition of $NH_4^+$-N ($r = 0.98$, $n = 44$, $p < 0.01$) and of $NO_3^-$-N ($r = 0.72$, $n = 38$, $p < 0.01$).

The annual deposition of organic N in total throughfall averaged 1.85 kg ha$^{-1}$, corresponding to 56 % of the total N in total throughfall (Figure 1). Organic N was released from the canopy in spring and autumn, but was retained in the canopy during the growing season (Figure 3). The annual release of organic N from the canopy averaged 0.27 kg ha$^{-1}$.

### 3.2. SOIL PERCOLATES

During the study period, 61, 33 and 11 % of the total annual throughfall, on average, passed through the O-, E- and B-horizons, respectively (Figure 1). The average monthly soil percolate from below the O-horizon ranged from 0 to 59 mm. Usually the soil was frozen from December to March and no percolation occurred. At snowmelt, and occasionally also after heavy rainfall in summer and autumn, soil water percolation was at its greatest. In the summer months, the average monthly amount of soil water percolate from below the B-horizon varied from 1 to 3 mm.

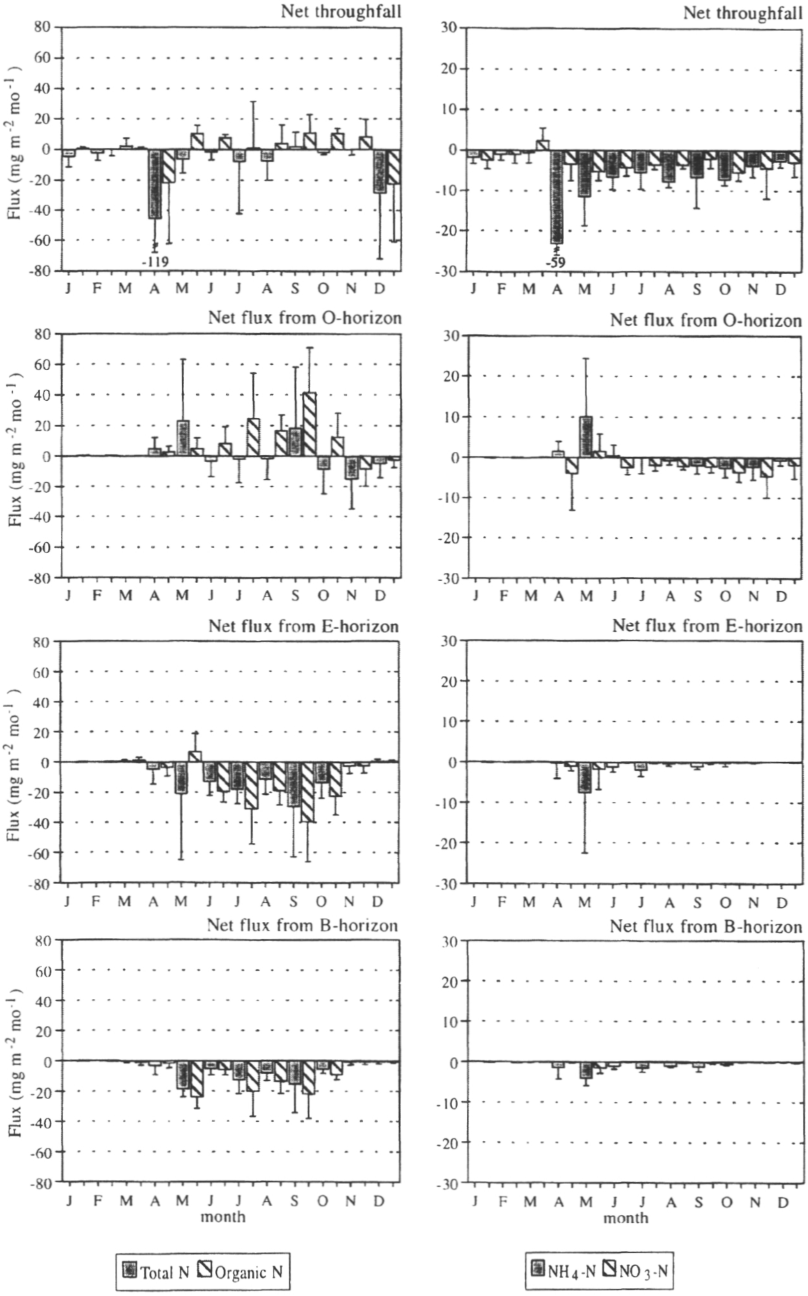

Fig. 3. Monthly mean (±SD) net flux (input-output) of different N compounds from forest and under different soil horizons in 1993-1996; negative values indicate retention and positive values indicates leaching.

Mean annual $NO_3^-$-N concentrations in the soil percolate from the O-horizon were lower than concentrations in throughfall, and remained almost constant deeper in the soil (Table I). Mean annual $NH_4^+$-N concentration of the soil percolate was similar at all depths. Mean annual organic N concentration in the soil percolate increased as it passed through the O-horizon, but decreased below the B-horizon.

Some leaching of $NO_3^-$-N from the O-horizon also was observed at spring snowmelt, but most of the $NO_3^-$-N was retained (Figure 3). The retention of $NO_3^-$-N in the O-horizon was considerable; 84 % (0.53 kg ha$^{-1}$ yr$^{-1}$) of the annual average total throughfall deposition. Monthly total throughfall deposition of $NO_3^-$-N and net retention of $NO_3^-$-N by the O-horizon were correlated for the period when soil water percolation occurred ($p < 0.01$, $r = 0.91$, $n = 30$). The retention of $NO_3^-$-N in the mineral soil layers also was high, especially in the B-horizon; output from below the B-horizon was only 2 % of the input to the forest stand via bulk deposition.

Some $NH_4^+$-N leaching from the O-horizon occurred at spring snowmelt (Figure 3). During the summer and autumn, $NH_4^+$-N was retained in the O-horizon. Monthly total throughfall deposition of $NH_4^+$-N and net retention of $NH_4^+$-N in the O-horizon during growing period also were correlated ($p < 0.01$, $r = 0.47$, $n = 29$). On an annual basis, $NH_4^+$-N output from below the O-horizon was balanced by the total throughfall deposition (Figure 1). The mineral soil horizons retained $NH_4^+$-N during the growing period; the output from below the B-horizon being 2 % of the bulk precipitation flux. The monthly flux of $NH_4^+$-N to the E- and B-horizons was correlated with the retention of $NH_4^+$-N by these horizons ($r = 0.94$, $p < 0.01$, $n = 30$ and $r = 0.99$, $p < 0.01$, $n = 28$, respectively). On an annual basis, 93 % of the $NH_4^+$-N input to the mineral soil horizons from O-horizon was retained.

Retention of inorganic N in total throughfall deposition by the soil was 0.43 kg ha$^{-1}$ yr$^{-1}$ for $NH_4^+$-N and 0.61 kg ha$^{-1}$ yr$^{-1}$ for $NO_3^-$-N. The organic horizon retained most of the $NO_3^-$-N (87 %) and the B-horizon retained most of the $NH_4^+$-N (56 %).

On an annual basis, 0.75 kg ha$^{-1}$ yr$^{-1}$ of organic N was leached from the O-horizon, but it was subsequently retained in mineral soil horizons (Figure 3). The retention of the organic N leached from O-horizon was 1.50 kg ha$^{-1}$ yr$^{-1}$ in the E-horizon and 0.96 kg ha$^{-1}$ yr$^{-1}$ in the B-horizon. Output from below the B-horizon was only 9 % of the bulk precipitation input of organic N.

The output of total N from below the B-horizon (0.19 kg ha$^{-1}$ yr$^{-1}$) was 5 % of bulk precipitation total N input (Figure 1). Almost 74 % of total N in the percolate from below the B-horizon was organic. In total, 2.46 kg of organic N was retained jointly by the forest canopy and soil, with the E-horizon retaining the most (61 %).

## 4. Discussion

### 4.1. RETENTION BY CANOPY

The critical load of N to coniferous forests has been estimated as 10-20 kg ha$^{-1}$ yr$^{-1}$, (Bobbink and Roelofs, 1995). In Finland, only 65 % of the inorganic N input in bulk precipitation is deposited as wet precipitation (Tuovinen *et al.*, 1990). Dry deposition of N is dominated by $HNO_3$, $NO_2^-$-N and $NH_3$-N; particulate nitrate and particulate ammonium sulfate are also notable components (Tuovinen *et al.*, 1990). On the basis of the values given by Tuovinen *et al.* (1990), we estimated that the inorganic dry deposition of N to the study area is 1.23 kg ha$^{-1}$ yr$^{-1}$, and the total deposition of N is about 5.0 kg N ha$^{-1}$ yr$^{-1}$.

The N in total throughfall represents the balance between leaching, canopy uptake and dry deposition (Parker, 1983). Total throughfall contained 52 % of the inorganic N present in bulk precipitation; a similar percentage has been observed in other coniferous forests (Rosén and Lundmark-Thelin, 1985; Helmisaari and Mälkönen, 1989; Hyvärinen, 1990; Starr and Ukonmaanaho, 1995). The retention of inorganic N in the canopy may be due to foliar uptake (Carlisle *et al.*, 1966; Tukey, 1970; Reiners and Olson, 1984), uptake by epiphytic lichens (Lang *et al.*, 1976; Reiners and Olson, 1984), and in the case of $NH_4^+$, by cation exchange (Tukey, 1970). Spruce canopies have been shown to be the strongest sinks for inorganic N and also to have the highest biomass of epiphytic lichens (Lovett, 1992). Of the mean annual retention of inorganic N by the forest canopy (1.19 kg ha$^{-1}$), most was $NH_4^+$-N. If the estimate for dry deposition is considered, the total N retention would be as much as 2.42 kg ha$^{-1}$ yr$^{-1}$.

The bulk deposition of $NH_4^+$-N was 26 % greater than that of $NO_3^-$-N, but the canopy retention of $NH_4^+$-N was 113 % greater than that of $NO_3^-$-N. Norway spruce utilizes $NH_4^+$ directly via the needles more readily than $NO_3^-$ (Bowden *et al.*, 1989; Eilers *et al.*, 1992; Lumme, 1994). Increasing inorganic N uptake with increasing inorganic N deposition may be a general characteristic of spruce canopies (Eilers *et al.*, 1992; Lovett, 1992).

The demand for N by coniferous forests is 30-70 kg ha$^{-1}$ yr$^{-1}$ (Nihlgård, 1972; Mälkönen, 1975; Finér, 1989; Helmisaari, 1995). On this basis, Likens *et al.* (1977) and Helmisaari (1995) concluded that N deposition supplies about 10 % of the annual N demand; the rest being supplied by recycling. According to our results, the bulk deposition of total N (including dry deposition) supplied about 3-8 % of the annual N requirement of the stand.

### 4.2. LEACHING OF N FROM SOIL

The mean annual $NO_3^-$-N concentration in the percolate from below the organic horizon was lower than in total throughfall. The uptake by roots and microorganisms is probably efficient; only 16 % of the $NO_3^-$-N deposition in total throughfall passed through the O-horizon (cf. Brinkmark, 1980; Helmisaari, 1995). Some leaching of $NO_3^-$-N occurred in spring, during snowmelt, when the biological uptake is small and water flux is high. The deposition of $NO_3^-$-N during winter months was higher than during growing season. As a result, the water equivalent of the snowpack contains more $NO_3^-$ than a corresponding amount of rainfall deposition (Overrein *et al.*, 1980). Nitrate leaching from soil during snowmelt has also been reported by Helmisaari and Mälkönen (1989) and Pardo *et al.* (1995). Stream water $NO_3^-$-N concentrations also are typically higher during the spring meltwater than during other seasons in boreal forest ecosystems (Overrein *et al.*, 1980).

As also observed by Helmisaari and Mälkönen (1989), $NH_4^+$-N was leached from the O-horizon in spring, during snowmelt, and retained during the summer and autumn. Bringmark (1980) suggested that the higher retention of $NH_4^+$-N in autumn compared to other times of the year corresponds to uptake resulting from a larger population of fungi and bacteria at this time. On an annual basis, however, the input and output of $NH_4^+$-N for the O-horizon were in balance. Helmisaari (1995) reported a net annual leaching of $NH_4^+$-

N from the organic horizon. However, her study site was a 100-year-old Scots pine stand. Norway spruce roots have been shown to prefer $NH_4^+$ over $NH_3$- (Ek, 1993; Lumme, 1994). Nevertheless, the $NH_4^+$-N percolating from the O-horizon during spring snowmelt was effectively retained in the mineral soil (93 % of the annual input). This retention of $NH_4^+$-N may not have been entirely due to uptake by roots and micro-organisms, however, as $NH_4^+$ ions can be retained by cation exchange and some may have been converted to $NO_3^-$ through nitrification. Persson and Wirén (1995) have reported nitrification at a depth of 10-20 cm and deeper in acid mineral soil, although they found none in the humus layer.

In conclusion, the spruce canopy acted as a sink for N (retaining almost 50 % of total deposition) and there was little leaching of N beyond the B-horizon (7 % of the N in throughfall deposition). Because the amount of $NO_3^-$-N leaching from the B-horizon was low (<0.1 kg ha$^{-1}$ yr$^{-1}$) and represented such a small fraction of the amount arriving to the O-horizon (<2 % of $NO_3^-$-N and <1 % of inorganic N), the ecosystem clearly can not be considered N saturated (cf., Tietema *et al.* 1994). In the future, we intend to examine the effect of harvesting on $NO_3^-$-N leaching.

## Acknowledgments

The authors wish to thank Prof. Hannu Mannerkoski for comments to the manuscript. Ms. Seija Repo and Ms. Anita Pussinen for analyzing the water samples, Ms. Leena Karvinen for preparing the figures and tables, and Mr. Timo Pelkonen, Mr. Esko Kokkonen and Mr. Markku Tiainen for field work. We also wish to acknowledge the National Forest and Park Service for making the site available.

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# PAPER II

Piirainen, S., Finér, L. & Starr, M. 2002. Deposition and leaching of sulphate and base cations in a mixed boreal forest in Eastern Finland. Water, Air, and Soil Pollution, 131: 185-204.

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# DEPOSITION AND LEACHING OF SULPHATE AND BASE CATIONS IN A MIXED BOREAL FOREST IN EASTERN FINLAND

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(Received 15 June 1999; accepted 15 December 2000)

**Abstract.** Monthly fluxes of sulphate ($SO_4^{2-}$) and base cations ($Ca^{2+}$, $Mg^{2+}$, $K^+$) were studied from 1993 to 1996 as precipitation passed through forest vegetation and surface soil layers in an area receiving low and declining levels of atmospheric sulphate pollution. The canopy was dominated by mature Norway spruce (*Picea abies* Karsten) and the soil was a podzol developed on glacial till material. The mean annual bulk deposition of $SO_4^{2-}$ collected in the open was 136 $mol_c$ $ha^{-1}$ and that of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ was 44, 11 and 25 $mol_c$ $ha^{-1}$, respectively. The annual total throughfall deposition of $SO_4^{2-}$ was 318 $mol_c$ $ha^{-1}$ and that of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ was 151, 64 and 181 $mol_c$ $ha^{-1}$, respectively. Sulphate was the dominant anion accompanying the base cations leached from the canopy. More than half (58%) of the annual total throughfall deposition of $SO_4^{2-}$ was retained by the O-horizon and only 15% leached from below the B-horizon. The annual leaching of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ from below the B-horizon was 14, 25 and 9% of the annual total throughfall deposition, respectively. The transport of base cations through the soil was predominantely countered by $SO_4^{2-}$ anions.

**Keywords:** lysimeter, mature forest, percolate, podzol, throughfall

## 1. Introduction

Sulphate ($SO_4^{2-}$) sulphur has been a major anionic component in precipitation throughout much of Europe and North America since industrial times (e.g., Parker, 1983; Bredemeier, 1988; Lindberg, 1992). Because of the large reactive surface presented to the atmosphere, forest canopies are particularly efficient in trapping both wet and dry deposition (Ulrich, 1983; Bredemeier, 1988; Lindberg and Garten, 1988). The dry deposition of sulphur (S) is dominated by sulphur dioxide ($SO_2$) gas and $SO_4^{2-}$ salt aerosols (Lindberg, 1992). The scavenged $SO_2$ is oxidized to form $SO_4^{2-}$ anions in the moisture either on the foliage surfaces or inside the stomata. The deposition of S is therefore primarily the deposition of $SO_4^{2-}$ and it generates $H^+$ ions and acidification.

Studies in which the total deposition (wet + dry + cloud/fog) of S has been measured have shown that it is sufficiently similar to the flux of $SO_4^{2-}$-S in total throughfall (throughfall + stemflow) that the total throughfall $SO_4^{2-}$-S flux can be used as a measure of the total deposition of S to the forest floor (Lindberg *et*

*Water, Air, and Soil Pollution* **131:** 185–204, 2002.
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*al*., 1986; Hultberg and Grennfelt, 1992; Lindberg, 1992; Draaijers *et al*., 1997; Veltkamp and Wyers, 1997; Lindberg and Garten, 1988; Lindberg and Lovett, 1992). Because of the close 1:1 relationship between total deposition of S and total throughfall $SO_4^{2-}$-S, the difference between total throughfall and wet-only deposition is a reasonable measure of S dry deposition, particularly in areas where cloud/fog deposition is low.

Organic S is characteristically low to negligible in open precipitation (David *et al*., 1987) but may constitute up to half of total S concentrations in throughfall collected in North American hardwood and coniferous temperate forests (Homann *et al*., 1990; Mitchell *et al*., 1989). Unpublished data from the coniferous and mixed forests of Finland also indicate that up to half of total S concentrations in throughfall may be organic S (Ukonmaanaho, pers. communications).

The deposition of $SO_4^{2-}$ is not only important because of the associated acidity ($H^+$) but also because $SO_4^{2-}$ anions facilitate the leaching of cations. Sulphate is a relatively mobile anion and can accompany cations exchanged into solution from either vegetation or soil particle surfaces by $H^+$ ions from deposition (Johnson and Cole, 1977). Sulphate deposition can therefore enhance the leaching of cations, both acid ($H^+$ and $Al^{3+}$) and base cations: calcium ($Ca^{2+}$), magnesium ($Mg^{2+}$), potassium ($K^+$) and sodium ($Na^+$) (Cronan *et al*., 1978; Johnson and Cole, 1980; Singh *et al*., 1980; Foster *et al*., 1989; David *et al*., 1991a). The base cations are also major plant nutrients. The leaching of base cations from the foliage (which are initially derived from the soil) and the soil due to the deposition of $SO_4^{2-}$ anions therefore results in both acidification and loss of fertility of the soil.

Sulphur is also an essential plant nutrient, being primarily taken-up in the form of $SO_4^{2-}$ anions. The biological conversion of $SO_4^{2-}$ to organic S results in the removal of $SO_4^{2-}$ anions from solution and thereby reduced cation leaching. However, upon mineralization of the organic S, the original $SO_4^{2-}$ and associated acidity ($H^+$ ions) are returned to the soil solution. Another important process by which $SO_4^{2-}$ anions are removed from the soil solution is through sulphate adsorption. The adsorption of $SO_4^{2-}$ by soil is strongly related to the content of iron (Fe) and aluminium (Al) oxides and is an important process in the illuvial B-horizon of podzolic soils (Johnson, 1980; Johnson and Todd, 1983; Singh, 1984; Karltun and Gustafsson, 1993; Singh *et al*., 1980; Gustafsson and Jacks, 1993). Sulphate adsorption is, however, partly reversible (Harrison and Johnson, 1992; Karltun, 1995; Gobran *et al*., 1998). Lowered soil solution $SO_4^{2-}$ concentrations and increased pH would therefore tend to result in the desorption of previously adsorbed sulphate and the release of the associated acidity. This may be an important process controlling how soils respond when atmospheric inputs of S decrease (Harrison and Johnson, 1992; Bishop *et al*., 1995).

In response to emission controls there has been a significant progressive reduction in sulphate concentrations in precipitation since the 1980s throughout Europe (Meesenburg *et al*., 1995; Tørseth and Semb, 1995). Sulphate concentrations and loads in precipitation over Finland always have been relatively low (Helmisaari and

Mälkönen, 1989; Hyvärinen, 1990; Järvinen and Vänni, 1994, 1996) compared to western and central Europe, but a significant reduction in $SO_4^{2-}$ concentrations over the 1990s has also been observed (Ruoho-Airola *et al*., 1998; Ukonmaanaho *et al*., 1998).

As part of an ongoing project, the aim of which is to evaluate the effects of forest harvesting on the leaching of nutrients to surface waters (Finér *et al*., 1997), we have monitored bulk deposition in the open, throughfall, stemflow and soil percolate from a site in eastern Finland. In this paper we examine the role of $SO_4^{2-}$ in controlling the flux of base cations from the canopy and soil of a mature old growth forest before harvesting operations in an area of low and decreasing S deposition. Previous studies on cations leaching have been mainly carried out at sites with relatively high deposition loads of S or N.

## 2. Material and Methods

### 2.1. STUDY AREA

The data presented were collected from three 50 × 50 m sample plots located in a mature mixed boreal forest in eastern Finland (63°51′N, 28°58′E, 220 m a.s.l.). The site is 36 km from the nearest town, Nurmes (11,000 inhabitants), which has no significant S emissions, and 110 km from Kostamus and 440 km from St. Petersburg, which are both industrial cities in Russia. The prevailing wind directions are from the south (Finnish Meteorological Institute, 1991) and long-range air pollution is transported from Europe, southern Finland and occasionally from Kostamus and St. Petersburg.

Norway spruce (*Picea abies* Karsten) dominates the stand, but Scots pine (*Pinus sylvestris* L.), white and silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth) and European aspen (*Populus tremula* L.) are also present. Norway spruce accounts for 53%, Scots pine 33% and the deciduous tree species 14% of the total stand volume of 260 $m^3$ $ha^{-1}$. The projected canopy coverage of the stand is 54%. The forest is uneven-aged, with the oldest trees being 170-yr-old. The site type is *Vaccinium-Myrtillus* type according to the classification of Cajander (Cajander 1949; Mikola, 1982). The field layer vegetation is dominated by dwarf shrubs (*Vaccinium vitis-idaea* L. and *V. myrtillus* L.) and the bottom layer by feather mosses (*Pleurozium schreberi Brid.* and *Hylocomium splendens (Hedw.) B. S. and G.)*. The soil, derived from sandy till, is a weakly developed iron-podzol according to the classification of Kubiëna (1953) and a Haplic Podzol according to the FAO (1988) with a clay content of <2%. The stone content of the soil, determined by the rod penetration method of Viro (see Tamminen and Starr, 1994), is 28%. The underlying bedrock is composed of granodiorite. Some soil characteristics are presented in Table I. The mean annual air temperature averaged + 0.9 °C and the annual temperature sum (sum of daily mean temperature exceeding + 5 °C) for the

TABLE I

Some chemical properties of the soil at the study site (C was measured with LECO CHN-1000 analyzer, CEC = cation exchange capacity measured from 0.1 M $BaCl_2$ extract. The average soil stoniness (22%) for the E- and B-horizons has been taken into account when calculating amounts per hectare. Fe and Al fractionations are according to van Reeuwijk (1995). For further details see Finér *et al.* (1997)

| | | O-horizon | E-horizon | B-horizon |
|---|---|---|---|---|
| Thickness of horizon | (cm) | 3.1 | 6.8 | 13.9 |
| C | (%) | 45.5 | 1.4 | 1.9 |
| $pH_{H_2O}$ | | 4.0 | 4.2 | 5.0 |
| N | (% o.m.) | 1.3 | 1.8 | 1.6 |
| CEC | ($\mu$mol$_c$g$^{-1}$) | 261 | 37 | 13 |
| CEC | (kmol$_c$ ha$^{-1}$) | 14.2 | 9.2 | 10.9 |
| Base saturation | (%) | 66 | 11 | 22 |
| Organic-Fe[a] | ($\mu$mol g$^{-1}$) | 5.8 | 14.6 | 35.9 |
| Organic-Fe[a] | (kmol ha$^{-1}$) | 0.3 | 6.9 | 32.6 |
| Organic-Al[a] | ($\mu$mol g$^{-1}$) | 16.1 | 17.2 | 123.4 |
| Organic-Al[a] | (kmol ha$^{-1}$) | 0.8 | 7.9 | 115.5 |
| Amorphous-Fe[b] | ($\mu$mol g$^{-1}$) | 7.8 | 13.7 | 133.4 |
| Amorphous-Fe[b] | (kmol ha$^{-1}$) | 0.4 | 6.5 | 128.4 |
| Amorphous-Al[b] | ($\mu$mol g$^{-1}$) | 6.4 | 7.9 | 315.7 |
| Amorphous-Al[b] | (kmol ha$^{-1}$) | 0.3 | 3.9 | 310.1 |
| Crystalline-Fe[c] | ($\mu$mol g$^{-1}$) | 8.0 | 16.5 | 92.2 |
| Crystalline-Fe[c] | (kmol ha$^{-1}$) | 0.4 | 7.8 | 86.5 |
| Crystalline-Al[3] | ($\mu$mol g$^{-1}$) | 1.0 | 1.0 | 0.0 |
| Crystalline-Al[3] | (kmol ha$^{-1}$) | 0.0 | 0.5 | 0.0 |

[a] Sodium pyrophosphate extractable.

[b] Ammonium oxalate extractable – sodium pyrophosphate extractable.

[c] Sodium citrate-dithionate-bicarbonate extractable – ammonium oxalate extractable.

period 1993–1996 was 860 °C. A more detailed description of the study area is presented by Finér *et al.* (1997).

## 2.2. DEPOSITION AND SOIL WATER SAMPLING

Throughfall at the sample plots and bulk precipitation in a clearing nearby were collected during the four-year period, 1993–1996, using permanently open bulk collectors. For bulk precipitation in the open 5 plastic collectors (each 130.7 cm$^2$) were used during the snow-free period and three plastic bucket snow collectors (each 298.7 cm$^2$ for winter 1992–1993 and 1146.1 cm$^2$ thereafter) were used during the winter. The collectors were placed about 0.6 m above the ground during the snow-free period and 1.5 m above the ground in winter.

Throughfall was collected on each sample plot using 16 collectors during the snow-free period and 8 collectors for snow during winter. The same type of collectors as used for bulk deposition in the open were used and were placed systematically at equal distances along each side of the plots.

Stemflow was collected during 1993–1995 at two of the sample plots. At each plot, five white or silver birches, Scots pines and Norway spruces representing different diameter classes (>6 cm) were fitted with spiral type stemflow collectors made of silicon tubing.

The bulk precipitation and throughfall collectors were emptied once a week during the snow-free period and once a month during winter. On each sampling occasion the volume collected was recorded. Weekly or monthly bulk precipitation and throughfall samples were composited by plot for analysis. Stemflow was collected weekly and only during the snow-free period. The volume of the samples was recorded before being composited by species and plot for chemical analyses.

Soil water (percolate) was sampled at each sample plot using nine systematically placed (on a grid) zero-tension lysimeters installed at three depths: under the humus layer (O-horizon), under the eluvial (E) horizon (at 12 cm depth), and 23 cm below the boundary of eluvial and illuvial (B) horizons (at 35 cm depth). At plot 2 one lysimeter and at plot 3 three lysimeters under the B-horizon were rejected from the study because they did not collect water. The lysimeters under the O-horizon were of the Jordan type (Jordan, 1968); a plexiglass gutter, closed at the ends with a collecting area of 420 cm$^2$. The other soil water collectors were made of a plastic funnel (filled with quartz sand) with a collecting area of 299 cm$^2$ fitted to a 2 L plastic sample collection bottle. The particle size of the quartz sand varied from 0.8 to 1.2 mm. Sand (particle size 0.2–2 mm) was also the main fraction of the E- and B-horizons (41 and 39%, respectively). The lysimeters were emptied on the same day as the bulk precipitation and throughfall collectors and irregularly from October to April, when the water in collectors was not frozen. The samples from each lysimeter were analysed separately.

We use the term *total throughfall* to mean throughfall plus stemflow. The term *leaching* is used to mean the output of solutes in the percolate from below a soil horizon which exceeds the input to that horizon. The term *retention* is used when the input is greater than the output.

### 2.3. LABORATORY ANALYSES AND CALCULATIONS

All samples were kept cold and transported to laboratory the day after sampling. The bulk precipitation, throughfall and stemflow samples were filtered through a Schleiche and Schuell No. 589[1] filter and the soil percolate samples through a Schleiche and Schuell No. GF 52 glass wool filter. All the filtrates were stored in a freezer (–18 °C) before analysis. Sulphate concentrations were determined by

TABLE II

Mean (volume weighted) annual concentrations (+ standard error, n = 4) of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$, $K^+$ and $Na^+$ in bulk precipitation (BP), throughfall (TF), stemflow (SF) and soil percolate from under the different soil horizons $\mu mol_c\ L^{-1}$ during 1993–1996. For each ion values followed by the same letter do not significantly ($p > 0.0125$) differ from each other

| | $SO_4^{2-}$ | | $Ca^{2+}$ | | $Mg^{2+}$ | | $K^+$ | | $Na^+$ | |
|---|---|---|---|---|---|---|---|---|---|---|
| BP | 27.1 (1.0) | a | 8.6 (2.3) | a | 2.1 (0.4) | a | 5.0 (1.2) | ab | 8.7 (0.7) | a |
| TF | 64.6 (3.0) | c | 28.3 (1.3) | b | 13.0 (0.5) | b | 35.6 (1.0) | cd | 14.4 (1.0) | be |
| SF | 155.8 (12.0) | d | 130.6 (4.5) | e | 44.6 (4.7) | d | 183.6 (10.5) | e | 31.7 (3.6) | cf |
| O-horizon | 45.9 (2.8) | b | 66.3 (0.6) | d | 32.0 (1.6) | cd | 58.4 (11.6) | bd | 19.8 (2.7) | abd |
| E-horizon | 68.5 (6.4) | bc | 52.3 (3.0) | cd | 30.6 (3.6) | bc | 63.7 (10.1) | d | 25.8 (2.9) | ceg |
| B-horizon | 80.0 (7.4) | bc | 41.4 (4.4) | bc | 26.9 (1.4) | c | 39.2 (9.6) | acd | 29.4 (1.8) | fg |

ion chromatography (Dionex series 100 or 300 or 500) and $Ca^{2+}$, $Mg^{2+}$, $K^+$ and $Na^+$ concentrations were determined by flame atomic absorption spectrophotometer (Perkin-Elmer 5000). Detection limit (DL) of $SO_4^{2-}$ was 4.2 $\mu mol_c\ L^{-1}$ and that for $Ca^{2+}$, $Mg^{2+}$, $K^+$ and $Na^+$ 5.0, 1.6, 1.3 and 2.2 $\mu mol_c\ L^{-1}$, respectively. In the years 1993–1995 the DL of $Ca^{2+}$ was 15.0 $\mu mol_c\ L^{-1}$. If the concentration was under DL, a value of half DL was substituted. Only base cation concentrations in bulk precipitation were occasionally below DL. In the years 1993–1995 80% of $Ca^{2+}$ concentrations were below DL and thereafter 26%. The percentage of sample with concentrations below DL for $Mg^{2+}$ was 42%, for $K^+$ 24%, and for $Na^+$ 4%.

The dry deposition of S captured by the forest canopy was calculated as the difference between total throughfall and bulk deposition $SO_4^{2-}$. This was based on the assumption that total throughfall $SO_4^{2-}$ deposition is a measure of total (wet + dry) S deposition (Lindberg *et al.*, 1986; Hultberg and Grennfelt, 1992; Lindberg, 1992; Draaijers *et al.*, 1997; Veltkamp and Wyers, 1997). The fraction of S in bulk deposition originating from seawater, calculated from the S/Na ratio in seawater and assuming that the Na in rainwater originated only from the sea (Kulmala *et al.*, 1990), was estimated at 2.3%. The dry deposition of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ was estimated using the ratio of $Na^+$ in bulk deposition to $Na^+$ in total throughfall minus bulk deposition. This is based on the assumption that $Na^+$ is not exchanged in canopy (Ulrich, 1983; Draaijers *et al.*, 1997) and that the dry deposition characteristics of $Na^+$ are the same as for $Ca^{2+}$, $Mg^{2+}$ and $K^+$.

Monthly and annual deposition and leaching fluxes were calculated from plot mean concentrations (weighted by contribution of individual collectors), the volume of water collected on the plot, and the combined collection area of all collectors. Paired *t*-tests with a probability of Bonferroni's inequality (Snedecor and Cochran, 1980) were used to test if the mean concentrations and fluxes (of the 4 yr) of $SO_4^{2-}$ and base cations in bulk precipitation, throughfall and soil horizon percolate significantly differed from each other. Pearson product-moment correlation was used to examine the relationship between monthly $SO_4^{2-}$ and base cation concentrations in throughfall and soil percolate. SAS for Windows 6.11 (SAS® , 1994) was used for performing the statistical analyses.

## 3. Results

### 3.1. BULK PRECIPITATION, THROUGHFALL AND STEMFLOW

The mean annual precipitation during 1993–1996 was 502 mm (±52 mm) with monthly values ranging from 11 to 116 mm. Precipitation fell mainly as snow from November to April, and accounted for 45% of total annual precipitation. The amount of annual throughfall was 88–100% of precipitation and stemflow was almost negligible (1.3–2.7% of total throughfall). Monthly total throughfall ranged from 9 to 114 mm.

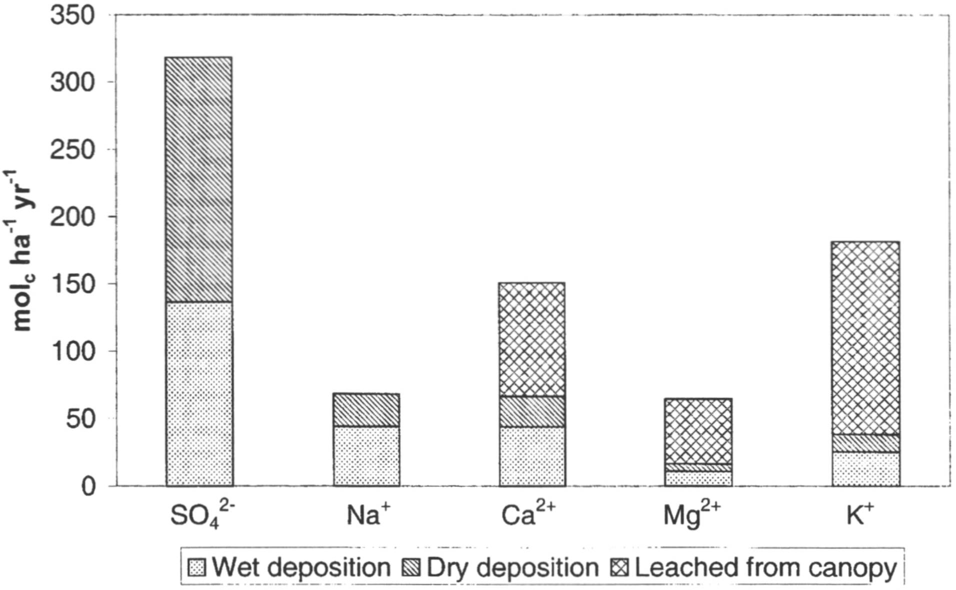

*Figure 1.* The mean total annual throughfall deposition of $SO_4^{2-}$, $Na^+$, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ to the forest floor, and its distribution between the bulk deposition, dry deposition and canopy leaching during 1993–1996. $SO_4^{2-}$ and $Na^+$ are not assumed to leach from the canopy.

The mean annual concentrations of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ in throughfall and stemflow were significantly higher than those in bulk precipitation (Table II). On a charge basis $Na^+$ and $Ca^{2+}$ were the dominant base cations in bulk precipitation and $K^+$ the dominant base cation in throughfall and stemflow. Over the study period, the mean annual bulk deposition of $SO_4^{2-}$ in the open was 136 $mol_c$ $ha^{-1}$ (±17 $mol_c$ $ha^{-1}$) and that in total throughfall 318 $mol_c$ $ha^{-1}$ (±9 $mol_c$ $ha^{-1}$) (Figure 1). Stemflow accounted for 7% of the annual total throughfall deposition of $SO_4^{2-}$. Dry deposition accounted for 57% (±5%) of the annual total throughfall deposition of $SO_4^{2-}$ to the forest floor. The mean annual bulk deposition of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ was 44, 11 and 25 $mol_c$ $ha^{-1}$ (±26, ±5, ±14 $mol_c$ $ha^{-1}$), respectively. The corresponding total throughfall depositions were 3.4, 6.0 and 7.2 times greater (Figure 1). Canopy leaching of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ accounted for 56% (±18%), 75% (±5%) and 79% (±10%), respectively, of the mean annual total throughfall depositions. The estimated dry deposition of $Ca^{2+}$ was 15% (±5%) of the mean annual total throughfall deposition and that for $Mg^{2+}$ and $K^+$ were 9% (±1%) and 7% (±2%), respectively. Stemflow accounted for 8% of $Mg^{2+}$, 14% of $Ca^{2+}$ and 10% of $K^+$ of the mean annual total throughfall.

Monthly total throughfall $SO_4^{2-}$ deposition was strongly ($p < 0.05$) correlated with total throughfall $Ca^{2+}$ and $Mg^{2+}$ deposition (r = 0.83 and r = 0.79, respectively). The corresponding correlation for $K^+$ was somewhat weaker (r = 0.70).

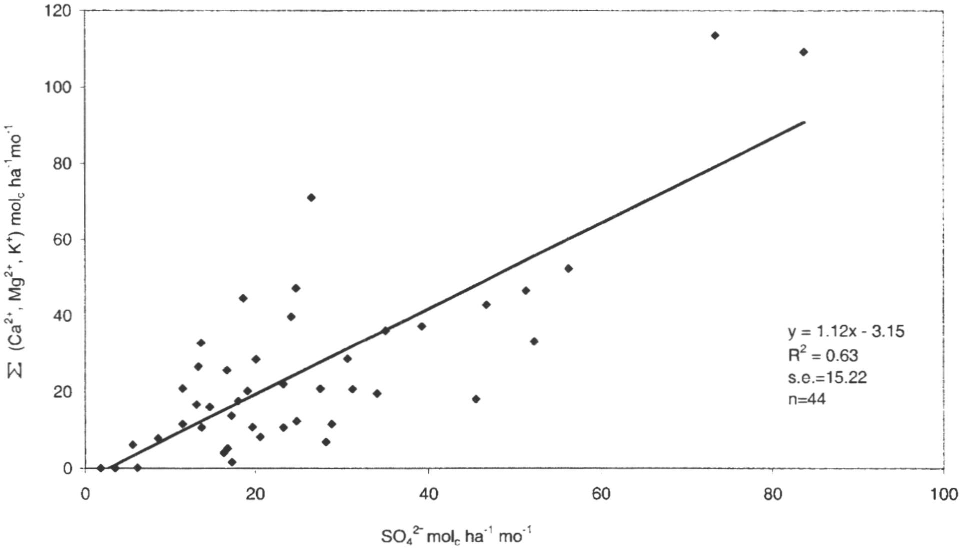

*Figure 2.* The regression line and the observed values showing the relationship between the monthly deposition of $SO_4^{2-}$ in total throughfall and the sum of base cations ($Ca^{2+}$, $Mg^{2+}$, $K^+$) leached from the canopy during 1993–1996.

Total throughfall $SO_4^{2-}$ deposition was also correlated with the sum of base cations leached from the canopy ($p < 0.05$) (Figure 2).

Monthly bulk and total throughfall depositions of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ peaked in the spring with a secondary peak in the autumn (Figure 3). Canopy leaching of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ was greater during the summer months than during the winter months, when precipitation fell as snow.

## 3.2. SOIL PERCOLATES

There was a marked progressive decrease in the amount of water as it percolated deeper in soil. On average, 64% (±10%) of the annual amount of throughfall passed through the O-horizon, 34% (±3%) through O + E-horizons and 12% (±4%) through O + E + B-horizons. The monthly percolate through the O-horizon ranged from 0 to 114 mm during the unfrozen period. The soil was usually frozen from December to March and no percolation occurred. Percolation was at its greatest at snowmelt, and occasionally also after heavy rainfall in summer and autumn. The mean monthly percolation from below the B-horizon varied from 0 to 9 mm during the summer months.

Sulphate concentrations in the O-horizon percolate were lower than those in throughfall (Table II). Calcium and $Mg^{2+}$ concentrations in the O-horizon percolate were higher than in throughfall but remained almost constant throughout the soil profile. On a charge basis $Ca^{2+}$ and $K^+$ were the dominant cations in the O-, E- and B-horizon percolates.

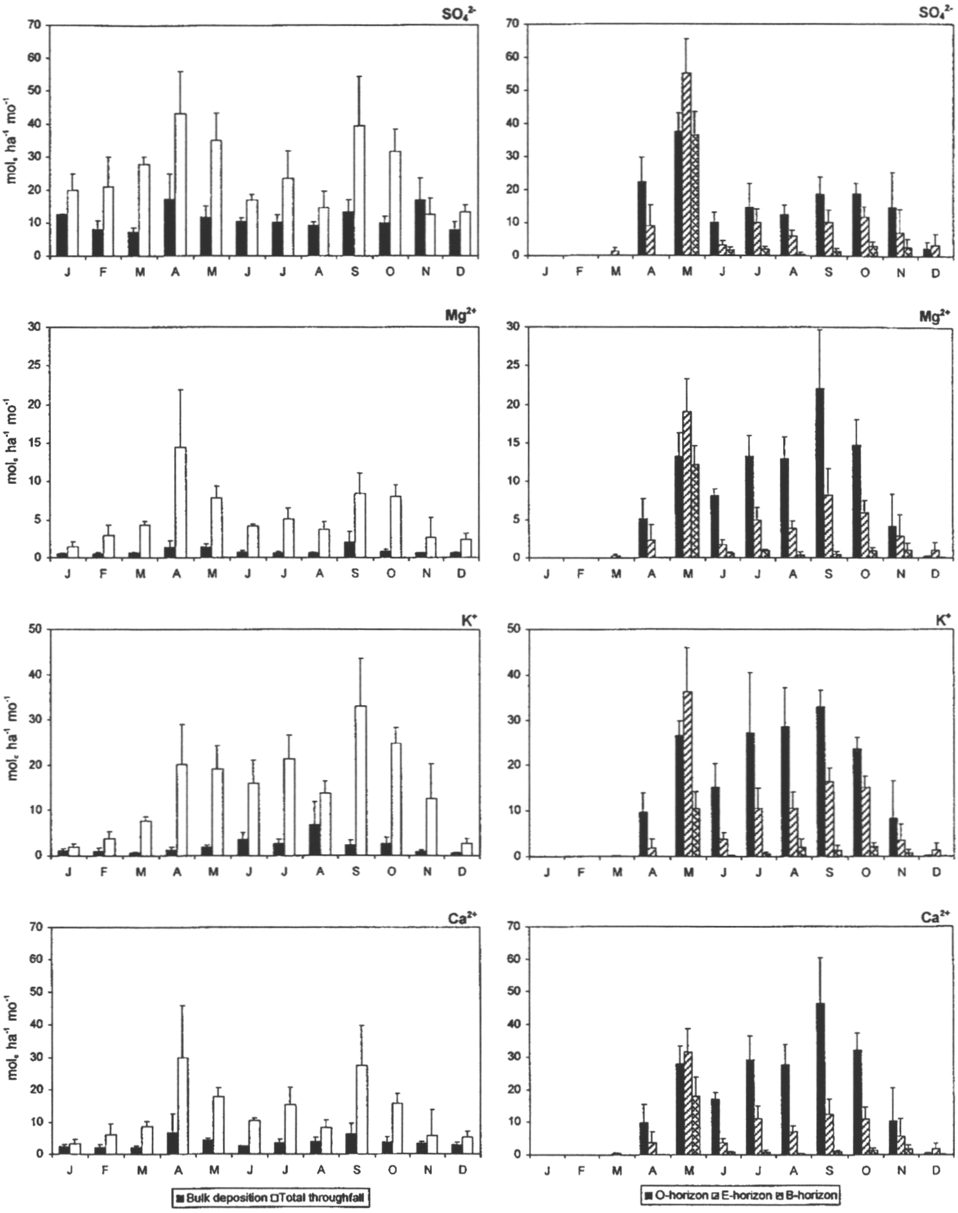

*Figure 3.* The mean monthly input of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ in the bulk deposition and total throughfall and the output in soil percolates from under the O-, E- and B-horizons during 1993-1996. (Bars are standard errors of the mean, n = 4).

TABLE III

Mean annual deposition (+standard error of the mean, n = 4) of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$, $K^+$ and $Na^+$ in total throughfall and the retention (–) or the leaching (+) from under the different soil horizons ($mol_c$ $ha^{-1}$ $yr^{-1}$) during 1993–1996

| | $SO_4^{2-}$ | $Ca^{2+}$ | $Mg^{2+}$ | $K^+$ | $Na^+$ |
|---|---|---|---|---|---|
| Total throughfall | 318 (5) | 151 (9) | 64 (3) | 181 (10) | 68 (3) |
| Soil retention/leaching | | | | | |
| O-horizon | –184 (7) | +43 (14) | +28 (8) | –14 (27) | –12 (5) |
| E-horizon | –27 (13) | –111 (17) | –45 (8) | –72 (22) | –16 (3) |
| B-horizon | –60 (8) | –59 (4) | –32 (5) | –80 (8) | –21 (5) |
| Percolated below B-horizon | 47 (7) | 23 (4) | 16 (3) | 15 (3) | 20 (3) |

The mean annual retention of $SO_4^{2-}$ by the O-horizon was 184 $mol_c$ $ha^{-1}$ (Table III). A further 27 $mol_c$ $ha^{-1}$ was retained by the E-horizon and 60 $mol_c$ $ha^{-1}$ by the B-horizon. Only 15% of the annual total throughfall deposition of $SO_4^{2-}$ appeared in the percolate from below the B-horizon. Most of the annual leaching of $SO_4^{2-}$ occurred during the spring months, when snowmelt was taking place (Figure 3).

Calcium and $Mg^{2+}$ leaching from the O-horizon exceeded annual total throughfall inputs (Table III). However, the $Ca^{2+}$ and $Mg^{2+}$ leached from the O-horizon was subsequently retained in the E-horizon. Leaching of $Ca^{2+}$ from below the B-horizon averaged 15% and that of $Mg^{2+}$ 25% of total throughfall inputs. Some of the $K^+$ input in total throughfall was retained in the O-horizon, but most passed into the mineral soil. The leaching of $K^+$ from below the B-horizon was 8% of the total throughfall input. Most of the annual leaching of base cations from below the B-horizon also occurred during the spring months (Figure 3).

There was a significant positive correlation ($p < 0.05$) between the monthly leaching of $SO_4^{2-}$ and the sum of base cations ($Ca^{2+}$ + $Mg^{2+}$ + $K^+$) for each soil horizon (Figure 4). However, the charge balance of the sum of base cations exceeded that of $SO_4^{2-}$ in the percolate from below the O- and E-horizons, indicating that anions other than $SO_4^{2-}$ were involved in the leaching of base cations from these horizons. The charge balance of the sum of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ and $SO_4^{2-}$ in the percolate from below the B-horizon was, however, nearly balanced, indicating that base cation leaching from below the horizon was primarily controlled by $SO_4^{2-}$ leaching.

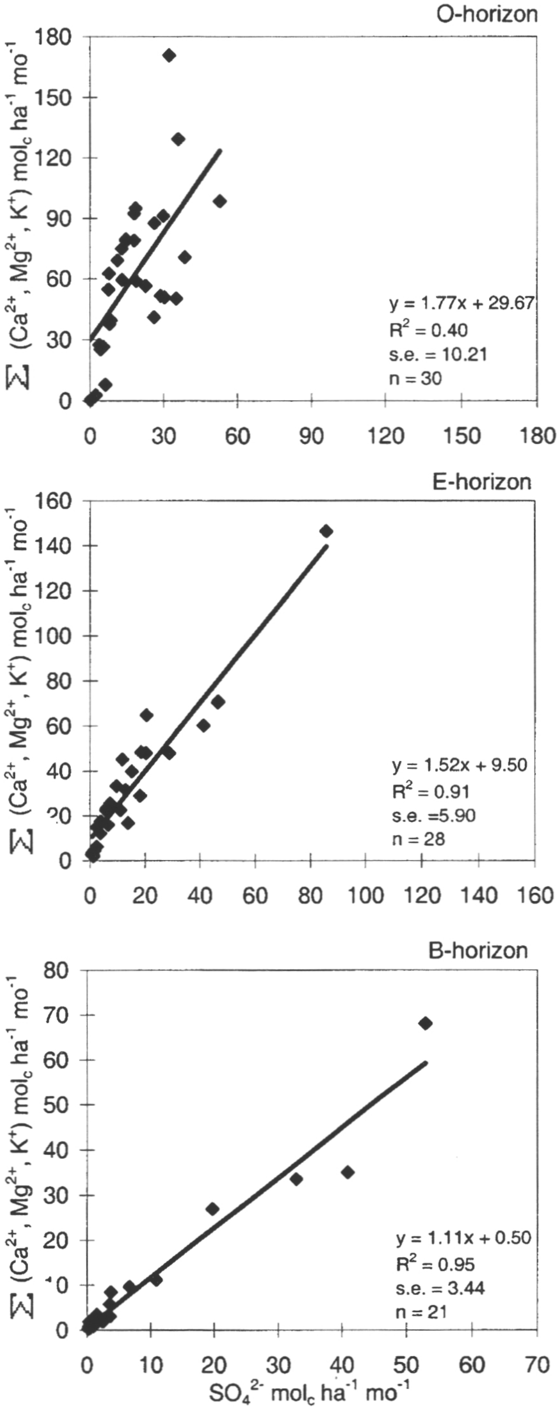

*Figure 4.* The regression line and the observed values showing the relationship between the monthly deposition of $SO_4^{2-}$ and the sum of base cations ($Ca^{2+}$, $Mg^{2+}$, $K^+$) in soil percolates from under different soil horizons (O-, E- and B-horizons) during 1993–1996.

## 4. Discussion

### 4.1. CANOPY WASHOFF AND LEACHING

The annual total throughfall (i.e., throughfall + stemflow) deposition of $SO_4^{2-}$ in our stand (318 $mol_c$ $ha^{-1}$ $yr^{-1}$) was at the lower end of the range of 130–2500 $mol_c$ $ha^{-1}$ $yr^{-1}$ reported for coniferous forests for North America, and Western and Northern Europe (Helmisaari and Mälkönen, 1989; Hultberg and Grennfelt, 1992; Lindberg and Lovett, 1992; Lindroos *et al.*, 2000). Estimated dry deposition of S accounted for 57% of total deposition of S in our stand. This fraction was slightly higher than the range of 10–50% reported by Lindberg and Lovett (1992) for the North American forests but within the range of 50–70% reported for Norway spruce dominated stands in Germany and the Nordic countries (Bredemeier, 1988; Hultberg and Grennfelt, 1992). The proportion of S dry deposition collected in the open with bulk collectors has been estimated to be 35–45% in Finland (Tuovinen *et al.*, 1990).

The proportions of gaseous $SO_2$, aerosol $SO_4^{2-}$ and particulate $SO_4^{2-}$ that make-up total S dry deposition to the forest floor varies with location (Lindberg and Lovett, 1992). In forests which are distant from local emission sources, such as our stand, or have a significant seasalt impact, particulate S deposition dominates (Lindberg and Lovett, 1992). Plant canopies take up gaseous sulphur through the stomata and by sorption to leaf and bark surfaces (Murphy and Sigmon, 1990). This gaseous sulphur is oxidised to sulphate in the forest canopy and washed-off as aerosol and particulate sulphate during rainfall episodes (Lindberg *et al.*, 1986; Lindberg and Lovett, 1992; Draijeers *et al.*, 1997). Uptake of dry S deposition by the foliage may occur, but this is probably offset by the foliar leaching of S which is derived from the soil (Draijeers *et al.*, 1997).

At our site, the bulk deposition of $Mg^{2+}$ was at the lower end and that of $Ca^{2+}$ and $K^+$ in the middle of the ranges reported for monitoring stations in Northern Europe (Bernes, 1993; Järvinen and Vänni, 1994, 1996; Lindroos *et al.*, 1999) and North America (Ragsdale *et al.*, 1992). The base cations in total throughfall originate from wet and dry deposition and from the leaching of foliage and bark. The dry deposition of base cations ($Ca^{2+}$, $Mg^{2+}$, $K^+$ and $Na^+$) occurs only in particulate form. In our study the dry deposition of base cations accounted for 14% of the total deposition of base cations. The estimation of base cation dry deposition was based on the assumption that the dry deposition characteristics of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ are similar to those of $Na^+$. This assumption is only valid if the particle size distributions and therefore the deposition rates of these ions are similar. Draijeers *et al.* (1997) found that the estimates of dry deposition calculated in the same way as we did corresponded fairly well with the amount of dry deposition estimated with other methods, indicating that the assumption made above is valid. Potassium was leached from the canopy in greater amounts than $Ca^{2+}$ and $Mg^{2+}$ in our stand. This has also been shown to occur in other studies (Hyvärinen, 1990; Hansen, 1996;

Draijeers *et al.*, 1997). This is because K primarily occurs in ionic form within plant cells, whereas Ca and Mg are more associated with cell wall tissue. Our results indicated greater leaching of $K^+$ and other base cations during the growing season than during the dormant season while Hansen (1996) did not observe any differences in $K^+$ leaching between seasons in a Norway spruce stand in Denmark. Our lower canopy leaching of base cations during winter was explained by the colder winters (lower reactivity) and snowfall.

The close 1:1 relationship between the monthly sum of $K^+$, $Ca^{2+}$ and $Mg^{2+}$ leached from the canopy and the deposition of $SO_4^{2-}$ in total throughfall indicated that base cation leaching from the canopy is strongly controlled by the amount of $SO_4^{2-}$ deposition. Protons associated with the $SO_4^{2-}$ ions in total throughfall and those produced in foliage as a result of the $SO_2$ oxidation are exchanged for base cations in the canopy which then appear in throughfall (Ulrich, 1983; Bredemeier, 1988, Westling *et al.*, 1995; Draijeers *et al.*, 1997). In areas of high ammonium ($NH_4^+$) deposition, the uptake of $NH_4^+$ in the canopy is countered by the leaching of base cations (Draaijers *et al.*, 1997). In our study, the canopy uptake of $NH_4^+$ ions was small (58 $mol_c$ $ha^{-1}$ $yr^{-1}$, Piirainen *et al.*, 1998) and partly balanced by the uptake of nitrate ($NO_3^-$) ions. Ammonium therefore does not play a significant role in the canopy leaching of base cations observed in our stand, as could be expected from its remote location.

### 4.2. LEACHING FROM SOIL

In this study the amount of percolate water was measured with zero-tension lysimeters. Although this is the traditional method and has widely been used in leaching studies, the reliability of hydrological fluxes determined with zero-tension lysimeters may be questioned. Forest soils are non-uniform because of the presence of soil horizons, possible strata in parent material, and the pattern in tree stem location and roots. The flow of water in forest soils can thus be expected to show relatively large spatial variation. But perhaps more importantly, simply installing the lysimeters, which involves cutting roots and excavating a hole in the soil, will disturb pore connectivity and thus the flow of water. Furthermore, a boundary will be formed between the intact soil and quartz sand in the top of lysimeter, which may influence the downward movement of water. Conversely, the lysimeters may not be able to handle all the water moving downwards through the soil during snowmelt. Nevertheless, we consider that lysimeters did collect most of the downward moving water, since the mor humus and the soil in the E- and B-horizons consisted mainly (60%) of macro pores (Piirainen, unpublished data). Also the texture of the intact mineral soil did not differ much from the quartz sand used in the lysimeters. An alternative method for computing the water flow in soil would have been to use a hydrological model as has been done by Mitchell *et al.* (1992). However, for a model, many parameters are needed for soil, vegetation and climate, and some of them have to be estimated indirectly. Thus the results of models need not to be more realistic than those obtained with zero-tension lysimeters.

In our stand some 42% of the annual throughfall $SO_4^{2-}$ deposition passed through the humus layer. This suggests that $SO_4^{2-}$ was rapidly taken up by plant roots and microbes in the humus layer (Strickland *et al.*, 1986; Fitzgerald *et al.*, 1988; David *et al.*, 1991b). Since 51% of the fine roots were located in the humus layer (Finér *et al.*, unpublished), most of the uptake can be expected to take place from the humus layer. Studies indicate that the annual uptake of S by coniferous forests varies from 40 to 190 $mol_c$ $ha^{-1}$ (Nihlgård, 1972; Finér, 1991). The retention of ca. 60 $mol_c$ $SO_4^{2-}$-S $ha^{-1}$ in the humus layer we observed could therefore be explained by the uptake. Laboratory experiments have also shown that the humus layer can retain a high fraction of added $SO_4^{2-}$ (Berdén and Nilsson, 1996). However, net annual leaching of $SO_4^{2-}$ from the humus layer in the field has been reported (Helmisaari and Mälkönen, 1989; Mitchell *et al.*, 1992), indicating net S mineralization.

The leaching of $SO_4^{2-}$ from below the B-horizon was only 15% of the total annual throughfall deposition of $SO_4^{2-}$. Most of this leaching occurred in spring, during snowmelt. The retention of $SO_4^{2-}$ in podzols has been observed in several studies (e.g., Singh *et al.*, 1980; Gustafsson and Jacks, 1993; Gobran *et al.*, 1998). The adsorption of $SO_4^{2-}$ in the soil mainly takes place in the B-horizon and onto the surfaces of aluminium and iron oxides and hydroxides that have accumulated there, amorphous forms in particular (e.g. Johnson, 1980; Johnson and Todd, 1983; Singh, 1984; Gustafsson and Jacks, 1993; Karltun and Gustafsson, 1993). The reaction is partly reversible and pH dependent, increasing with decreasing pH (Harrison and Johnson, 1992; Karltun, 1995; Gobran *et al.*, 1998). We also found that most of the retention of $SO_4^{2-}$ in the soil took place in the B-horizon where there were relatively high concentration of amorphous Fe and Al (Table I) which is typical for Fennoscandinavian podzols (e.g. Olsson and Melkerud, 1989; Ilvesniemi *et al.*, 2000).

The annual leaching of base cations ($Ca^{2+}$, $Mg^{2+}$, $K^+$, and $Na^+$) below the B-horizon was 16% of total throughfall inputs (Table III). This indicated that considerable retention of base cations takes place within the humus layer and soil or take-up by the vegetation. The annual leaching of $Ca^{2+}$ and $Mg^{2+}$ from the O-horizon exceeded total throughfall inputs. This suggested that decomposition resulted in a net release of $Ca^{2+}$ and $Mg^{2+}$. However, $Ca^{2+}$ and $Mg^{2+}$ leached from the O-horizon was subsequently retained in the mineral soil. Potassium and $Na^+$ showed retention in all soil horizons, including the O-horizon. Values for the annual uptake of K by boreal coniferous forests varied from 220 to 390 $mol_c$ $ha^{-1}$ (Mälkönen, 1975; Finér, 1989; Helmisaari, 1995). If we assume similar rates of K uptake at our stand, total throughfall deposition of K would account for 41–94% of annual uptake of K. Since 66–87% of total throughfall was derived from foliar leaching this indicated relatively tight cycling of K. Annual uptake values for Ca and Mg reported by boreal coniferous forests were 360–1210 and 230–400 $mol_c$ $ha^{-1}$, respectively (Mälkönen, 1975; Finér, 1989; Helmisaari, 1995).

Again assuming similar rates of uptake at our stand, total throughfall deposition could only provide 11–46% of the uptake for Ca and 14–32% for Mg.

The adsorption of base cations in the mineral soil is also related to the adsorption $SO_4^{2-}$. By removing $SO_4^{2-}$ ions from the soil solution anions necessary to accompany cations are reduced and therefore base cations are withdrawn from solution (Johnson and Cole, 1977; David *et al.*, 1991a). These base cations could be retained by exchange sites or taken-up by roots. The significant positive correlation between the output of $SO_4^{2-}$ and the sum of base cations from below each soil horizon (Figure 4) suggested that $SO_4^{2-}$ is important in controlling the leaching of base cations even though $SO_4^{2-}$ anion balanced less than 50% of the base cations charge in the O- and E-horizons. Sulphate anion balanced nearly all the base cations leaching from below the B-horizon. Organic anions were probably also important in the leaching of base cations from the O- and E-horizons. Krug and Isaacson (1984) and Lundström (1993) have shown that there is greater leaching of organic acids from the O-horizon than from the mineral soil horizons of podzolic soils. Unlike soils with high levels of nitrification (Foster *et al.*, 1989; Carnol *et al.*, 1997), $NO_3^-$ production at our study site was very small (Piirainen *et al.*, 1998) and $NO_3^-$ leaching therefore played an insignificant role in the leaching of base cations.

## Acknowledgements

The authors wish to thank Ms. Seija Repo and Ms. Anita Pussinen for analyzing the water samples, Ms. Leena Karvinen for preparing the figures and tables and Mr. Timo Pelkonen, Mr. Esko Kokkonen and Mr. Markku Tiainen for the field work and Mr. Jaakko Heinonen for advice on statistical analyses. We also wish to acknowledge the Finnish Forest and Park Service for making the site available for our research project. We also thank Prof. Hannu Mannerkoski and the anonymous reviewers for their comments to the manuscript.

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# PAPER III

Piirainen, S., Finér, L., Mannerkoski, H. & Starr, M. 2002. Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant and Soil 239: 301-311.

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*Plant and Soil* **239:** 301–311, 2002.
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# Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons

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Received 18 April 2001. Accepted in revised form 18 December 2001

*Key words:* ammonium, deposition, DOC, DON, harvesting, lysimeter, nitrate, percolate

## Abstract

Effects of clear-cutting on the dissolved fluxes of organic C (DOC), organic N (DON), $NO_3^-$ and $NH_4^+$ through surface soil horizons were studied in a Norway spruce dominated mixed boreal forest in eastern Finland. Bulk deposition, total throughfall and soil water from below the organic (including understorey vegetation and, after clear-cutting, also logging residues), eluvial and illuvial horizons were sampled weekly from 1993 to 1999. Clear-cutting was carried out in September 1996. The removal of the tree canopy decreased the deposition of DOC and DON to the forest floor and increased that of $NH_4^+$ and $NO_3^-$ but did not affect the deposition of total N (DTN, <3 kg ha$^{-1}$ a$^{-1}$). The leaching of DOC and DON from the organic horizon increased over twofold after clear-cutting (fluxes were on an average 168 kg C and 3.3 kg N ha$^{-1}$ a$^{-1}$), but the increased outputs were effectively retained in the surface mineral soil horizons. Inorganic N deposition was mainly retained by the logging residues and organic horizon indicating microbial immobilization. Increased $NO_3^-$ formation reflected as elevated concentrations in the percolate from below the mineral soil horizons were observed especially in the third year after clear-cutting. However, the changes were small and the increased leaching of DTN from below the illuvial horizon remained small (<0.4 kg ha$^{-1}$ a$^{-1}$) and mainly DON. Effects of clear-cutting on the transport of C and N to surface waters will probably be negligible.

## Introduction

The amounts of the dissolved forms of organic carbon (DOC) and total nitrogen (DTN) flowing annually through undisturbed boreal forest ecosystems are very small compared to the pools of C and N in the vegetation and soil. The C and N in soil is largely bound to dead organic matter and the decomposition and dissolution of this pool are the main source of DOC and dissolved organic nitrogen (DON) in soil water (e.g. Guggenberger and Zech, 1993; Qualls et al., 1991; Sollins and McCorison, 1981). Other possible sources are the leaching from litter (Qualls and Haines, 1991) or exudates from living roots, microorganisms and fungi. DON usually dominates DTN fluxes through soil in undisturbed forests (Currie et al., 1996; Fahey et al., 1985; Piirainen et al., 1998; Qualls et al., 1991; Smolander et al., 2001). In boreal coniferous forests, tree growth is generally limited by the availability of inorganic forms of N i.e. nitrate ($NO_3^-$) and ammonium ($NH_4^+$), usually resulting in tight cycling and low leaching losses (Tamm, 1991). In soils with high N availability, some leaching of $NO_3^-$ can occur (e.g. Gundersen et al., 1998).

Clear-cutting is the major disturbance in boreal forests of Fennoscandia. It results in a sudden increase in organic matter to the forest floor in the form of logging residues and severed roots, and can increase the risk of N leaching, particularly $NO_3^-$ (e.g. Kubin, 1998; Likens et al., 1970; Rosén and Lundmark-Thelin, 1987; Vitousek et al., 1979). The input of N

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can initiate eutrophication of surface waters (e.g. Ahtiainen, 1992; Lepistö et al., 1995; Likens et al., 1970; Tamm et al., 1974; Wiklander et al., 1991). Leaching of $NO_3^-$ can also induce leaching of base cations from soil (Likens et al., 1969). However, the leaching of N has been found to be relatively short-term when the responses of understorey vegetation and microbial communities have been fast enough to stabilize the inorganic N fluxes (Barg and Edmonds, 1999).

We know little about the effects of clear-cutting on DOC and DON formation and leaching and most of the studies have been carried out in temperate (Qualls et al., 2000; Robertson et al., 2000; Sollins and McCorison, 1981; Tate and Meyer, 1983) rather than in boreal forests (Smolander et al., 2001). Increased DOC or DON concentrations in solutions beneath the logging residues or in the soil have been observed in those studies. In laboratory conditions, microbial activity has been increased by elevating moisture and temperature in the organic horizon, which has further increased DOC production (Christ and David, 1996). According to McLaughlin et al. (2000), this increased DOC formation probably indicates incomplete organic matter oxidation to $CO_2$. The increased fluxes of DON and DOC can affect nutrient availability in soil. DOC as negatively charged compound can transport nutrients and heavy metals through the soil and affect their solubility (Tipping et al., 1990) and also to acid base status of soil solution (Cronan and Aiken, 1985; Johnson and Cole, 1980). DOC is also central to the process of podzolization, leading to the development of the eluvial E- and illuvial B-horizons (Lundström et al., 2000).

The influence of clear-cutting on DOC exports from soil to streams has usually been found to be negligible or the export has even diminished (reviewed by Hope at al., 1994). The reason for this can be related to the reduced DOC formation attributed to decreased litterfall, as speculated by Tate and Meyer (1983). However, the DOC concentration in the stream does not directly reflect the DOC production from the organic matter, since the retention by the mineral soil can be high. How big is the retention capacity of mineral soil after clear-cutting is not known. Soil C pools have also been found to be reduced after clear-cutting of boreal forests (Olsson et al., 1996), which might also indicate elevated release of C as $CO_2$ into the atmosphere with decreased loss by leaching.

The aims of this study were to (1) find out the changes in the fluxes of DOC and DTN to and below the surface soil horizons (2) find out if the leaching of $NO_3^-$ is increased and (3) discuss if the changes in soil C or N pools due to leaching losses after clear-cutting of a Norway spruce dominated mixed boreal forest. The study is a part of an ongoing project that is evaluating the effects of clear-cutting and site preparation on nutrient cycling, soil, soil water, and stream water quality and yield in forested headwater catchments (Finér et al., 1997).

## Materials and methods

### *Study area*

The data presented were collected from three 50 m × 50 m sample plots during January 1993 – December 1999. Plots are located in the Kangasvaara catchment in eastern Finland (63° 51′N, 28° 58′E, 220 m a.s.l.) (Finér et al., 1997). The forest was an old-growth mixed coniferous forest stand dominated by Norway spruce (*Picea abies* Karsten), but Scots pine (*Pinus sylvestris* L.), white and silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth) and European aspen (*Populus tremula* L.) were also present (Table 1). In September 1996 clear-cutting was done manually at two of plots and one remained uncut. Stems with bark were removed and logging residues (branches, leaves and tree tops <8 cm diameter) were left evenly distributed on the cut plots.

The site type was classified as the *Vaccinium-Myrtillus* type (Cajander, 1949; Mikola, 1982). The field layer vegetation was dominated by dwarf shrubs (*Vaccinium vitis-idaea* L. and *V. myrtillus* L.) and the bottom layer by feather mosses (*Pleurozium schreberi Brid.* and *Hylocomium splendens* (Hedw.) B. S & G.). The forest floor consists of a litter and mor humus layer with an average thickness of 3 cm. The soil, derived from sandy till, is a weakly developed iron-podzol with a clay content of < 2%. The stoniness of the soil, determined by the rod penetration method of Viro (see Tamminen and Starr, 1994), was 28%. Some detailed soil characteristics for plots are presented in Table 2.

The mean annual air temperature on the site averaged +0.7°C and the annual temperature sum (sum of daily mean temperature exceeding +5°C) 903°C for the period 1993–1999. A more detailed description of the study area is presented by Finér et al. (1997).

*Table 1.* Tree stand characteristics of the uncut plot and cut plots 1 and 2 and carbon (C) and nitrogen (N) pools in trees and fine roots before harvesting in 1996. The C and N amounts in logging residues are estimated from the element content of tree crown (Finér et al., 2002)

| | uncut plot | Plot 1 | Plot 2 |
|---|---|---|---|
| Stem volume with bark, m$^3$ ha$^{-1}$ | 260 | 230 | 290 |
| Spruce, % | 78 | 50 | 33 |
| Pine, % | 11 | 36 | 51 |
| Deciduous, % | 11 | 14 | 16 |
| C in above ground tree biomass, 1000 kg ha$^{-1}$ | 79.05 | 72.41 | 88.74 |
| N in above ground tree biomass, kg ha$^{-1}$ | 291 | 251 | 284 |
| C in above ground logging residues, 1000 kg ha$^{-1}$ | – | 16.97 | 18.54 |
| N in above ground logging residues, kg ha$^{-1}$ | – | 166 | 179 |
| C in vegetation fine roots, 1000 kg ha$^{-1}$ | 3.43 | 2.95 | 2.87 |
| N in vegetation fine roots, kg ha$^{-1}$ | 56 | 47 | 43 |

*Table 2.* Some chemical properties of the soil at the uncut plot and cut plots 1 and 2. The average soil stoniness (28%) for the E- and B-horizons has been taken into account when calculating the amounts per hectare

| | O-horizon | | | E-horizon | | | B-horizon | | |
|---|---|---|---|---|---|---|---|---|---|
| | uncut | plot 1 | plot 2 | uncut | plot 1 | plot 2 | uncut | plot 1 | plot 2 |
| Thickness of horizon, cm | 3.0 | 3.5 | 2.8 | 7.1 | 6.0 | 7.2 | 12.8 | 13.7 | 15.3 |
| pH$_{H_2O}$ | 4.22 | 3.84 | 3.93 | 4.22 | 4.24 | 4.22 | 4.95 | 4.98 | 5.20 |
| C, % | 38.8 | 49.8 | 47.8 | 1.5 | 1.6 | 1.1 | 2.2 | 2.2 | 1.4 |
| C, 1000 kg ha$^{-1}$ | 18.74 | 26.16 | 18.83 | 7.50 | 6.19 | 5.99 | 17.87 | 18.13 | 17.45 |
| N, % | 1.06 | 1.17 | 1.23 | 0.07 | 0.07 | 0.05 | 0.11 | 0.09 | 0.07 |
| N, kg ha$^{-1}$ | 528 | 625 | 500 | 364 | 274 | 300 | 859 | 747 | 875 |
| Amorphous-Fe[1], $\mu$mol g$^{-1}$ | 6.7 | 9.5 | 7.2 | 16.4 | 13.9 | 10.9 | 132.2 | 129.7 | 138.5 |
| Amorphous-Fe[1], kmol ha$^{-1}$ | 0.4 | 0.5 | 0.3 | 8.1 | 5.4 | 6.1 | 107.3 | 109.0 | 169.0 |
| Amorphous-Al[1], $\mu$mol$^{-1}$ | 5.6 | 6.3 | 7.3 | 11.9 | 4.0 | 7.7 | 234.0 | 342.1 | 370.9 |
| Amorphous-Al[1], kmol ha$^{-1}$ | 0.3 | 0.3 | 0.3 | 5.9 | 1.5 | 4.3 | 190.1 | 287.6 | 452.6 |

[1] Ammonium oxalate extractable – sodium pyrophosphate extractable.

### *Deposition and soil water sampling*

Bulk precipitation in a nearby clearing and throughfall on the sample plots were collected during a four-year period before clear-cutting (1993–1996) and a three-year period afterwards (1997–1999) using permanently open bulk precipitation collectors. For bulk precipitation in the open, five plastic collectors (each 131 cm$^2$) were used during the snow-free period and three bigger ones (each 299 cm$^2$ for winter 1992–1993 and 1146 cm$^2$ thereafter) in winter. The collectors were placed 0.6 m above ground level during the snow-free period and 0.9 m higher in winter. Throughfall was collected at each sample plot using 16 collectors during the snow-free period and 8 collectors for snow in winter. The same type of collectors as used for bulk precipitation in the open were used and placed systematically at equal distances along each side of the plots. After clear-cutting, throughfall was collected only at the uncut plot. Stemflow was collected during 1993–1995 at two of the sample plots. For the following years, stemflow fluxes were taken as the average of the measured years. At each plot, five white or silver birches, Scots pines and Norway spruces representing different breast height diameter classes (>6 cm) were fitted with a spiral type stemflow collectors made of silicon tubing.

The bulk precipitation and throughfall collectors were emptied once a week during the snow-free period and once a month during winter. On each sampling occasion, the volume of water collected was recorded. Weekly or monthly bulk precipitation and throughfall samples were pooled by plot for analyses. Stemflow was collected weekly and only during the snow-free period.

Soil water (percolate) was sampled at each sample plot using nine zero-tension lysimeters installed in the plots at each of three depths: below the organic (O), below the eluvial (E) (at 14, 11 and 11 cm depth from the soil surface at the uncut plot and cut plots 1 and 2, respectively) and below the upper part of the illuvial (B) horizon (36, 37 and 32 cm depth from soil surface, respectively). The three lysimeters below the B-horizon in the cut plot 2 and two lysimeters below the 0-horizon in the cut plot 1 did not or only before July 1998 collect water and were therefore rejected from further calculations. The lysimeters below the O-horizon were of the Jordan type (Jordan, 1968); a plexiglass gutter, closed at the ends with a collecting area of 420 cm. The other soil water collectors were made of a polythene plastic funnel, filled with quartz sand, with a collecting area of 299 cm$^2$ fitted to a 2 L sample collection bottle. The particle size of the quartz sand varied from 0.8 to 1.2 mm. The main soil texture class in the E-and B-horizons was also sand (41 and 39%, respectively) allowing free water flow from soil to lysimeter as discussed by Piirainen et al. (2002). The lysimeters were emptied on the same day as the precipitation and throughfall collectors when the water in the soil was not frozen (mostly from April to October) and they were not bulked for analysis. Care was taken during the clear-cutting operation not to disturb the lysimeter installations.

We use the terms *total throughfall* to mean throughfall plus stemflow, *leaching* to mean output of percolate in excess of input for a soil horizon and *retention* when inputs are greater than outputs. The water and element fluxes below the O-horizon are affected by the processes in organic horizon, understorey vegetation and, after clear-cutting, also logging residues.

### *Laboratory analyses and data handling*

All samples were kept cold and transported to the laboratory the day after sampling. The bulk precipitation, throughfall and stemflow samples were filtered through a Schleiche & Schuell no. 589[1] paper filter and soil percolate through a Schleiche & Schuell no. GF 52 glass wool filter. All the filtrates were stored in a freezer (−18 °C) until further analyses. Nitrate concentrations were determined by ion chromatography (Dionex series 100 or 300 or 500) and $NH_4^+$ and dissolved total N (DTN) were determined by flow injection analysis (Tecator FIA-Star 5020). Dissolved organic N (DON) was calculated by subtracting dissolved inorganic N (DIN; sum of $NH_4^+$-N and $NO_3^-$-N) concentrations from DTN. Dissolved organic carbon (DOC) was measured with a Shimadzu TOC-5000 analyzer. The detection limit (DL) for $NO_3^-$-N varied from 0.01 to 0.02 mg L$^{-1}$ and those for $NO_4^+$-N, DTN and DOC from 0.04 to 0.06, 0.1 and from 0.5 to 1.0 mg L$^{-1}$, respectively. If the concentration was less than DL, a value half of DL was used in further calculations. The percentages of samples below DL were 55%, 43% and 1% for the $NO_3^-$, $NH_4^+$ and DTN, respectively. DOC concentrations were always above the detection limit.

Monthly and annual deposition and fluxes in the soil for plots were calculated from mean concentrations (weighted by the contribution of individual collectors), the volume of water collected and the total collection area of all collectors. The differences between elemental concentrations at the uncut and cut plots were tested using the MULTTEST procedure with bootstrap *p*-value adjustment [SAS for Windows 6.12 (SAS®, 1997)]. The MULTTEST procedure does not assume normal distribution and takes into account the problem of multiple comparisons (Westfall and Young, 1993). A critical probability level of 0.05 was used to indicate significant differences. The statistical testing was done on annual and seasonal (spring, summer, autumn) values before and after clear-cutting. Monthly, logarithm transformed concentration values were used as the data set. The seasons before and after clear-cutting were analyzed separately and if no differences were found between the plots before clear-cutting, differences thereafter were considered to be the result of clear-cutting. The spring season included the months April and May starting from snow melt (first sampling in March in 1994), summer June through August, and autumn September through October (November in 1994 and 1999 and December in 1996). Correlations between concentrations were calculated from monthly plot data.

## Results

### *Water*

The mean annual bulk precipitation before clear-cutting (1993–1996) was 503 (±51) mm and after cutting (1997–1999) 480 (± 72) mm. Mean annual total throughfall was 92 (±8)% of bulk precipitation. Snowfall accounted for 39% of the annual bulk precipitation during the study period and mainly fell from November to April.

Before clear-cutting 84 (±12), 48 (±7) and 4 (±2)% of the annual total throughfall, on the average, passed through the O-, E- and B-horizons at the cut plot 1. The corresponding values at the cut plot 2 were 57 (±23), 20 (±4) and 5 (±0)%. After cutting, the proportion of precipitation passing through the understorey vegetation and the O-horizon remained similar, except in 1999 at the cut plot 1 when it decreased to 67%. Percolation below the E-horizon as a percentage of precipitation was not changed after cutting, but percolation below the B-horizon increased at both plots, being 7% at the plot 1 in 1999 and 9 and 6% at the plot 2 in 1997 and 1998.

### *DOC concentrations and fluxes*

The mean annual deposition of DOC in bulk precipitation was 12.1 (±1.2) kg $\text{ha}^{-1}$ and fivefold in total throughfall (stemflow accounted for 7.1% of the deposition) for the period 1993–1996 (Figure 1). After clear-cutting, the annual deposition of DOC to the forest floor decreased considerably averaging 8.4 (±1.0) kg $\text{ha}^{-1}$, since inputs were bulk deposition and not throughfall.

DOC concentrations in the soil percolate from below the O-horizon tended to be higher after clear-cutting at both cut plots, with seasonal increases being significant (Figure 2). Seasonal DOC concentrations in the percolate from below the E- and B-horizons were significantly higher only once. The annual leaching of DOC from the O-horizon increased twofold at the cut plot 1 and fivefold at the cut plot 2; the increase being the highest in the second year after cutting (Figure 1). Before cutting, the retention of DOC in the mineral soil was most efficient by the E-horizon at both plots, with outputs being 40 (±8) and 30 (±9)% of inputs to the horizon at the plots 1 and 2. After cutting, the efficiency of the E-horizon remained at least as high as that of the B-horizon at the plot 1, but increased further at the plot 2. The retention of DOC by the E- and B-horizons together was high with only slightly increased fluxes below the B-horizon at the cut plot 2 in 1997 and 1998. Before clear-cutting, the mineral soil at the cut plot 1 retained 155 (±33) and that at the cut plot 2, 101 (±30) kg $\text{ha}^{-1}$ of the percolating DOC inputs. After cutting, the retention was 198 (±52) and 132 (±29) kg $\text{ha}^{-1}$ $\text{a}^{-1}$ at plots 1 and 2, respectively. As a result of clear-cutting, the mineral soil pool of C increased by 129 and 96 kg $\text{ha}^{-1}$ over the three-year period after cutting at the plots 1 and 2, respectively.

### *Nitrogen concentrations and fluxes*

The annual total throughfall input of DTN before clear-cutting averaged 2.6 (±0.2) kg $\text{ha}^{-1}$ and deposition input to the forest floor did not differ much after cutting (2.5±0.5 kg $\text{ha}^{-1}$). However, the proportion of different forms of N did change. The annual deposition of DON decreased by more than 55% at both plots while that of $NH_4^+$-N increased, on an average, by 67% and that of $NO_3^-$-N by 48% (Figure 1).

Concurrent with the effect on DOC concentrations, clear-cutting resulted in increased DON concentrations in the percolate from below the O-horizon (Figure 2). The concentrations of DON and DOC were highly correlated ($r$=0.96 before clear-cutting, and $r$=0.92 after cutting). There were also some occasional elevated monthly DON concentrations in the percolates from below the E- and B-horizons, but these were not significant.

The increase in DON leaching from the O-horizon after clear-cutting was twofold at the plot 1 and fourfold at 2 (Figure 1). Before cutting, the annual retention of DON by the E-horizon averaged 49 (±8)% of the inputs at the plot 1 and 59 (±13)% at the plot 2. The degree of DON retention by the E-horizon did not change after cutting at the plot 1 but increased at the plot 2. Fluxes of DON from below the B-horizon were small (<0.2 kg $\text{ha}^{-1}$ $\text{a}^{-1}$) and changed little after cutting. The C/N ratio in the soil percolates below the O-, E- and B-horizons were 54, 46 and 39, respectively. After clear-cutting only the C/N ratio of the B-horizon percolate changed, decreasing to 20.

Ammonium concentrations were higher in the percolate from below the E-horizon than those from below the O- or B-horizons, both before and after clear-cutting (Figure 3). Clear-cutting increased $NH_4^+$ concentrations in the percolates from below both horizons, although the peak in the increase occurred later at the plot 2 than at the plot 1. Some elevated concentrations were also observed in the percolate from below the B-horizon. Clear-cutting also increased $NO_3^-$ concentrations in the O- and E-horizon percolates and highly elevated concentrations were occasionally observed also in the percolate from below the B-horizon (Figure 3).

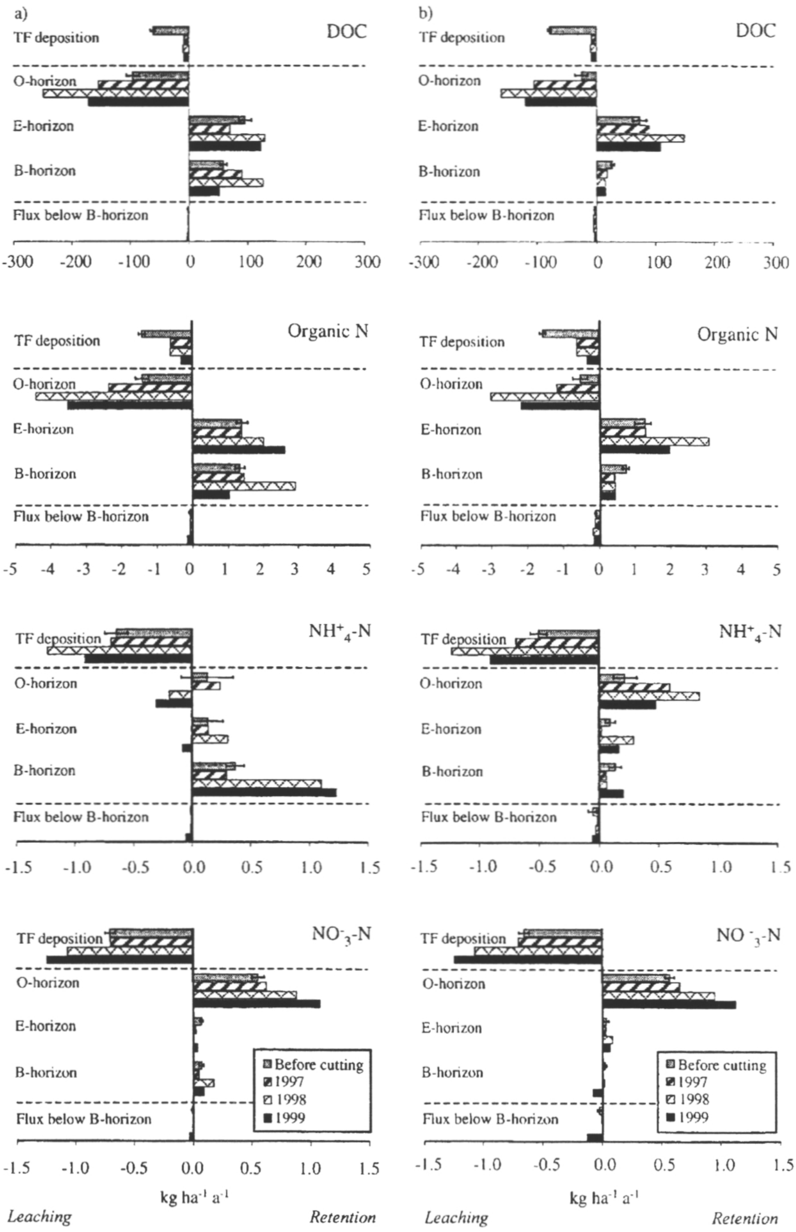

*Figure 1.* Mean annual total throughfall deposition (TF), flux below the B-horizon and the leaching from or retention in the different soil horizons of DOC and different forms of N before (1993–1996) and after (1997–1999) clear-cutting at the cut plots 1 (a) and 2 (b). The standard errors of the mean are indicated, *n*=4. The O-horizon includes understorey vegetation, and, after clear-cutting, logging residues.

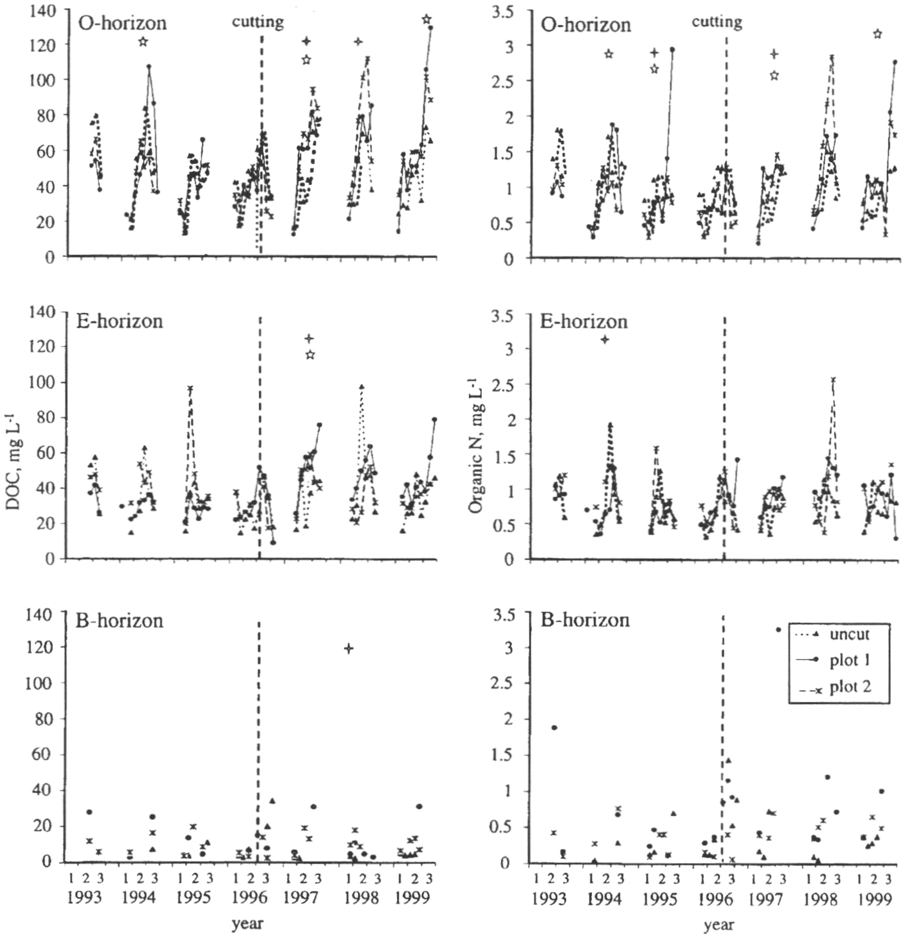

*Figure 2.* The mean monthly DOC and DON concentrations (mg $L^{-1}$) in the soil percolate from below the different soil horizons in 1993–1999 at the uncut plot and cut plots 1 and 2. Clear cutting was done in September 1996. 1=April–May, 2=June–August, 3=September–October. Significant differences ($p<0.01$) between the uncut and cut plot 1 are indicated with ✦ and those between the uncut and plot 2 with ☆.

Before clear-cutting, the between year variation in $NH_4^+$ fluxes were relatively high from the O-horizon, especially at the plot 1 (Figure 1). In the first year after clear-cutting, there was retention of $NH_4^+$ by the O-horizon at both cut plots. This retention persisted at the plot 2 during the second and third year but there was leaching at the plot 1. There was also retention of $NH_4^+$ by the E- and B-horizons during the first year after clear-cutting, but in the third year the E-horizon at the plot 1 showed $NH_4^+$ leaching. There was also an increased flux of $NH_4^+$-N below the B-horizon at the plot 1, but the amount was small and only increased from an average of 0.01 (±0.0 1) kg $ha^{-1}$ $a^{-1}$ before clear-cutting to 0.05 kg $ha^{-1}$ after clear-cutting in 1999.

The annual deposition input of $NO_3^-$ to the forest floor was mainly retained (>80%) by the O-horizon before and after clear-cutting (Figure 1). The E- and B-horizons at both cut plots also retained inputs before clear-cutting resulting in a low flux of $NO_3^-$-N from below the B-horizon (<0.03 kg $ha^{-1}$ $a^{-1}$). After clear-cutting, there was a little increase in $NO_3^-$ leaching and the flux from below the B-horizon was higher in the third year especially at the plot 2.

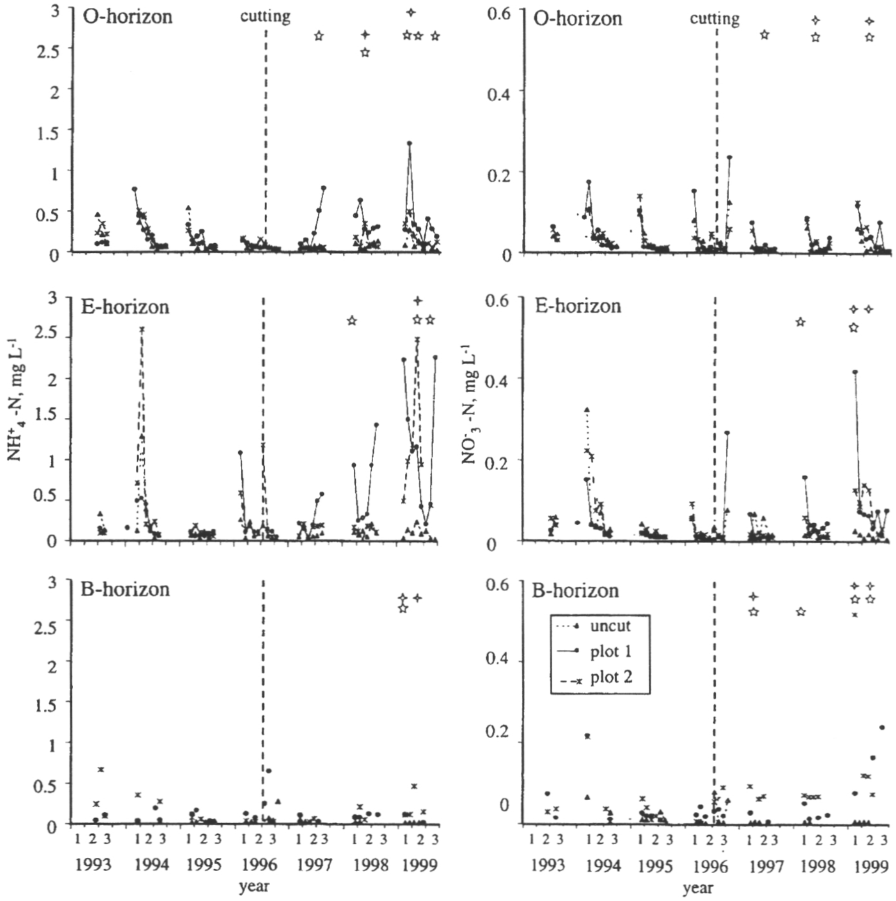

*Figure 3.* The mean monthly $NH_4^+$-N and $NO_3^-$-N concentrations (mg $L^{-1}$) of the soil percolates from below the different soil horizons in 1993–1999 at the uncut plot and cut plots 1 and 2. Clear-cutting was done in September 1996. For abbreviations and symbols see Figure 2.

Before clear-cutting, the mineral soil retained 3.4 (plot 1) and 2.2 (plot 2) kg N $ha^{-1}$ $a^{-1}$ (97 and 93% of DTN input, respectively). In the second year after cutting, the level of DTN retention enlarged a little, resulting in an addition of 4.7 or 2.1 kg $ha^{-1}$ to the mineral soil pool over the three-year study period at plots 1 or 2, respectively. The addition was mainly in the form of DON (>65%).

## Discussion

Our results show that throughfall was substantially enriched in DOC as a result of canopy leaching (Tukey, 1970). On average, 62 and 79 kg $ha^{-1}$ $a^{-1}$ of DOC was leached from the canopy at the plots 1 and 2, respectively. These amounts are similar to those reported by Curie et al. (1996) and Michalzik and Matzner (1999) for coniferous forests in the temperate zone. The greater tree biomass at the plot 2 (Table 1) might explain the higher amounts of DOC in the throughfall at this plot. Nevertheless, the throughfall input of C to the forest floor was small compared to that associated with canopy litterfall input, which was about 15-times greater (958 kg C $ha^{-1}$ $a^{-1}$) (Finér et al., 2002).

As shown in our previous study (Piirainen et al., 1998), N deposition was low (<3 kg $ha^{-1}$ $a^{-1}$) compared to that in the temperate zone of Europe (e.g.

Hjellbrekke, 2000). The throughfall input of DTN to the forest floor before clear-cutting was mostly (56%) organic (Piirainen et al., 1998) and inorganic (79%) afterwards. Thus, the removal of trees at cutting significantly increased the throughfall deposition of DIN, whereas the total deposition of N was not affected. Prior to clear-cutting the combined throughfall and canopy litterfall input of N to the forest floor averaged 15 kg $ha^{-1}$ $a^{-1}$ (Finér et al., 2002). Clear-cutting resulted in the abrupt cessation of regular throughfall and canopy litterfall inputs of both C and N, but there was a large and single addition in the form of logging residues (Table 1).

Before clear-cutting, leaching from the tree crown was an important source of DOC to the O-horizon. The increase in DOC concentrations after cutting in the percolate could result from the decomposition and leaching of soluble organic matter from both the organic layer and logging residues (Qualls et al., 2000). The increased DOC flux (plot average 504 kg $ha^{-1}$ over three-year period) from the O-horizon was small compared to the C pools associated with logging residues and the organic horizon (Tables 1 and 2). However, the leaching of DOC was at a higher level throughout the three-year period and thus likely to continue for several years as a result of the decomposition of logging residues. But the amounts will be small compared to those released to the atmosphere as $CO_2$ by respiration of microbes (Mattson et al., 1987).

The retention of DOC by the mineral soil was effective both before and after clear-cutting. This retention could take place through the accumulation of humic substances in the soil or increased respiration losses to the atmosphere. However, as the C/N ratio of the percolate from below the O- and B-horizons did not change after clear-cutting, significant mineralization of the soluble organic complexes is not indicated. According to Qualls and Haines (1992), the biodegradability of DOC in mineral soil percolate is usually relatively low due to the abundance of hydrophobic and hydrophilic acids and no changes in the composition of DOC after clear-cutting have been observed in earlier studies (Smolander et al., 2001). The accumulation of humic substances in the mineral soil is known to be strongly associated with Fe and Al sesquioxides (Jardine et al., 1989; McDowell and Wood, 1984), especially amorphous Al hydroxides (Kaiser and Zech, 1998). Amorphous Fe and Al hydroxide concentrations were relatively high in the upper part of the E-horizon (Table 2), as is typical for Podzols, and the greatest retention would therefore have been expected to occur there. However, we found the retention of DOC to be evident also in the E-horizon. The reasons could be the decomposition to $CO_2$ or biological immobilization, because chemical retention or precipitation of C to the E-horizon should not be important according to podzolization theories (e.g. Lundström et al., 2000).

Nitrogen leaching from the O-horizon prior to clear-cutting was dominated by DON (Piirainen et al., 1998), as has also been shown in temperate and boreal forests (Curie et al., 1996; Fahey et al., 1985; Qualls et al., 1991; Smolander et al, 2001). The flux of DTN (>70% organic) from below the 0-horizon increased after cutting but the loss over the three-year period was small (plot average 12 kg $ha^{-1}$) compared to pool of N in the logging residues and O-horizon (Tables 1 and 2). The fluxes of both inorganic and organic forms of N in the percolate from below the O-horizon were increased after clear-cutting, with no changes in the proportions of these fractions. Other studies have shown a marked increase in DIN leaching from below the O-horizon. In coniferous forests in the boreal zone, this increase has been due to $NH_4^+$ (Rosén and Lundmark-Thelin, 1987), but in richer sites in the temperate zone the increase has been due to $NO_3^-$ (Sollins and McCorison, 1981). In our forest, the proportion of $NH_4^+$-N out of the flux of DIN from below the O-horizon was greater than that of $NO_3^-$-N and it increased further after cutting. The $NH_4^+$ ions were efficiently retained by the mineral soil, probably to cation exchange sites and some were likely also nitrified to $NO_3^-$.

Nitrate concentrations in soil percolates were low prior to clear-cutting (Piirainen et al., 1998), substantially lower than reported in other studies carried out in coniferous forests (e.g. Smolander et al., 2000; Vitousek and Matson, 1984), and remained low also after clear-cutting. The $NO_3^-$-N deposition in this remote site was very low ($<1.5$ $ha^{-1}$ $a^{-1}$; Piirainen et al., 1998) and it would probably be strongly immobilized in such a N-limited ecosystem, where nitrification is also probably insignificant, even after clear-cutting (Vitousek et al., 1982). However, we observed some small increases in $NO_3^-$ concentrations in the percolate from below different soil horizons after clear-cutting (Figure 3). This could be due to reduced root uptake or increased nitrification. Nitrification is known to be stimulated by an increase in $NH_4^+$ availability, pH (e.g. Martikainen, 1984) and temperature and soil moisture after forest cutting (Matson and Vitousek, 1981) or reduction in allelopathic inhibitors (Paavolainen et al., 1998). Increased $NO_3^-$ concentrations in groundwater or streams have been observed in clear-cutting studies done in N-limited forests (Ahtiainen and Huttunen, 1999; Kubin, 1998). However, the output have remained small compared to those from areas with high N deposition loads and N saturation (cf. Adamson et al., 1987; Wiklander et al., 1991), or a high potential for nitrification (cf. Stevens and Hornung, 1988).

Clear-cutting increased at least temporarily the mineral soil C pool by 0.5% and that of N by 0.3% in the three-year study period. The low C and N leaching losses below the B-horizon indicated that the release of nutrients from the mineral soil pool was not influenced by clear-cutting. The logging residues will most probably be a long-term source of N for the regeneration of new vegetation at our site.

## Conclusions

The leaching of DON and DOC from below the O-horizon increased more than that of DIN, which was mostly immobilized in the logging residues and O-horizon, indicating increased microbial activity soon after clear-cutting. However, C was retained in the mineral soil and the flux of DOC below the B-horizon did not increase markedly. The flux of DIN below the B-horizon after clear-cutting remained small <0.4 kg $ha^{-1}$ $a^{-1}$ although occasional elevated flux of $NO_3^-$ indicated increased nitrification. Over the three-year period following clear-cutting a plot average of 112 kg $ha^{-1}$ C and 3.3 kg $ha^{-1}$ N were added to the mineral soil pools. These amounts are very small compared to the pools in the soil.

## Acknowledgements

The authors wish to thank the laboratory staff of the Finnish Forest Research Institute Joensuu Research Centre and Central laboratory, Vantaa for analysing the water samples, Ms. Leena Karvinen for preparing the Figures and Tables, Mr. Jaakko Heinonen for statistical advice and qualified technicians for the field work. We also wish to acknowledge the Finnish Forest and Park Service for making the site available for our research project and Ministry of Agriculture and Forestry for partly funding the project.

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*Section editor: R.F. Hüttl*

IV

# PAPER IV

Piirainen, S., Finér, L., Mannerkoski, H. & Starr, M. Effects of forest clear-cutting on the sulphur, phosphorus and base cation fluxes through podzolic soil horizons (submitted to Biogeochemistry).

# Effects of forest clear-cutting on the sulphur, phosphorus and base cation fluxes through podzolic soil horizons

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Key words: calcium, harvesting, leaching, lysimeter, magnesium, potassium
Running head: nutrients leaching after forest clear-cutting
Article type: general paper

# Abstract

Clear-cutting considerably alters the flow of nutrients through the forest ecosystem. These changes are reflected in soil solution concentrations and fluxes. The effects of clear-cutting (stems only) on the fluxes of water soluble phosphorus (P), sulphur (S) and base cations (Ca, Mg and K) through a podzolic soil were studied in a Norway spruce dominated mixed boreal forest in eastern Finland. Nitrification was negligible and deposition, particularly of S, low. Fluxes were monitored for four years before and for three years after cutting. Bulk deposition, total throughfall (throughfall+stemflow) and soil percolate from below the organic (O), eluvial (E) and illuvial (B) horizons were collected. Deposition loads to the forest floor were less after clear-cutting, averaging 1.7 kg S, 0.84 kg Ca, 0.14 kg Mg, 0.64 kg K and 0.10 kg P ha$^{-1}$ a$^{-1}$. Fluxes of total S and sulphate ($SO_4^{2-}$) from below the O-horizon were also lower after clear-cutting, total S averaging 2.0 kg ha$^{-1}$ a$^{-1}$. Although, the proportion of $SO_4^{2-}$ retained by the soil decreased, the fluxes of total S from below the B-horizon remained similar. The flux of P (mainly inorganic) from below the O-horizon increased threefold (6.9 kg ha$^{-1}$) in 1997-1999 compared to the period before cutting. The sum of base cation fluxes from below the O-horizon increased twofold. The fluxes of phosphate ($PO_4^{3-}$) from below the O- and E-horizons were correlated with those of $K^+$ (r=0.37), but not those of $Ca^{2+}$ and $Mg^{2+}$. Increased fluxes of P and base cations to the mineral soil generated only slightly increased fluxes below the B-horizon. The retention of base cations and P in the mineral soil indicates there would be little increased leaching to ground and surface waters after clear-cutting.

# Introduction

Clear-cutting eliminates the interaction between canopy and deposition, diminishes uptake from the soil by vegetation, and replaces the continuous litterfall to the soil by a sudden and large input of dead organic matter in the form of logging residues. These changes all increase the potential for nutrient leaching from the soil. The removal of tree canopy decreases the deposition of sulphur (S) and base cations (Ca, Mg and K) to the forest floor; the former because there is no interception of suspended dry deposition (e.g. Mayer & Ulrich 1977) and the latter because there is no foliar leaching (e.g. Tukey Jr 1970). The return of phosphorus (P) to the forest floor is less affected because the dry deposition and foliar leaching of P are small (e.g. Parker 1983). However, the most of the flux of P, Ca and Mg to the forest floor is via litterfall (e.g. Helmisaari 1995). The throughfall flux of S and K can be much higher than that in litterfall (e.g. Nihlgård 1972). The amount of S, P and base cations contained in logging residues i.e. branches, needles, stumps and roots is large compared to the annual litterfall fluxes of conifers (e.g. Nihlgård 1972). Increased decomposition after clear-cutting has been shown to result in both sulphate ($SO_4^{2-}$) leaching from logging residues (Cortina and Vallejo 1994) or S retention (Gosz et al. 1973). When the C/P ratio is low enough as not to limit mineralization, phosphate ($PO_4^{3-}$) is readily released from dead organic matter (Berg & Staaf 1980). Base cations, especially K, have also been released rapidly from logging residues (Robertson et al. 2000; Staaf & Olsson 1994; Stevens et al. 1995; Titus & Malcolm 1992). Solutes released from logging residues may be immobilized, at least temporarily, in the expanding microbial population or taken up by the developing ground vegetation. They may also be adsorbed by the soil through ion exchange or precipitation reactions, or they may leach to ground and surface waters.

Large leaching losses of base cations from clear-cut areas have been reported for temperate forests along with the formation of nitrate ($NO_3^-$) (Dahlgren & Driscoll 1994; Homann et al. 1994; Likens et al. 1969; Titus et al. 1997). However, studies carried out in the boreal zone with low N deposition and potential of nitrification are lacking. Since the leaching of cations must be accompanied by an equivalent amount of anions and $NO_3^-$ concentration in soil water from such sites is low, anions other than $NO_3^-$ must be involved. Carbonate/bicarbonate ($CO_3^{2-}$/$HCO^{3-}$) concentrations are low in acid forest soils (Bohn et al. 1985) and chloride, ($Cl^-$) being conservative and highly mobile, is unlikely to be markedly altered by clear-cutting. Adsorption of $SO_4^{3-}$ and $PO_4^{3-}$ ions by Al and Fe oxides can be expected to be efficient in podzolic soils (Gobran et al. 1998; Gustafsson & Jacks 1993; Singh et al. 1980) and for $PO_4^{3-}$, partly irreversible (Stevenson & Cole 1999). Phosphate can also form insoluble complexes with Ca and Mg (Johnson & Cole 1980). Incomplete mineralization of organic matter, which is typical for podzolic soils, produces soluble and, under low pH conditions, negatively charged organic ions. Organic anions may therefore be involved in cation leaching. However, the leaching of organic matter has not been shown to markedly increase by clear-cutting (see review by Hope et al., 1994).

Besides the loss of nutrients from the soil and associated decline in site productivity, it is important to know whether clear-cutting increases inputs, especially those of P, to surface waters. Increased fluxes of P to groundwater (Kubin 1995) and surface waters (Ahtiainen & Huttunen 1999) have been reported to occur in the boreal zone after cuttings.

In this paper, we present results concerning the effects of clear-cutting (stems only) on the fluxes of water soluble P, S and base cations through the surface soil horizons of a Norway spruce dominated mixed boreal forest. We hypothesise that even after such a major disturbance of a ecosystem, the mineral soil will effectively retain any increased nutrient fluxes, resulting in small leaching losses. The study is a part of an ongoing research project evaluating the effects of clear-cutting and site preparation on nutrient fluxes through the soil and on stream water quality and yield (Finér et al. 1997).

# Material and methods

## Study area

The data presented were collected from three adjacent 50 m x 50 m sample plots located on a south easterly facing slope in the Kangasvaara catchment in eastern Finland (63° 51'N, 28° 58'E, 220 m a.s.l.) (Finér et al. 1997). The boreal forest was an old-growth mixed coniferous stand dominated by Norway spruce (*Picea abies* Karsten). Scots pine (*Pinus sylvestris L.*); white and silver birch (*Betula pubescens* Ehrh. and *Betula pendula* Roth) and European aspen (*Populus tremula* L.) were also present (Table 1). Clear-cutting was done at two of the plots in September 1996 and one plot left uncut to serve as a control. Only stems (with bark) were removed, the logging residues (branches, leaves and tree tops < 8 cm diameter) being left, evenly distributed over the surface.

***Table 1.* Tree stand characteristics of the uncut plot and clear-cut plots 1 and 2 and annual litterfall elemental fluxes before clear-cutting in 1996, and logging residues elemental pools.**

| | uncut plot | plot 1 | plot 2 |
|---|---|---|---|
| Stem volume with bark, $m^3$ $ha^{-1}$ | 260 | 230 | 290 |
| Spruce, % | 78 | 50 | 33 |
| Pine, % | 11 | 36 | 51 |
| Deciduous, % | 11 | 14 | 16 |
| P in above ground litterfall, kg $ha^{-1}$ | 1.6 | 1.6 | 1.9 |
| S in above ground litterfall, kg $ha^{-1}$ | 1.1 | 0.9 | 1.1 |
| Ca in above ground litterfall, kg $ha^{-1}$ | 16.0 | 10.6 | 14.4 |
| Mg in above ground litterfall, kg $ha^{-1}$ | 2.0 | 1.6 | 2.2 |
| K in above ground litterfall, kg $ha^{-1}$ | 4.7 | 2.8 | 3.8 |
| P in above ground logging residues, kg $ha^{-1}$ | . | 20 | 21 |
| S in above ground logging residues, kg $ha^{-1}$ | . | 15 | 15 |
| Ca in above ground logging residues, kg $ha^{-1}$ | . | 144 | 138 |
| Mg in above ground logging residues, kg $ha^{-1}$ | . | 17 | 18 |
| K in above ground logging residues, kg $ha^{-1}$ | . | 69 | 75 |

According to the Finnish site type classification (Cajander 1949; Mikola1982), the site was classified as a *Vaccinium-Myrtillus* type. Accordingly, the field layer was dominated by dwarf shrubs (*Vaccinium vitis-idaea* L. and *V. myrtillus* L.) and the bottom layer by feather mosses (*Pleurozium schreberi Brid.* and *Hylocomium splendens (Hedw.) B. S. & G.)*. The forest floor consisted of a litter and mor humus layer with an average thickness of 3 cm. The soil, derived from sandy till material, was a weakly developed iron-podzol with a clay content of <2 %. The stone content of the till, determined by the rod penetration method of Viro (Tamminen & Starr 1994), was 28 %. Further characteristics of the soil are presented in Table 2.

The mean annual air temperature averaged +0.7 °C and the annual temperature sum (sum of daily mean temperature exceeding +5 °C) 903 °C for the period 1993-1999. A more detailed description of the study area has been presented elsewhere (Finér et al. 1997).

***Table 2*. Some chemical properties and elemental pools in soil horizons at the uncut plot and clear-cut plots 1 and 2.**

| | O-horizon | | | E-horizon | | | B-horizon | | |
|---|---|---|---|---|---|---|---|---|---|
| | Uncut | plot 1 | plot 2 | uncut | plot 1 | plot 2 | uncut | plot 1 | plot 2 |
| Thickness of horizon, cm | 3.0 | 3.5 | 2.8 | 7.1 | 6.0 | 7.2 | 12.8 | 13.7 | 15.3 |
| $pH_{H2O}$ | 4.22 | 3.84 | 3.93 | 4.22 | 4.24 | 4.22 | 4.95 | 4.98 | 5.20 |
| C, %[a] | 38.8 | 49.8 | 47.8 | 1.5 | 1.6 | 1.1 | 2.2 | 2.2 | 1.4 |
| P, %[b] | 0.06 | 0.07 | 0.08 | 0.01 | 0.02 | 0.01 | 0.03 | 0.06 | 0.08 |
| P, kg $ha^{-1}$ [b] | 30.7 | 38.0 | 32.4 | 43.3 | 75.5 | 76.3 | 229.6 | 526.3 | 942.4 |
| S, kg $ha^{-1}$ [b] | 50.0 | 58.8 | 45.5 | 31.9 | 60.7 | 23.9 | 302.5 | 379.5 | 516.5 |
| Ca, kg $ha^{-1}$ [b] | 164.1 | 148.5 | 132.3 | 139.8 | 153.0 | 337.0 | 494.0 | 474.6 | 2218.3 |
| Mg, kg $ha^{-1}$ [b] | 42.6 | 26.7 | 27.1 | 174.7 | 409.7 | 159.9 | 1716.6 | 1639.2 | 3026.4 |
| K, kg $ha^{-1}$ [b] | 61.5 | 63.7 | 58.7 | 63.6 | 77.0 | 88.9 | 244.8 | 283.8 | 625.3 |
| CEC, $kmol_c$ $ha^{-1}$ [c] | 12.8 | 14.5 | 11.0 | 17.8 | 16.6 | 19.1 | 15.1 | 10.9 | 8.8 |
| Base saturation, % | 69 | 61 | 69 | 13 | 9 | 9 | 17 | 15 | 32 |

[a] determined with Leco CHN analyzer

[b] O-horizon: nitric acid + $H_2O_2$ digest; E- and B-horizons: nitric + hydrochloric acid digest

[c] effective cation exchange capacity (0.1 M $BaCl_2$ extraction).

## Deposition and soil water sampling

Bulk precipitation in a nearby clearing and throughfall at the sample plots were collected during the four-year period prior to clear-cutting (1993-1996) and for three years afterwards (1997-1999) using permanently open bulk precipitation collectors. For bulk precipitation in the open, five plastic collectors (each 131 $cm^2$) were used during the snow-free period and three bigger ones (each 299 $cm^2$ during 1992-1993 and 1146 $cm^2$ thereafter) during the winters. The collectors were placed

0.6 m above ground level during the snow-free period and at 1.5 m in winter. Throughfall was collected at each sample plot using 16 collectors during the snow-free period and 8 collectors for snow in winter. The same type of collectors as for bulk precipitation in the open were used and placed systematically at equal distances along each side of the plots. After clear-cutting, throughfall was collected only at the uncut plot. Stemflow was collected during 1993-1995 at two of the plots. For the following years, stemflow fluxes were taken as the average of the measured years. At each plot, five white or silver birches, Scots pines and Norway spruces representing different breast height diameter classes (>6 cm) were fitted with a spiral type stemflow collector made of silicon tubing.

The bulk precipitation and throughfall collectors were emptied once a week during the snow-free period and once a month during winter. On each sampling occasion the volume collected was recorded. Weekly or monthly bulk precipitation and throughfall samples were pooled by plot for chemical analyses. Stemflow was collected weekly and only during the snow-free period, and pooled by tree species for chemical analysis.

Soil water (percolate) was sampled at each sample plot using nine zero-tension lysimeters installed in the plots at each of three depths: below the organic (O), below the eluvial (E) (at 14, 11 and 11 cm depths from the soil surface at uncut plot and cut plots 1 and 2, respectively), and below the upper part of the illuvial (B) horizon (at 36, 37 and 32 cm depths from the soil surface, respectively). Particular care was taken during installation to avoid damage to the field layer vegetation and roots. The three lysimeters below the B-horizon in the cut plot 2 and two lysimeters below the O-horizon in the cut plot 1 did not collect water or only before July 1998, and were therefore excluded from subsequent calculations. The lysimeters below the O-horizon were of the Jordan type (Jordan 1968); a plexiglass gutter, closed at the ends with a collecting area of 420 cm$^2$. The lysimeters installed below the E- and B-horizons were made of a polythene plastic funnel, filled with quartz sand, with a collecting area of 299 cm$^2$ fitted to a 2 L sample collection bottle. The particle size of the quartz sand used varied from 0.8 to 1.2 mm. The main soil texture class in the E- and B-horizons was also sand (41 and 39 %, respectively), allowing free water flow from soil to lysimeter (Piirainen et al. 2002a). The lysimeters were emptied on the same day as the precipitation and throughfall collectors when not frozen (mostly from April to October) and each lysimeter sample was analysed separately. Care was taken during the clear-cutting operations not to disturb the lysimeter installations.

We use the term *total throughfall* to mean throughfall plus stemflow, *leaching* to mean output of percolate in excess of input for a soil horizon and *retention* when inputs are greater than outputs. *Base cations* refer to $Ca^{2+}$, $Mg^{2+}$ and $K^+$ ions. The water and element fluxes below the O-horizon were affected by forest floor, understorey vegetation and, after clear-cutting, also by the logging residues.

## Laboratory analyses and calculations

All the samples were kept cold and transported to laboratory the day after sampling. First, pH was measured (Radiometer PHM92 pH-meter) and thereafter the samples were filtered through a Schleiche & Schuell no. 589[1] filter; a Schleiche & Schuell no. GF 52 glass wool filter in the case of soil percolate samples. All the filtrates were then stored in a freezer (-18 °C) until further analyses. Sulphate and $PO_4^{3-}$ concentrations were determined by ion chromatography (Dionex series 100 or 300 or 500) and $Ca^{2+}$, $Mg^{2+}$ and $K^+$ concentrations by flame atomic absorption spectrophotometer (Perkin-Elmer 5000). Total S and P concentrations in a representative number of samples were measured with ICP emission spectrometer (ARL 3580 in 1993-1996 and TJA Iris Advantage thereafter). These data were used to derive equations to estimate total S and P concentrations using measured $SO_4^{2-}$ and $PO_4^{3-}$ concentrations (Table 3).

The detection limit (DL) for $SO_4^{2-}$-S was 0.07 mg $L^{-1}$ and those for $PO_4^{3-}$-P, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ 0.05, 0.10, 0.02 and 0.05 mg $L^{-1}$, respectively. For samples collected during 1993 - 1995, the DL of $Ca^{2+}$ was 0.30 mg $L^{-1}$ and that of $PO_4^{3-}$-P 0.02 mg $L^{-1}$ in 1996-1997. Detection limits of total S and total P were 0.07 and 0.15 mg $L^{-1}$ in 1993-1996 and 0.03 and 0.02 mg $L^{-1}$ thereafter. If the concentration was less than DL, a value of half DL was used in further calculations. Some 80 % of $Ca^{2+}$ bulk precipitation concentrations collected in 1993 - 1995 were less than DL and 13 % thereafter. The percentage of bulk precipitation samples with concentrations below DL was 76 % for $PO_4^{3-}$, 56 % for total P, 14 % for $Mg^{2+}$ and 6 % for $K^+$. For $PO_4^{3-}$ concentrations in soil percolate, 30 % of samples collected from below the O- and E-horizons were below DL and 80 % of samples from below the B-horizon.

Monthly and annual deposition fluxes were calculated from mean concentrations (volume weighted by contribution of individual collectors), the volume of water collected and the total collection area of all collectors. Significance differences in concentrations between uncut and cut plots were tested using the SAS MULTTEST procedure with bootstrap p-value adjustment [SAS for Windows 6.12 (SAS® 1997)]. This procedure does not assume normal distribution and takes into account the problem of multiple comparisons (Westfall & Young 1993). A critical probability level of 0.05 was used to indicate significant of differences. The statistical testing was done on annual and seasonal (spring, summer, autumn) values before and after clear-cutting. Monthly, logarithm transformed concentrations values were used as the data set. The seasons before and after clear-cutting were analyzed separately and if no differences were found between the plots before clear-cutting, differences thereafter were considered to be the result of clear-cutting. The spring season included the months April and May starting from snowmelt (first sampling in March in 1994), summer June through August, and autumn September through October (November in 1994 and 1999 and December in 1996). Correlations between solute concentrations were calculated from monthly plot data.

# Results

## Water fluxes

The results of the water fluxes have been earlier published by Piirainen et al. (2002b). Briefly, the annual bulk precipitation during the entire study period averaged 493 mm. Total throughfall averaged 92 % of this value (only uncut plot data for 1997-1999). Precipitation fell mainly as snow from November to April, and accounted for 39 % of the annual bulk precipitation. Of the annual amount of throughfall at the cut plot 1 before clear-cutting 84, 48 and 4 % passed through the O-, E- and B-horizons, respectively. The corresponding values at cut plot 2 were 57, 20 and 5 %. After cutting, the proportion of the annual incident precipitation passing through the understorey and ground vegetation and O-horizon remained similar, except in 1999 at cut plot 1 when it decreased to 67 %. The relative amount of percolation below the E-horizon (percentage of incident precipitation) also did not change after cutting, but that below the B-horizon increased, being 7 % at cut plot 1 in 1999 and 9 and 6 % at cut plot 2 in 1997 and 1998.

## pH

The pH of the precipitation reaching the forest floor increased after clear-cutting. The mean annual bulk precipitation pH (4.72 and 4.99 for the periods 1993-1996 and 1997-1999, respectively) was a little higher than that of the total throughfall (4.49 and 4.54, respectively). Clear-cutting, however, did not change the pH of the soil percolate from below the O-horizon, but did significantly increase that of the soil percolates from below the E-and B-horizons in 1998 and in the E-horizon in 1999 at cut plot 2 compared to the uncut plot (Figure 1). There were also occasional elevated monthly pH values in the soil percolates collected at cut plot 1 after cutting, but these changes were not significant.

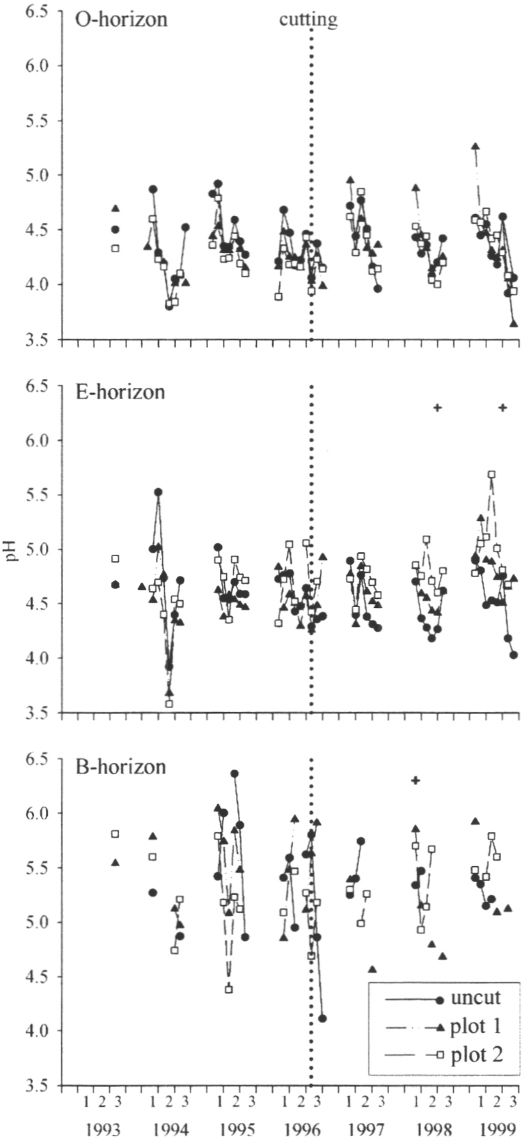

*Figure 1.* **Mean monthly pH in the soil percolate from below each horizon at the uncut plot and the cut plots 1 and 2 in 1993-1999. Clear-cutting was done in September 1996. 1=April-May, 2=June-August, 3=September-October. Significant differences ($p<0.05$) between the uncut and cut plot 1 are indicated with $^+$ and those between the uncut and plot 2 with *.**

## Sulphur and phosphorus concentrations and fluxes

Sulphate-S constituted about 90 % of total S deposition to the forest floor before clear-cutting and about 85 % thereafter. The deposition of $SO_4^{2-}$ to the forest floor decreased after clear-cutting by more than 50 % to an average of 1.72 kg S ha$^{-1}$ a$^{-1}$ in 1997-1999 (Figure 2). The deposition of organic S (total S - $SO_4^{2-}$-S) decreased somewhat less, being 0.46 kg S ha$^{-1}$ a$^{-1}$ in throughfall during 1993-1996 and 0.26 kg S ha$^{-1}$ a$^{-1}$ in 1997-1999 (bulk precipitation).

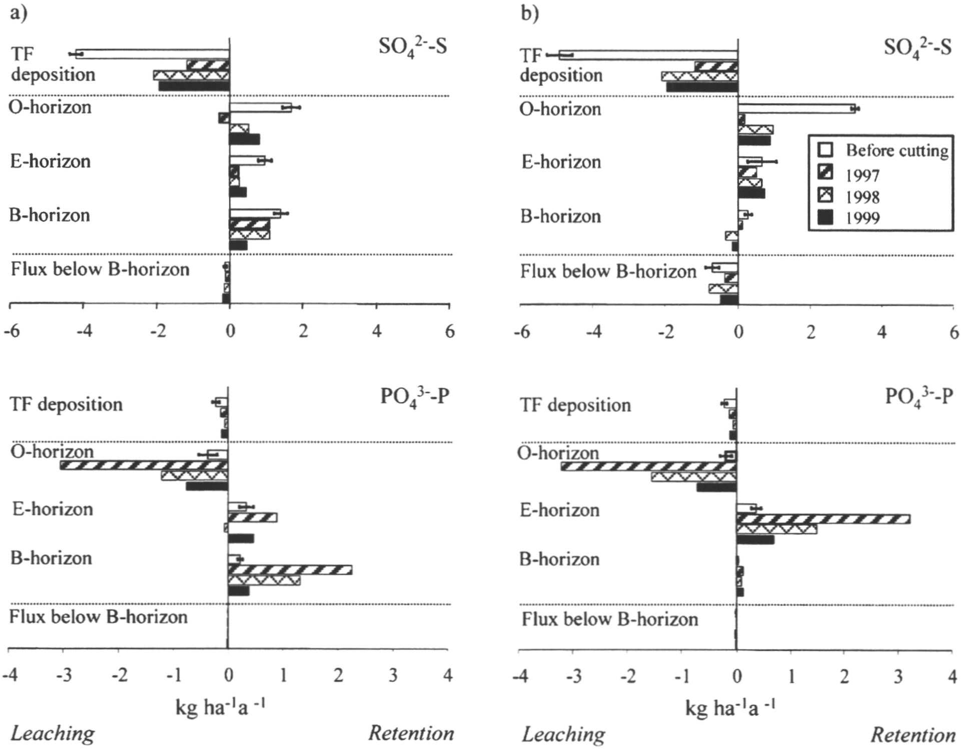

***Figure 2*. Mean annual total throughfall deposition (TF), flux below the B-horizon and the leaching from or retention in the different soil horizons of $SO_4^{2-}$ and $PO_4^{3-}$ before (1993-1996) and after (1997-1999) clear-cutting at the cut plots 1 (a) and 2 (b). The error bars are the standard error of the mean, n=4.**

Clear-cutting did not have clear effect on soil water sulphate concentrations but there were a few clear-cut plot seasonal values that were significantly different from uncut plot values (Figure 3). Before clear-cutting, 40 (±9) % of the annual throughfall input of $SO_4^{2-}$ at cut plot 1 and 67 (±8) % at cut plot 2 was retained by the O-horizon (Figure 2). After cutting, in 1997, the retention was only 16 % at cut plot 2 and leaching occurred at cut plot 1. The degree of $SO_4^{2-}$ retention by the O-

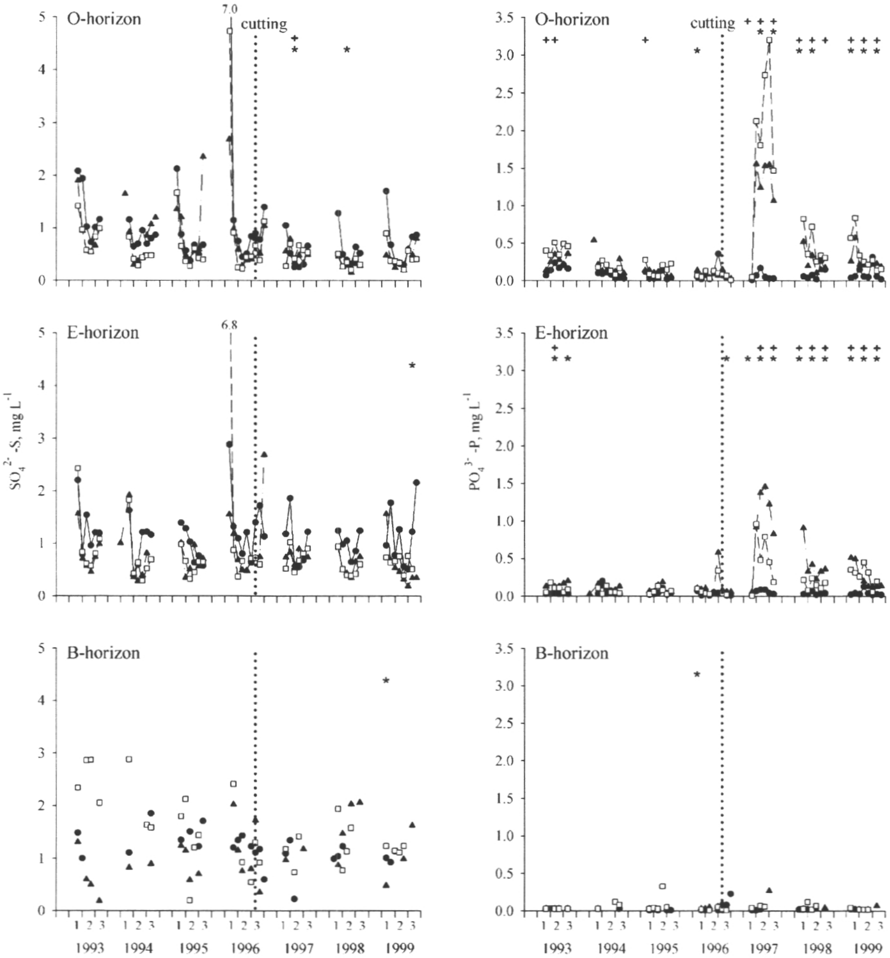

***Figure 3*. Mean monthly SO$_4^{2-}$ and PO$_4^{3-}$ concentrations (mg L$^{-1}$) in the soil percolate from below each horizon at the uncut plot and the cut plots 1 and 2 in 1993-1999. Clear-cutting was done in September 1996. For abbreviations and symbols see Figure 1.**

horizon partly recovered in subsequent years, being 42 and 46 % of the deposition input at cut plots 1 and 2, respectively, in 1999. The annual flux of SO$_4^{2-}$ below the O-horizon decreased by more than 30 % after cutting, being, on an average, 1.38 and 1.04 kg S ha$^{-1}$ a$^{-1}$ at cut plots 1 and 2, respectively.

Before clear-cutting, the retention of SO$_4^{2-}$ by the E-horizon averaged 38 (±13) % of the percolate input (that leaving the O-horizon) at cut plot 1 and 31 (±36) % at cut plot 2 (Figure 2). After cutting, retention decreased to 17 % but increased to its initial level in 1999 at cut plot 1. At cut plot 2, no such reduction was observed and in 1999 the retention was as high as 70 %. Before cutting, the retention of SO$_4^{2-}$ by the B-horizon was, on average, 90 (±7) % and 32 (±24) % of the input (that leaving the E-horizon) at cut plots 1 and 2, respectively (Figure 2).

After cutting, retention by the B-horizon decreased year by year at cut plot 1, being 71 % in 1999, and at cut plot 2 there was no retention by 1998 but rather leaching losses. Nevertheless, the overall flux of $SO_4^{2-}$ from below the B-horizon at both cut plots did not change after clear-cutting. Before cutting, the annual $SO_4^{2-}$ flux from below the B-horizon averaged 0.13 (±0.08) and 0.70 (±0.37) kg S $ha^{-1}$ $a^{-1}$ at cut plots 1 and 2, respectively. After cutting, corresponding values were 0.16 (±0.04) and 0.54 (±0.22) kg S $ha^{-1}$ $a^{-1}$. The apparent contradiction between decreasing retention while maintaining the same level of flux from below the B-horizon is due to an internal input of $SO_4^{2-}$ to the B-horizon percolate. This explains why the $SO_4^{2-}$ flux below the B-horizon relative to the incident deposition input increased from 3 (±2) to 9 (±1) % at cut plot 1 and from 15(±9) % to 31 (±7) % at cut plot 2 after cutting.

Organic S accounted for 31 % of the total S flux from below the O-horizon (Table 3) before clear-cutting. This corresponded to an average flux of organic S from below the O-horizon of 0.92 kg $ha^{-1}$ $a^{-1}$. Retention of organic S by the E-horizon was negligible, since the estimated average flux below the horizon was 0.95 kg $ha^{-1}$ $a^{-1}$. The flux of organic S from below the B-horizon was higher, averaging 1.15 kg $ha^{-1}$ $a^{-1}$. After clear-cutting, the proportion of total S leaching as organic S from below the O-horizon increased slightly to 39 %, but the actual flux (0.79 kg $ha^{-1}$ $a^{-1}$) decreased. Both the actual (1.72 kg $ha^{-1}$ $a^{-1}$) and relative fluxes of organic S from below the E-horizon increased. In contrast, the actual (0.39 kg $ha^{-1}$ $a^{-1}$) and relative organic S fluxes below the B-horizon decreased. However, the relationship between the total and inorganic S for both the E- and B-horizons were not strong (Table 3), and the organic S flux estimates for these horizons should be viewed with caution.

After clear-cutting, the annual deposition of $PO_4^{3-}$ to the forest floor slightly decreased, averaging 0.10 (±0.04) kg P $ha^{-1}$ $a^{-1}$ (Figure 2). Cutting increased $PO_4^{3-}$ concentrations in the percolate from below the O- and E-horizons, although some elevated concentrations on the cut plots were also observed in the period before cutting (Figure 3). Phosphate concentrations in the percolate from below the B-horizon remained low after cutting.

Due to the changes in $PO_4^{3-}$ concentrations, the leaching from the O-horizon increased after clear-cutting (Figure 2). In 1997, the increase was eightfold at cut plot 1 and 17-fold at cut plot 2. In the following years, $PO_4^{3-}$ leaching decreased but remained twofold and threefold compared to level before cutting at cut plots 1 and 2, respectively, in 1999.

The E-horizon retained 54 (±25) and 86 (±7) % of the annual O-horizon input at plots 1 and 2, respectively (Figure 2). In 1997, after cutting, retention decreased to 29 % at cut plot 1 and increased slightly to 96 % at cut plot 2. In the following years, occasional increased leaching from the E-horizon occurred at cut plot 1, but retention returned to the same level as before cutting at cut plot 2. The decreased retention of $PO_4^{3-}$ by the E-horizon at cut plot 1 was compensated for by increased retention in the B-horizon. Thus, the flux of $PO_4^{3-}$ below the B-horizon remained small, < 0.02 kg P $ha^{-1}a^{-1}$, at both plots. The relative flux of $PO_4^{3-}$ from below the B-horizon increased from 3 (±1) and 6 (±2) % of total throughfall deposition at cut plots 1 and 2, respectively, to a maximum of 8 % in 1999 at cut plot 1 and to 29 % in 1998 at cut plot 2. Over the three-year study period (1997-1999), 3.7 and 4.5 kg $ha^{-1}$ more $PO_4^{3-}$ was retained in the mineral soil, primarily in the B-horizon, at cut plots 1 and 2, respectively, compared to the period before cutting.

Some 45 % of the flux of total P from below the O-horizon was organic (Table 3) before clear-cutting (0.71 kg $ha^{-1}$ $a^{-1}$). The relative and actual flux of organic P from below the E-horizon was small (0.06 kg $ha^{-1}$ $a^{-1}$) and decreased further as it passed through the B-horizon (0.002 kg $ha^{-1}$ $a^{-1}$ from below the B-horizon). After cutting, the estimated flux of organic P from below the O-horizon increased, averaging 0.46 kg $ha^{-1}$ $a^{-1}$, but organic P as a proportion of total P decreased to 25 %. The flux of organic P below the E-horizon also increased a little after clear-cutting, averaging 0.13 kg $ha^{-1}$ $a^{-1}$, but decreased as a proportion of total P. The flux of organic P from below the B-horizon remained small (0.001 kg $ha^{-1}$ $a^{-1}$) after cutting.

***Table 3*. Linear regression models for estimating weekly total S (y) concentrations from $SO_4^{2-}$-S (x) and for estimating total P (y) concentrations from $PO_4^{3-}$-P (x) (mg $L^{-1}$) in bulk precipitation (BP), throughfall (TF) and percolate from below different soil horizons before (1993-1996) and after (1997-1999) clear-cutting.**

| Before clear-cutting | | After clear-cutting |
|---|---|---|
| BP | S: y=1.15x (R²=0.94, n=93 in 1993-1999)<br>P: y=1.12x (R²=0.97, n=93 in 1993-1999) | |
| TF | S: y=1.10x (R²=0.96, n=22)<br>P: y=1.53x (R²=0.79, n=22) | |
| O-horizon | S: y=0.32+1.29x (R²=0.78, n=122)<br>P: y=1.45x (R²=0.93, n=123) | S: y=0.54+1.21x (R²=0.79, n=129)<br>P: y=1.25x (R²=0.99, n=130) |
| E-horizon | S: y=0.65+1.24x (R²=0.50, n=138)<br>P: y=1.43x (R²=0.93, n=123) | S: y=1.85+0.98x (R²=0.05, n=178)<br>P: y=0.03+1.14x (R²=0.99, n=130) |
| B-horizon | S: y=1.27+0.71x (R²=0.35, n=36)<br>P: y=1.21x (R²=0.98, n=37) | S: y=2.11x (R²=0.79, n=26)<br>P: y=1.13x (R²=0.96, n=26) |

## Calcium, magnesium and potassium

The deposition fluxes of base cations to the forest floor decreased after clear-cutting (Figure 4). Potassium deposition decreased the most, over 80 % compared to the period before cutting. However, $K^+$ concentrations increased immediately after clear-cutting in the percolates from below the soil horizons (Figure 5). Occasional increased concentrations of $Ca^{2+}$ and $Mg^{2+}$ were also observed after clear-cutting, but the changes were much smaller than those of K (Figure 6).

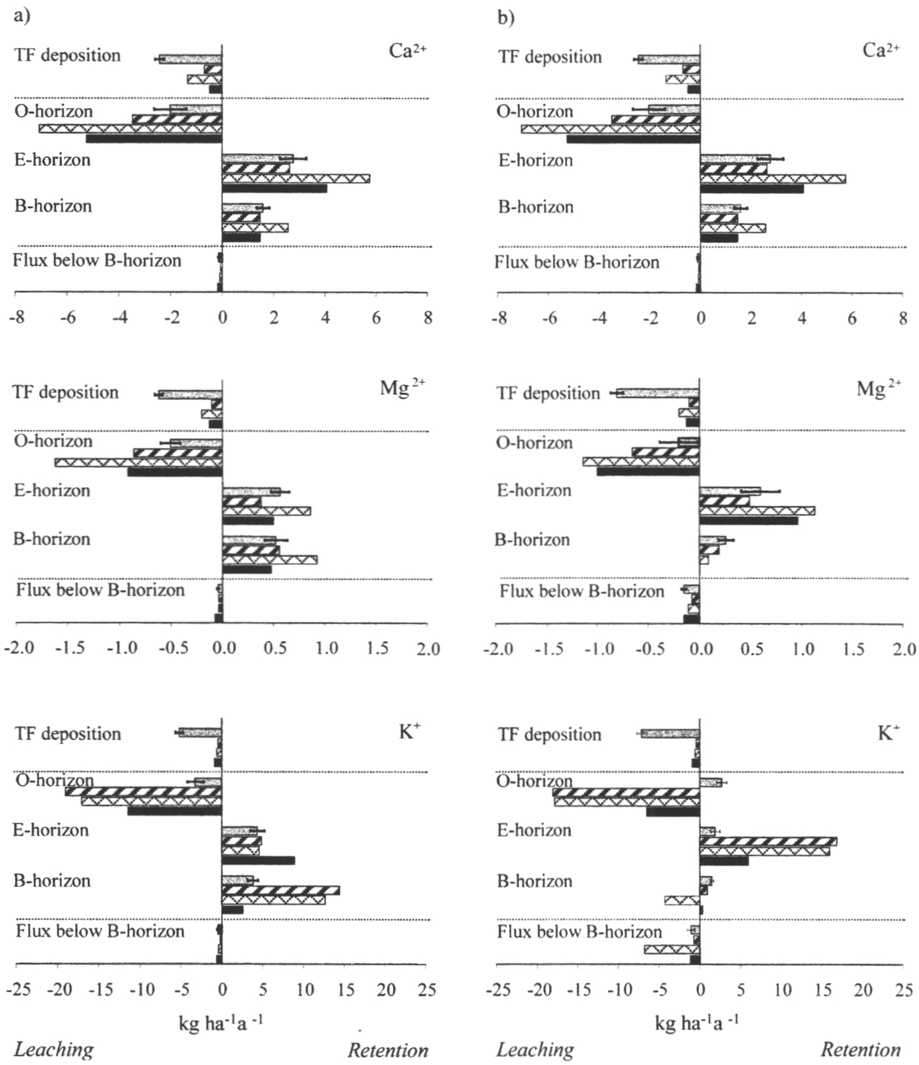

***Figure 4.* Mean annual total throughfall deposition (TF), flux below the B-horizon and the leaching from or retention in the different soil horizons of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ before (1993-1996) and after (1997-1999) clear-cutting at cut plots 1 (a) and 2 (b). For abbreviations and symbols see Figure 2.**

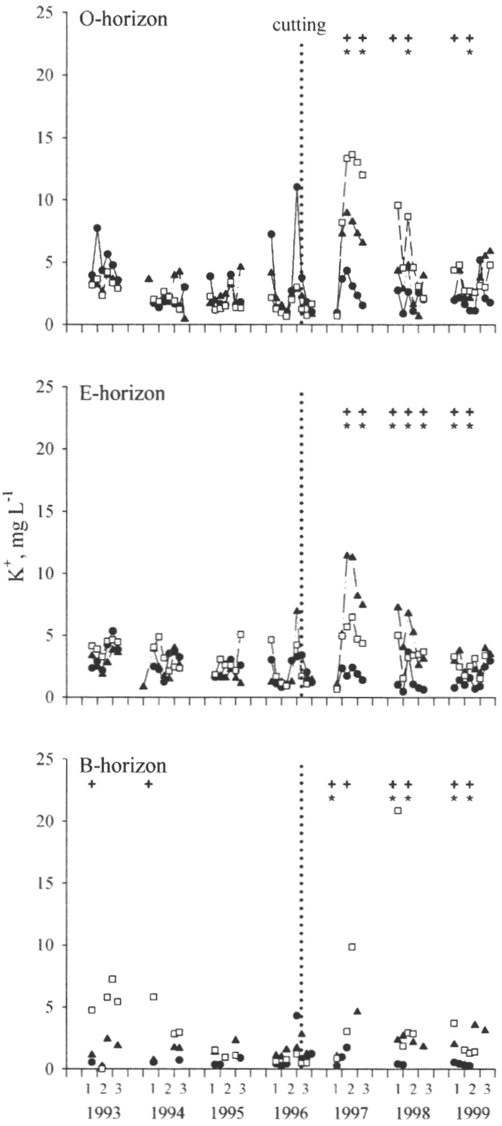

***Figure 5.*** **Mean monthly K concentrations (mg L$^{-1}$) in the soil percolate from below each horizon at uncut plot and cut plots 1 and 2 in 1993-1999. Clear-cutting was done in September 1996. For abbreviations and symbols see Figure 1.**

The observed increases in base cations concentrations after clear-cutting resulted in increased leaching of $Ca^{2+}$, $Mg^{2+}$ and $K^+$ from the O-horizon (Figure 4). The average increase in $Ca^{2+}$ leaching was threefold at both cut plots in 1997-1999. The corresponding increase for $Mg^{2+}$ was twofold at cut plot 1 and fivefold at cut plot 2 and those for K, fivefold and 14-fold. The increased flux of all base cations to the mineral soil was mostly retained by the E-horizon at cut plot 2. At cut plot 1, the E-horizon retained most of the $Ca^{2+}$ and the B-horizon most of the

$K^+$ and $Mg^{2+}$ fluxes (Figure 4). Some increased fluxes of $Mg^{2+}$ in 1999 at cut plot 1 and $K^+$ in 1998 at cut plot 2 below the B-horizon were observed after cutting, but were small compared to fluxes from below the O-horizon.

After cutting, monthly concentrations of $PO_4^{3-}$ and $K^+$ were significantly correlated in the percolate from below the O- (r=0.87, n=40) and E-horizons (r=0.87, n=39), but not so in the percolate from below the B-horizon. Correlations between percolate $PO_4^{3-}$ and other base cations concentrations were negligible as were those between $SO_4^{2-}$ and base cations.

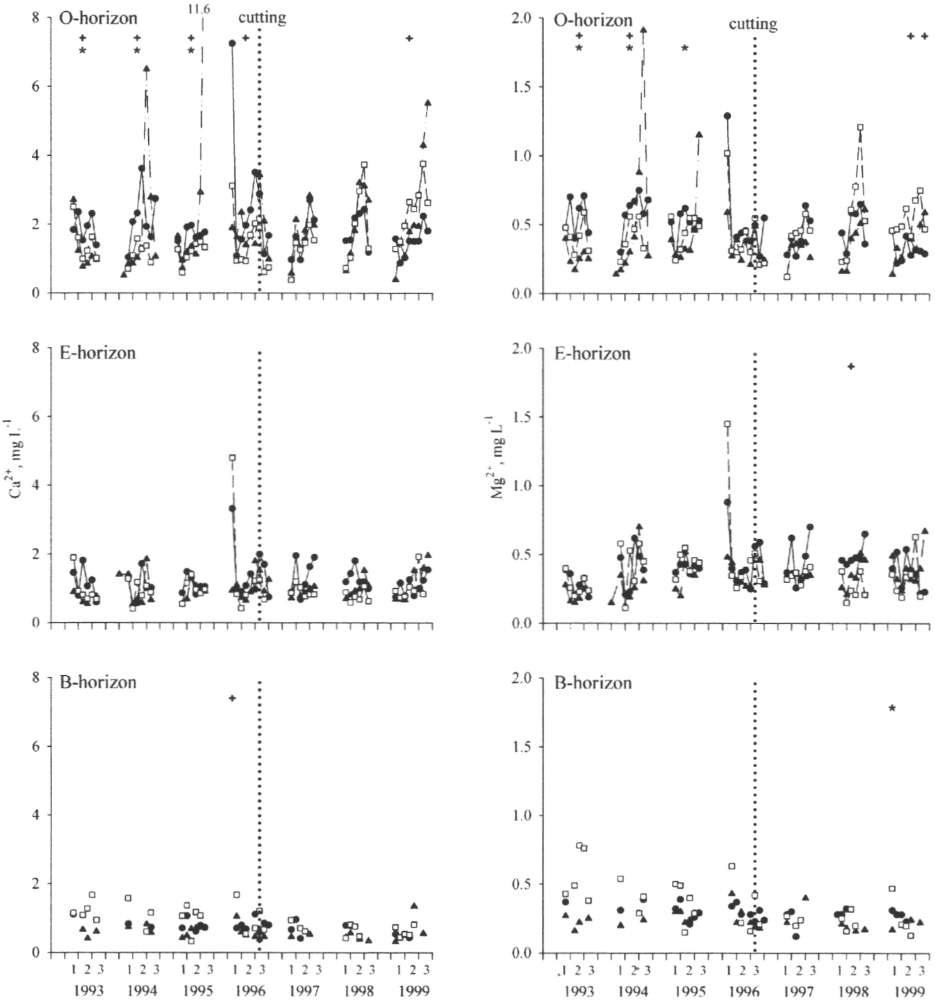

*Figure 6.* **Mean monthly Ca and Mg concentrations (mg $L^{-1}$) in the soil percolate from below the different soil horizons in 1993-1999 at uncut plot and cut plots 1 and 2. Clear-cutting was done in September 1996. For abbreviations and symbols see Figure 1.**

# Discussion

## Changes in S and P fluxes

Over half of the annual throughfall $SO_4^{2-}$ deposited was retained by the O-horizon before clear-cutting (Piirainen et al. 2002a). This probably occurred through vegetation uptake and microbial immobilization. Uptake of S by coniferous stands in the boreal zone varies from 1.5 to 6 kg $ha^{-1}$ (Finér, 1991; Nihlgård, 1972). In the first year after clear-cutting, the retention of $SO_4^{2-}$ decreased, with outputs exceeding input at plot 1, even though deposition inputs declined. Thus the logging residues and O-horizon were a net source of $SO_4^{2-}$ during the first year after clear-cutting. This may be explained by the sudden reduction in uptake and the net mineralization of organic. Nevertheless, the amount of S mineralised did not compensate for the loss of throughfall deposition and litterfall inputs on an annual basis (Table 1), and the flux of $SO_4^{2-}$ from below the O-horizon was less than before cutting. During the second and third years, however, the level of $SO_4^{2-}$ retention by the O-horizon recovered, probably in response to increased uptake by the understorey vegetation and microbial immobilization and/or reduced mineralization of organic S.

Reduced uptake probably also explains the observed decrease in the proportion of $SO_4^{2-}$ inputs retained by the E-horizon. The decrease in the proportion of $SO_4^{2-}$ retained by the B-horizon, however, is due to an input of $SO_4^{2-}$ from the B-horizon itself. While the inputs of soluble organic S to the B-horizon increased after clear-cutting, the leaching outputs of organic S decreased. Sulphate outputs from the B-horizon may therefore have been maintained after clear-cutting by the mineralization of soil organic S. Another internal source of $SO_4^{2-}$ that could enrich the B-horizon percolate is the pool of adsorbed $SO_4^{2-}$. Adsorption of $SO_4^{2-}$ by the soil is controlled the amount and degree of saturation of amorphous Al and Fe hydroxides (Harrison & Johnson 1992; Nodvin et al. 1986; Nodvin et al. 1988) and, depending on pH, is partly reversible (Harrison & Johnson 1992; Karltun 1995; Gobran et al. 1998). By definition, the B-horizon in Podzols is enriched in Al and Fe hydroxides and the soil at Kangasvaara was no exception (Piirainen et al. 2002a). The increase in percolate pH following clear-cutting could have resulted in desorption of $SO_4^{2-}$ from this pool thus enriching the percolate. However, the flux of $SO_4^{2-}$ below the B-horizon did not increase after clear-cutting. Neither have studies in temperate forests reported increased $SO_4^{2-}$ leaching after cutting (Fuller et al. 1987; Likens et al. 1970; Likens et al. 1977).

Before clear-cutting, $PO_4^{2-}$-P accounted for less than 55 % of total P, which is similar to proportions reported by Qualls et al. (2000) and Yavitt & Fahey (1986) for undisturbed forests. The observed elevated $PO_4^{3-}$ concentrations and the increased flux of total P from below the O-horizon over the three year period after cutting (6.9 kg $ha^{-1}$) may have been due to releases from logging residues (Table 1). The increase in $PO_4^{3-}$ leaching indicates that microbial activity was not limited by $PO_4^{3-}$ availability (Berg & Staaf 1980). The dominance of total P by $PO_4^{3-}$ in percolate from the O-horizon after clear-cutting (75 %) has also been observed in other studies (Qualls et al. 2000).

The increased flux of total P after clear-cutting was mostly retained in the E- and B-horizons. Phosphate anions and soluble organic P complexes are known to precipitate with Al and Fe in solution to form of highly insoluble Al and Fe-phosphate complexes and to be strongly adsorbed onto surfaces of Al and Fe of hydrous oxides (Schnitzer 1969; Stevenson & Cole 1999; Wood et al. 1984). The podzolic soil at Kangasvaara is rich in Al and Fe oxides, especially the upper B-horizon (Piirainen et al. 2002a). Strong $PO_4^{3-}$ sorption by the mineral soil would explain why increased leaching of P from clear-cut areas to the watercourses generally does not occur (Briggs et al. 2000; Stevens et al. 1995; Yanai 1998). However, increased stream water total P concentrations after cutting have been reported (Adamson & Hornung 1990; Ahtiainen & Huttunen 1999). This probably happens when runoff takes place mainly through the organic horizon, as during snowmelt, or if the ground-water table raises resulting in anaerobic conditions under which P becomes soluble (Armstrong 1982).

## Leaching of cations and accompanied anions

Clear-cutting resulted in the abrupt cessation of regular throughfall and canopy litterfall inputs of base cations to the forest floor but a large and single addition in the form of logging residues (Table 1). Potassium is rapidly leached and mineralized from logging residues (Staaf & Olsson 1994; Stevens et al. 1995; Titus & Malcolm 1992), which would account for the observed increase in $K^+$ fluxes to the mineral soil. The elevated flux of $Mg^{2+}$ from below the O-horizon was probably also related to the leaching from logging residues. However, the increases in the $Ca^{2+}$ fluxes were probably also due to leaching from the O-horizon (Hendrickson et al. 1989).

Before clear-cutting, the leaching of base cations from the O-horizon was most strongly correlated with the leaching of $SO_4^{2-}$ (Piirainen et al. 2002a). After cutting, this correlation disappeared and was replaced by the correlation between $K^+$ and $PO_4^{3-}$. Calcium and $Mg^{2+}$ concentrations were correlated with neither $PO_4^{3-}$ or $SO_4^{2-}$ concentrations after clear-cutting. Soluble organic matter concentrations, measured as dissolved organic carbon, in the percolate from below the O-horizon did increase after clear-cutting (Piirainen et al. 2002b) and the increased leaching of $Ca^{2+}$ and $Mg^{2+}$ to the mineral soil was therefore probably accompanied by organic anions, as were also the fluxes of base cations through the B-horizon.

Before clear-cutting, the forest ecosystem acted as a sink for deposition inputs of $K^+$ and $Ca^{2+}$, and for $Mg^{2+}$ and the deposition inputs and outputs from below the B-horizon were in balance (Piirainen et al. 2002a). After clear-cutting, the deposition inputs of base cations decreased resulting in less positive annual mass balances (Figure 4). However, the difference between input and output fluxes for the mineral soil (E+B-horizon) indicates that the soil pool of base cations increased after clear-cutting and that there was little additional base cations leaching below the B-horizon.

## Conclusions

The deposition inputs of $SO_4^{2-}$, $Ca^{2+}$, $Mg^{2+}$ and $K^+$ to the forest floor decreased immediately after clear-cutting while there was little change in $PO_4^{3-}$ inputs. Although there was increased leaching inputs of $Ca^{2+}$, $Mg^{2+}$, and especially $PO_4^{3-}$ and $K^+$ to the mineral soil over the first three years after clear-cutting, the source being the logging residues and organic horizon, most was retained in the mineral soil and there was no significant change in the leaching fluxes from below the B-horizon. It remains to be determined if these nutrients will continue to be retained on-site, either in the soil or in circulation within developing vegetation in the long-term, or whether they will eventually leach to ground and surface waters.

## Acknowledgements

The authors wish to thank laboratory staff of the Finnish Forest Research Institute at Joensuu Research Centre and the Central Laboratory, Vantaa, for analysing the water samples, Ms. Leena Karvinen for preparing the Figures and Tables, Mr. Jaakko Heinonen for statistical advise and the field staff for their competent field work. We also wish to acknowledge the Finnish Forest and Park Service for making the Kangasvaara site available for our research project and Ministry of Agriculture and Forestry for partly financing this study.

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# PAPER V

Finér, L., Mannerkoski, H., Piirainen, S., & Starr, M. 2002. Carbon and nitrogen pools in an old-growth, Norway spruce-mixed forest in eastern Finland and changes associated with clear-cutting. Forest Ecology and Management (in press).

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Forest Ecology and Management 5880 (2002) 1–14

Forest Ecology
and
Management

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# Carbon and nitrogen pools in an old-growth, Norway spruce mixed forest in eastern Finland and changes associated with clear-cutting

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Received 11 January 2001; received in revised form 19 October 2001; accepted 18 December 2001

## Abstract

The amount of carbon (C) and nitrogen (N) removed in stems after clear-cutting a forest in eastern Finland is compared with ecosystem pools (above- and below-ground tree and understorey vegetation biomass + organic layer + 0–60 cm mineral soil layer) and fluxes before clear-cutting. The tree stand was an old-growth mixed coniferous forest dominated by Norway spruce. The ecosystem pools of C and N before clear-cutting were 175 536 and 2848 kg ha$^{-1}$, respectively. Most (62%) of the C was in living vegetation, mainly trees (60%), whereas most of the N pool was in the soil (80%). C and N pools in understorey vegetation were small (<2% of ecosystem pools) and those in dead tree compartments were somewhat greater (C: 8%; N: 4%). The annual net uptake (defined as net accumulation in tree biomass + aboveground litterfall) of C by trees was 2013 kg ha$^{-1}$ and that of N was 16.1 kg ha$^{-1}$. The annual return to the forest floor as aboveground litterfall was 958 kg ha$^{-1}$ of C and 12 kg ha$^{-1}$ of N, corresponding to 48 and 75% of annual net uptake, respectively. During clear-cutting, some 239 m$^3$ ha$^{-1}$ of stemwood (overbark) were removed from the site, corresponding to 32% (C) and 3.0% (N) of the pre-harvest ecosystem pools; branches and foliage were left on site. The pool of C in living vegetation was reduced by 89% and that of N by 81% while the C pool in dead vegetation increased fourfold and that of N somewhat less. Our results indicate that clear-cutting will affect the C and N pools and fluxes of forest ecosystems significantly, even though some living trees are left on site. © 2002 Published by Elsevier Science B.V.

*Keywords:* Biomass; Birch; Harvesting; Nutrient fluxes; Norway spruce; Soil organic matter; Scots pine

## 1. Introduction

Boreal forest ecosystems in Fennoscandia and elsewhere are known to contain large amounts of carbon (C) and nitrogen (N) (Tamminen, 1991, 2000; Dixon et al., 1994; Kauppi et al., 1997; Liski and Westman, 1997b). However, most studies in Fennoscandia have dealt with forests managed to produce a single species through artificial regeneration and periodic selective felling, and relatively little is known about semi-natural, old-growth forests. Such forests are relatively rare in Finland, with less than 10% of the total forest area having stands older than 140 years and in southern Finland even less, 1.7% (Sevola, 1999). However, the structure and species composition of managed forests throughout Finland is set to change as a result

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0378-1127/02/$ – see front matter © 2002 Published by Elsevier Science B.V.
PII: S0378-1127(02)00019-1

of changes in management practices that took place during the 1990s. More forests will become like old-growth forests, i.e. mixed composition and retaining a significant proportion of dead trees to sustain biodiversity. Old-growth forests contain more biomass than younger managed forests and their nutrient cycling is tight, with minor losses by leaching (e.g. Miller, 1984). In view of increasing awareness about climate change and the role of forests as C sinks, information about the C and N pools and fluxes of such stands is needed (Karjalainen, 1996a; Liski et al., 1998).

Forest ecosystem pools and fluxes of C and N are strongly affected by disturbances, including forest management and fire (Johnson, 1992; Ryan et al., 1992; Bengtsson and Wikström, 1993; Karjalainen, 1996a,b). Nowadays, wild fires in Finland are effectively controlled but the forests will continue to be managed for timber and pulp production. The effects of strong management tend to reduce soil C and N pools while the effects of less intensive management practices have been ambiguous (Johnson, 1992; Olsson et al., 1996). Long-term simulation studies have indicated that the total ecosystem C pools in unmanaged, boreal forests are bigger than in managed forests (Bengtsson and Wikström, 1993; Karjalainen, 1996b). More experimental data are needed to validate the differences in the C and N pools and fluxes between managed and unmanaged forest ecosystems.

Removal of timber obviously results in an immediate loss of C and N from the ecosystem; the greatest reduction resulting from whole-tree harvesting (Morrison and Foster, 1979; Huntington and Ryan, 1990; Olsson et al., 1996). With conventional stem-only harvesting where only the stems with bark are removed from the forest, soil C and N pools are initially increased by the addition of logging residues to the forest floor. Although stem-only harvesting is still the most common practice, recent European legislation promoting the use of biomass for energy production may be expected to lead to increased whole-tree harvesting, especially at final cutting. Besides the loss of C and N associated with timber removals, harvesting also affects a number of soil processes ~~this~~ in turn affect the fluxes of C and N leaving soil. which

Litterfall and nutrient uptake by trees decrease and decomposition, denitrification, respiration, leaching and runoff all tend to increase after harvesting, resulting in losses of soil C and N ~~mainly~~ by leaching (Vitousek et al., 1981; Rosén, 1984; Tiedeman et al., 1988; Reynolds et al., 1995). However, the increase in leaching tends to disappear within a few years and is small compared to the biomass removals at harvesting (Mann et al., 1988; Reynolds et al., 1995). Leaching losses are also small compared to the soil pools and have no detectable effect on the soil C and N pools (McColl and Grigal, 1979; Sollins and McCorison, 1981; Huntington and Ryan, 1990). However, in some studies significant leaching losses of soil C and N have been reported (e.g. Mroz et al., 1985). The C and N pools of forests can also be severely reduced by heavy wild fires and by harvesting combined with intensive soil preparation (Johnson, 1992; Trettin et al., 1992).

The aim of this study is to give a detailed report on the amount of C and N stored in the vegetation biomass and soil of an old-growth, mixed boreal forest ecosystem before and after clear-cutting. Besides calculating C and N removal with harvesting, the changes in the fluxes due to removal of trees are also discussed. The results presented are a part of the VALU-project, the aim of which is to study the effects of clear-cutting on nutrient cycling, soil properties, soil water and stream water quality and ~~water~~ yield (Finér et al., 1997).

## 2. Material and methods

### 2.1. Site

The data presented were collected from three 50 m × 50 m permanent sample plots located in the Kangasvaara catchment in eastern Finland (63°51′N, 28°58′E, 220 m a.s.l.) (Finér et al., 1997). The plots were established in an old-growth, mixed coniferous stand dominated by Norway spruce (*Picea abies* Karsten), but with Scots pine (*Pinus sylvestris* L.), white and silver birch (*Betula pubescens* Ehrh. and *B. pendula* Roth) and European aspen (*Populus tremula* L.) also present (Table 1). There were dead trees in the stand, both standing and lying on the ground. The stand was uneven-aged, with the average age being 140 years and the oldest trees being 170 years old. The site type was *Vaccinium–Myrtillus*-type according to the classification of Cajander (1949). The field layer vegetation was dominated by dwarf shrubs (*Vaccinium vitis-idaea* L. and *V. myrtillus* L.) and the bottom layer

Table 1
Tree stand characteristics of the sample plots ($n = 3$) before harvesting in 1996 (standard deviation in parentheses)

| Characteristics | Pine | Spruce | Deciduous trees |
|---|---|---|---|
| Living trees | | | |
| No. of stems per hectare | 123 (94) | 1220 (178) | 242 (102) |
| Mean height (m) | 20.8 (1.0) | 8.1 (1.0) | 10.5 (3.1) |
| Mean breast height diameter (mm) | 308 (41) | 100 (18) | 115 (37) |
| Stem volume with bark ($m^3$ $ha^{-1}$) | 87 (61) | 137 (55) | 36 (9) |
| Volume growth with bark ($m^3$ $ha^{-1}$ $a^{-1}$) | 1.0 (0.8) | 1.9 (0.3) | 0.3 (0.1) |
| Dead trees | | | |
| Standing, volume ($m^3$ $ha^{-1}$) | 6 (3) | 2 (1) | 6 (4) |
| Fallen, volume ($m^3$ $ha^{-1}$) | 13 (34) | 4 (3) | 6 (0.4) |

by feather mosses (*Pleurozium schreberi* Brid. and *Hylocomium splendens* (Hedw.) B. S. & G.).

Clear-cutting with the conventional "stem-only" method was carried out according to the recommended standard practices in August–October 1996. Accordingly, only stands classified as mature for final cutting and not adjacent to the stream draining the catchment were felled. Two of the permanent plots were located in an area (8.3 ha) designated for clear-cutting while the third remained uncut. The total harvested area in five different compartments was 19.4 ha, i.e. 35% of the catchment area. Some living trees and all dead trees were left standing on the harvested area. All trees were removed from the two harvested plots.

The upland soils in the catchment were developed in sandy till with a clay content of <2% and classified as Haplic Podzols (FAO, 1988). The underlying bedrock is granodiorite. The stone content (>20 mm) of the upper 0.3 m layer of soil, determined by the steel rod penetration method of Viro (see Tamminen and Starr, 1994), was 0.28 $m^3$ $m^{-3}$. The mean annual air temperature was +1.2 °C, the effective temperature sum, 922 °C (sum of daily mean temperature exceeding +5 °C), and the amount of precipitation 709 mm for the period 1992–1995 (for more details about the site, see Finér et al., 1997).

### 2.2. Stand measurements and estimation of biomass, C and N pools

#### 2.2.1. Aboveground biomass of trees

*2.2.1.1. Living trees.* Biomass estimations were made using allometric functions. The dbh (mean of two breast height diameter measurements in right-angled directions, accuracy 1 mm), tree height (accuracy 0.1 m) and crown length (accuracy 0.1 m) of all living trees (height > 1.3 m) on the three permanent plots were measured in September 1992 and again in July–August 1996.

Sample tree harvesting for derivation of biomass functions was carried out in August–September 1996 in connection with the clear-cutting. Using stratified random sampling based on the distribution of dbh values, nine Scots pine, nine Norway spruce and seven birch trees growing on the two harvested plots were selected, and carefully felled. Derivation of the allometric biomass functions and subsequent calculation of biomass components (stemwood, stem bark, foliage, and branches) at the stand-level was carried out as described in detail by Finér (1989). The number of sample branches from each sample tree was 10 and they were taken at regular intervals from along the entire length of the crown.

*2.2.1.2. Standing dead trees.* The dbh and height were measured from living trees and the degree of decay was determined of all standing dead trees (height > 1.3 m) on the sample plots in August 1996. The following decay classes were used for standing dead stems. (1) Tree recently died, but bark and branches had not yet dropped. (2) Had lost its bark, but some could exist on the base of the stem; deciduous trees had kept their bark but the stemwood had started to decay and most branches had dropped. (3) Stem was largely decayed and soft and only bark kept it standing; usually only deciduous trees were recorded into this class. (4) Stem had lost all branches and was dry; usually only conifers were recorded into this class (Metsäntutkimuslaitos, 1995). The stem volume of dead trees was calculated using well-established allometric functions (Laasasenaho, 1982) and the stand stem volume per hectare of standing dead trees was calculated. A total of eight dead Scots pine, 11 Norway spruce and eight birch trees, representing different dbh classes and degrees of decay, were sampled for the determination of wood density. The biomass of standing dead stems was calculated by multiplying the stem volume per hectare by the mean density of the sample trees (weighted by dry-weight) of each decay class and summing the values.

*2.2.1.3. Fallen dead trees.* The diameter (mean of two measurements taken in right-angled directions, accuracy 10 mm) at both ends of all fallen dead stems (>5 m length and >40 mm diameter), length (accuracy 0.1 m) and degree of decay of the stems on the sample plots were determined in August 1996. Five decay classes were distinguished for the fallen dead stems based on the structural integrity, wood texture and the presence of bark, branches and other vegetation. In decay class 1, the stem had recently fallen with no decay whereas in class 5 the stem was soft, powdery and covered totally by epiphytes (Metsäntutkimuslaitos, 1995). The volume of dead fallen stems was calculated assuming a truncated cone form. A disc was cut from the middle of the stem (13 Scots pines, 19 Norway spruces and 14 birches representing different diameters and degrees of decay) for determination of wood density. The stand-level biomass of fallen dead stems was calculated in the same way as described for standing dead trees.

Fallen dead branches were collected in August 1996 from 20 quadrats (each 4 $m^2$) located systematically along the sides of each of the three permanent plots. Logging residues (without foliage) were collected in June 1997 from 30 quadrats (each 4 $m^2$) located on a line crossing the clear-cut area near the plots. The biomass of the dead branches and logging residues were divided by tree species into different diameter classes (<1, 1–3, 3–5 and >5 cm), their dry mass ψ determined and the results were used to calculate stand-level values.

ψ were

#### 2.2.2. Aboveground biomass of understorey vegetation

Aboveground biomass sampling of understorey vegetation was carried out in July 1996. The aboveground parts of the bush layer vegetation were harvested from the same quadrats as fallen dead branches. The zero level for sampling for all layers was taken as the lower level of the living moss layer. The bush layer mainly consisted of young saplings ($0.5 < \text{height} < 1.3$ m). Aboveground parts of the field and bottom layer vegetation were harvested from smaller subplots (each 0.25 $m^2$) located in the bush layer quadrats. The sampled field layer consisted of the aboveground parts of dwarf shrubs, herbs and grasses, and tree seedlings (height $< 0.5$ m). The bottom layer consisted of bryophytes and litter. The aboveground biomass and N content of the understorey vegetation was calculated from dry mass of the harvested biomass samples.

#### 2.2.3. Belowground biomass of trees and understorey vegetation

The living stump and coarse root biomass of Scots pine and Norway spruce were calculated using the allometric functions presented by Marklund (1988) and those of birch with functions presented by Finér (1989). The biomass of all living fine (diameter $< 2$ mm) and small (2–10 mm) roots was determined by the core method. Twenty cylindrical cores of the soil were systematically taken from each permanent plot. The cores were taken from the organic horizon (core diameter 137 mm) and upper 20 cm mineral soil layer (core diameter 35 mm) in August 1996. The living roots were extracted from the cores by hand and divided into living conifer roots, and living roots of deciduous trees and understorey vegetation. These fractions were further divided into fine and small roots and their dry mass was determined. These data were then used to calculate the stand-level biomass of stumps and roots.

#### 2.2.4. Soil C and N pools

Volumetric samples of the organic layer and mineral soil were taken in August 1993 for calculating the soil C and N pools. On each permanent plot from nine systematically located points, five samples were taken from the organic and 0–20 cm mineral soil layers and pooled by sampling point and layer. These samples were used to determine bulk density. C and N were analysed from three samples pooled from the nine samples on each plot. The size of individual samples from the organic layer was 10 cm × 10 cm and the samples were extracted with a knife. For the mineral soil, samples were taken with a stainless steel corer (diameter 35 mm). For the 20–40 and 40–60 cm mineral soil layers, four pits were excavated close to the permanent sample plots and from each pit two samples were taken horizontally with a corer (diameter 72 mm and height 50 mm) in the middle of the respective layer and pooled to make one sample per layer for analysis. The stone content (>20 mm) of the soil was determined by the steel rod penetration method of Viro (see Tamminen and Starr, 1994) at 30 systematically located points on each plot. The average thickness of the organic layer was 31 mm and the stone content was 28 $m^3 m^{-3}$.

↓ biomass and

### 2.2.5. *Laboratory analyses*

The biomass and soil samples were dried to constant mass at 60 °C. Mineral soil samples were sieved through a 2 mm sieve and the mass of the 2–20 and <2 mm fractions were recorded. Subsamples of biomass and soil samples were taken for dry mass determination at 105 °C. Total N concentrations (% of dry-weight) were determined by the Kjeldahl method (Halonen et al., 1983) in organic soil samples. Carbon concentrations in organic soil samples and also N concentrations in mineral soil (<2 mm fraction) samples were determined using a LECO CHN-600 analyser. Since the soils have no carbonate, the total C concentrations correspond to organic C concentrations.

### 2.2.6. *Calculation of biomass C and N pools, litterfall fluxes and removals at clear-cutting*

The pools of C and N in all biomass compartments in 1996 (before clear-cutting) were calculated from the biomass determined as described above. For the C pool, all biomass compartments were assumed to have a C content of 52%. For the N pool, the samples taken for biomass and wood density determination were also analysed for total N concentrations. The N concentration (% dry-weight) value for each compartment was multiplied by the corresponding biomass value to calculate N pools. In calculating the ecosystem N pools in different aboveground tree compartments, the dry-weight weighted mean N concentration of the respective compartments of the sample trees was used. For Scots pine and Norway spruce, we used the stemwood N concentrations to calculate the N content of the stumps and the N concentration of small roots (diameter 2–10 mm) to calculate the N content of coarse roots since we did not sample these compartments for N concentration analyses. For birch, the N contents of all belowground compartments were calculated using the N concentration values for stemwood since we did not have data for birch root N concentrations. Our procedures probably somewhat underestimated the N pool in roots for birch since the N concentrations of roots are known to be higher than those in stemwood and increase with the decrease in root diameter (Helmisaari, 1991; Finér, 1992). The biomass C and N pools of aspen were calculated using the biomass functions and N concentrations for birch.

The biomass C and N pools of the trees in 1992 were also calculated in order to obtain values for annual C and N net accumulation in tree biomass. The stand measurements made in 1992 were used to calculate the C pool in the same way as described above. For N, the biomass values from 1992 were multiplied by the N concentrations determined in 1996. The mean annual C and N net accumulation in trees were calculated as the difference between 1992 and 1996 values divided by the number of intervening years. The current net annual uptake of C and N was calculated by summing the current annual net accumulation in biomass (aboveground and belowground) and the current annual aboveground litterfall values described below.

Tree canopy litterfall was collected from September 1992 to 1996 using 16 funnels during summer (June–September) and eight funnels during winter (collecting area 0.5 $m^2$, collecting height 1.3 m) in each plot. The funnels were emptied once a week during summer and once a month during winter. Litterfall was separated into branches and twigs, foliage, and the rest and their dry mass and N concentrations were determined.

The amount of C and N removed in stem harvesting, retained in the living and dead trees, and left in logging residues on the harvested area around two of the permanent sample plots were calculated for each tree species. Only trees with dbh > 8 cm were harvested and they were cut as close to the ground as possible and the removed volume was recorded by the harvester with an error of <4%. The species, dbh and height of trees (dbh > 8 cm) left on the site were

↑ on and

measured after harvesting. The biomass of stumps, roots, branches and foliage stripped from the stems and left on site were estimated using the relationships established between these biomass fractions and stemwood from the 1996 data described above. Estimation of the proportion of fine (diameter < 1 cm) branches out of the total branch biomass was based on the logging residue data collected from the site. The mean content of C and N in stumps, roots, stems (over bark), branches and foliage were then used to calculate C and N removals and logging residue inputs to the forest floor.

The C and N pools in the soil compartment (organic layer + 0–60 cm mineral soil layer) were calculated from the C and N concentrations (of the <2 mm fraction in the case of the mineral soil), bulk density and the layer thickness values on each plot. The subsequent values for the mineral soil were also corrected for gravel (2–20 mm) and stone (>20 mm) contents as described by Tamminen (1991). The C and N pools fixed in living roots were subtracted from the soil C and N pools.

## 3. Results

### 3.1. Ecosystem C and N pools

Ecosystem (i.e. total biomass + soil) C and N pools for the old-growth Norway spruce-dominated stand were 175 536 and 2848 kg ha$^{-1}$, respectively (Fig. 1, Table 2). Most of the C pool (62%) was in living vegetation, mainly the trees (60%), whereas most of the N was in the soil pool (80%), with only 15% in the living trees. The mineral soil C and N pools were larger than the pools associated with the organic layer. The pools of C and N in the aboveground understorey vegetation compartment were very small (<2%), and those in dead tree compartments and litter only somewhat larger (8 and 4% of ecosystem C and N pools).

Norway spruce accounted for 53% of the total stem volume of living trees, Scots pine 33% and deciduous tree species 14% (Table 1). The distribution of stem C pools among species was very similar to that of their volumes, but that of N was two times greater in deciduous tree species (30%) than their share of the stand stem volume (Fig. 2). Norway spruces had relatively more N in their foliage and branches compared to Scots pine and deciduous trees (Fig. 2). The stem volume of the dead trees was 12% of the total (living + dead) stem volume. Scots pine and deciduous tree species accounted for most of the stem volume of dead trees as well as for the C and N pools (Fig. 3). The C and N pool in dead branch compartment consisted mainly of the spruce branches.

The current mean annual stem volume growth of the stand was 3.2 m$^3$ ha$^{-1}$ (Table 1). Norway spruce accounted for 60% of the growth, Scots pine 31%

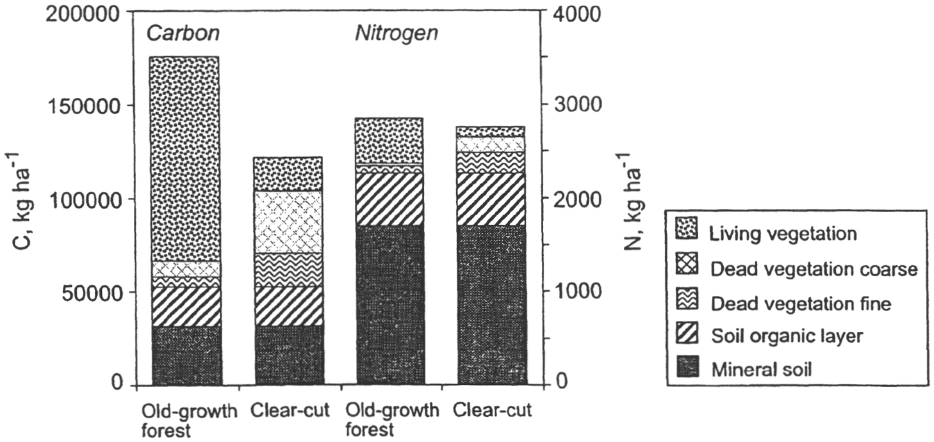

Fig. 1. Distribution of C and N pools into the living and dead vegetation (dead stems, roots, leaves, branches and litter) and soil in the old-growth forest before and after clear-cutting. Fine dead vegetation fraction includes leaves, branches diameter <1 cm and fine roots diameter <2 mm.

Table 2
Carbon and nitrogen pools (kg ha$^{-1}$) in living and dead vegetation and soil before harvesting (standard deviation in parentheses ($n = 3$))

| Compartment | C | N |
|---|---|---|
| Vegetation aboveground | | |
| Living trees | 80068 (8213) | 276.0 (21.0) |
| Bush layer | 677 (362) | 6.6 (3.1) |
| Field layer | 436 (112) | 10.9 (2.7) |
| Moss layer | 751 (135) | 18.1 (3.7) |
| Standing dead stems | 2636 (412) | 6.0 (1.5) |
| Fallen dead stems | 2720 (314) | 11.7 (1.3) |
| Fallen dead branches | 6422 (3890) | 50.5 (29.3) |
| Litter in moss layer | 1673 (462) | 45.7 (13.9) |
| Vegetation belowground | | |
| Stumps and coarse roots | 21875 (2046) | 94.5 (8.1) |
| Fine and small roots of conifers | 3345 (487) | 44.9 (7.7) |
| Fine and small roots of other species | 2075 (385) | 25.1 (2.6) |
| Soil | | |
| Organic layer | 21244 (4256) | 551.0 (65.7) |
| Mineral soil (0–0.6 m) | 31614 (1024) | 1707.0 (99.0) |

and deciduous trees 9%. The mean annual net uptake of C was 2013 kg ha$^{-1}$ and that of N 16.1 kg ha$^{-1}$ (Table 3). Norway spruce took up almost the same proportion of C and N as their share of stem volume growth, whereas Scots pine took up relatively less and deciduous trees relatively more. The return of C and N

Table 3
Annual accumulation of C and N in above- and belowground tree biomass compartments and aboveground litterfall (kg ha$^{-1}$ a$^{-1}$) by species (values are means calculated from three plots; standard deviation in parentheses; total represents the annual uptake)

| | C | N |
|---|---|---|
| Pine | | |
| Aboveground | 166.4 (267.1) | 0.32 (0.70) |
| Belowground | 45.8 (36.9) | 0.21 (0.17) |
| Needle litterfall | 79.1 (47.0) | 0.86 (0.56) |
| Other litterfall | 130.6 (98.5) | 1.19 (0.88) |
| Total | 421.9 | 2.58 |
| Spruce | | |
| Aboveground | 562.6 (170.5) | 2.02 (1.73) |
| Belowground | 112.4 (48.3) | 0.50 (0.29) |
| Needle litterfall | 142.9 (81.6) | 1.86 (0.82) |
| Other litterfall | 389.4 (76.8) | 4.71 (1.00) |
| Total | 1207.3 | 9.09 |
| Deciduous | | |
| Aboveground | 133.1 (33.4) | 0.98 (0.29) |
| Belowground | 34.4 (10.4) | 0.05 (0.02) |
| Leaf litterfall | 69.5 (35.6) | 1.17 (0.54) |
| Other litterfall | 146.4 (17.7) | 2.21 (0.22) |
| Total | 383.4 | 4.41 |
| Stand total | 2012.6 | 16.08 |

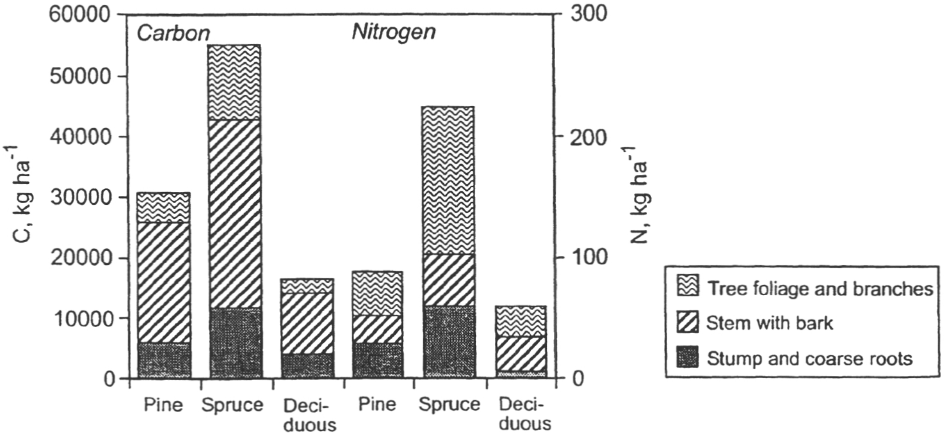

Fig. 2. Distribution of C and N in the different tree compartments of the living Scots pines, Norway spruces and deciduous trees in the old-growth forest.

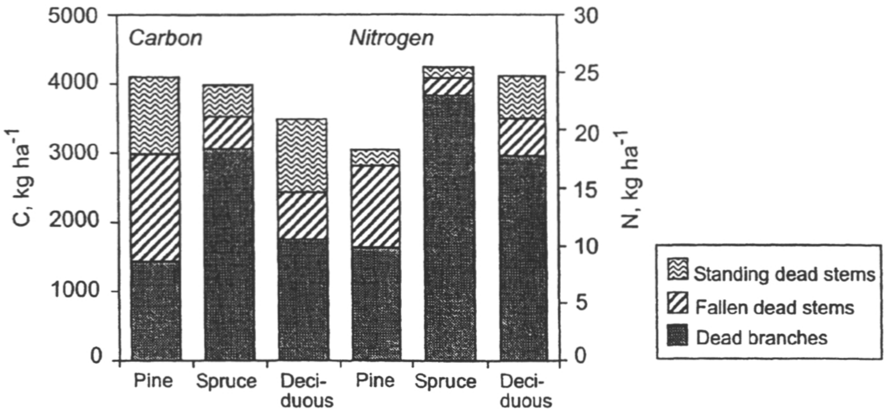

Fig. 3. Distribution of C and N in the different dead tree compartments by tree species in the old-growth forest.

to the forest floor as aboveground litterfall was 48 and 75% of the net annual uptake.

### 3.2. *Removal at clear-cutting*

The stem removal at harvesting amounted to 239 $m^3$ $ha^{-1}$, of which 33% was Scots pine, 54% Norway spruce, and the rest (13%) deciduous trees. In terms of C and N, 56 402 kg $ha^{-1}$ C and 84.8 kg $ha^{-1}$ N were removed from the site (Fig. 4). These removals corresponded to 32 and 3% of the ecosystem C and N pools (Fig. 1). The pool of C in living vegetation was reduced by 89% and that of N by 81%. The C pool in dead vegetation increased fourfold while that of N was somewhat less. The proportion of C in fine fractions of dead vegetation (leaves, fine branches and fine roots) decreased from 40 to 35% and that of N from 72 to 59%, respectively. Some 21 $m^3$ $ha^{-1}$ of living trees were left on the clear-cut area, of which 8 $m^3$ $ha^{-1}$ was Scots pine, 7 $m^3$ $ha^{-1}$ Norway spruce and 6 $m^3$ $ha^{-1}$ deciduous trees. More C was removed with the stems than was left on the site in the form of logging residues. For N, three times more was left in the logging residues than removed in the harvested stems. The amounts of C (8376 kg $ha^{-1}$) and N (29.3 kg $ha^{-1}$) in the living trees left after clear-cutting were 15 and 35% of the amounts removed in stems and, respectively, greater and smaller than the

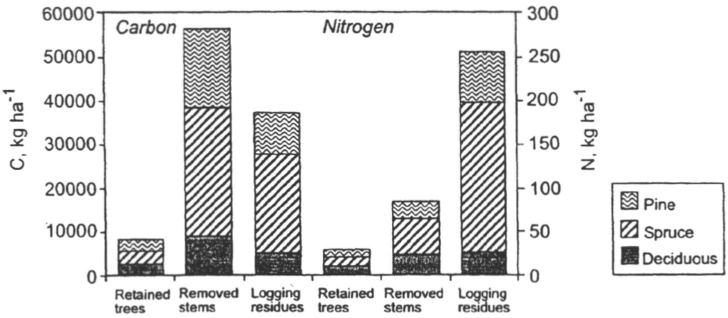

Fig. 4. The C and N amounts fixed at the clear-cutting in the retained and in the removed Scots pines, Norway spruces and deciduous trees and those left in the logging residues.

pools in living understorey vegetation (aboveground). After clear-cutting, most of the ecosystem C was in dead vegetation while the soil remained the largest pool of ecosystem N.

## 4. Discussion

### 4.1. C and N pools and fluxes before clear-cutting

The living vegetation, especially the trees, was the main pool of C, and the soil, was the main pool of N in our old-growth forest ecosystem. Earlier studies concerned with boreal forest biomass, and N pools and fluxes in Fennoscandia are relatively few and have mainly been carried out in managed stands of either pure Scots pine (Mälkönen, 1975; Paavilainen, 1980; Finér, 1989, 1991~~a,b~~; Helmisaari, 1995), Norway spruce (Nykvist, 1971; Nihlgård, 1972; Finér, 1989) or birch (Mälkönen, 1978; Mälkönen and Saarsalmi, 1982). The standing volume, biomass and amounts of N fixed in trees in our old-growth forest were at the upper end of the range reported in these studies. Only Havas and Kubin (1983) and Kubin (1983) have published biomass and nutrients fixed in an old-growth Norway spruce-dominated forest in Finland. In their studies, the total tree biomass and its N content were smaller than in ours, even though their forest had grown unmanaged for 250 years. The more northern location of their site probably explains this difference.

Dead trees and litter constituted significant pools of C and N in our old-growth forest ecosystem (Table 2). Russian studies in an old-growth forest south from our site, found even more in coarse dead wood 17 000 kg C ha$^{-1}$, corresponding to 20% of the total aboveground tree biomass (Krankina and Harmon, 1995). The amount of coarse dead wood in our forest was somewhat underestimated because we did not include stumps and coarse roots of dead trees. Biomass, C and N pools in dead trees have seldom been reported for managed boreal forests. This is presumably because dead trees are removed at thinning and the fraction is not considered important. Most of the C and N in standing and fallen dead trees were associated with Scots pines and deciduous trees, which would be compatible with the successional change away from pioneer birch and pine forest to spruce forest.

The current mean annual stem volume growth of the forest, 3.2 m$^{3}$ ha$^{-1}$, was somewhat less than the average growth of forests in Finland (3.8 m$^{3}$ ha$^{-1}$ a$^{-1}$, Sevola, 1999) but more than that reported for 120-year-old unmanaged *Myrtillus*-type forests in Finland (2.5–2.7 m$^{3}$ ha$^{-1}$ a$^{-1}$) (Koivisto, 1959). The annual net uptake of 2013 kg ha$^{-1}$ C and 16.1 kg ha$^{-1}$ N by the trees was within the range reported for boreal coniferous forests (Mälkönen, 1975; Finér, 1991~~a,b~~; Helmisaari, 1995), whereas the N uptake was much smaller than that in a birch-dominated boreal forest studied by Mälkönen (1978). Our results support the finding that birch binds more N per unit of biomass than conifers (Mälkönen, 1978; Finér, 1989). However, all such values are likely to be underestimates of uptake since they do not include accumulation in fine root biomass, belowground litterfall, dying of trees or canopy exchange.

Throughfall at Kangasvaara has been monitored (Piirainen et al., 1998, ~~2000~~ H2002). Calculations indicate that some ~~58~~ H 71 kg ha$^{-1}$ a$^{-1}$ of organic C was leached from the canopy while 1.2 kg ha$^{-1}$ a$^{-1}$ inorganic N was taken up by the foliage. Adding the canopy leaching of organic C to the current annual net uptake value increases the value for uptake by 3.~~1~~5% to 20~~7~~84 kg ha$^{-1}$. The canopy uptake of N is already accounted for in the net uptake estimate. The annual atmospheric deposition of organic C to the forest was 12.1 kg ha$^{-1}$ and that of total N was 3.8 kg ha$^{-1}$. We have estimated the annual leaching flux of C and N from below the illuvial B-horizon in the permanent plots ~~at~~ H is 2.8 and 0.2 kg ha$^{-1}$, respectively (Piirainen et al., 1998, ~~2000~~ H 2002). There is thus net accumulation of atmospheric inputs of organic C and total N within the ecosystem.

Belowground litterfall, especially that of fine roots, may account for more than half of the annual biomass production (Axelsson and Axelsson, 1986; Helmisaari et al., 2000). However, the net accumulation of C and N in fine and small roots is probably insignificant since fine root biomass has been found to be relatively constant in forests after canopy closure (e.g. Vogt et al., 1983; Laiho and Finér, 1996). The forest at our site was monitored for 4 years before the clear-cutting, but no estimates of tree death were made.

The understorey vegetation represented an insignificant pool of C and N compared to total ecosystem pools, and even to the tree biomass pool. However, element pools in understorey vegetation can be expected to be rather variable since they depend on species composition and abundance, which in turn will depend on many factors, including location, site type, tree species composition and shading. Nevertheless, the understorey vegetation pools of C and N at our site were within the range reported for the few studies carried out in *Myrtillus*- or *Vaccinium*-type forests dominated by Norway spruce or Scots pine (Nykvist, 1971; Mälkönen, 1975; Havas and Kubin, 1983; Kubin, 1983; Helmisaari, 1995; Mäkipää, 1995).

The annual uptake of C and N by the understorey vegetation was not estimated in our study. Mälkönen (1975) reported aboveground biomass production and nutrient uptake of understorey vegetation in three *Myrtillus*- or *Vaccinium*-type Scots pine-dominated forests. He found that the annual aboveground biomass production of understorey vegetation was, on average, 32% of the aboveground total biomass. The corresponding value reported by Helmisaari (1995) for three *Vaccinium*-type Scots pine-dominated forests was 17%. If we assume that the aboveground biomass production of the understorey vegetation at our site was 25% of the total standing biomass and its N content was the same as in the standing biomass, the annual uptake by the understorey vegetation at our site would be 466 kg ha$^{-1}$ C and 8.9 kg ha$^{-1}$ N. These values would correspond to as much as 22% of the annual net uptake of C and 56% of that of N by trees. The relative importance of understorey vegetation in fixing C has been shown to decrease with increasing stand age whereas its role in fixing N remains high, even in old forests (Mälkönen, 1975; Helmisaari, 1995).

The pools of C and N in the organic layer were close to the average reported for *Myrtillus*-type forests in Finland (Tamminen, 1991, 2000; Liski and Westman, 1995, 1997a). However, the organic matter and N pools were at the lower end of the ranges reported by Tamminen (1991, 2000) for the 0–30 cm mineral soil layer. According to Liski and Westman (1995), mineral soil below 1 m depth to the groundwater level may contain 18–28% of the total pool of C in the soil. However, the organic matter at this depth is probably much more stable and beyond forest management effects than that in the upper soil layer. Sampling to this depth would also be prohibitive for most studies.

### 4.2. *Effects of clear-cutting*

We estimated the C and N content of logging residues from the total pools in trees before harvesting and the measured stem volume of harvested trees and their C and N content. We also harvested all logging residues from 30 squares (each 4 m$^2$) after harvesting (data not shown). The logging residues were distributed very irregularly on the site and the coefficient of variation for the logging residue biomass was large (>100%). Thus, we should have carried out a much more intensive sampling to get an accurate estimate of the logging residue C and N pools.

Most of the studies on the effects of forest harvesting in boreal forests have focussed on the possible nutrient losses from the soil and reductions in subsequent forest growth (Mälkönen, 1976; Mälkönen and Saarsalmi, 1982; Jacobson et al., 1996; Rolff and Ågren, 1999). Since forest growth is limited by N availability, most attention has been paid to N (e.g. Mälkönen et al., 1990; Jacobson et al., 2000). Stemwood removals at clear-cutting significantly reduced the ecosystem C pool at our site and the loss was relatively less for N. The annual uptake of C (124 kg ha$^{-1}$) and N (0.6 kg ha$^{-1}$) by the trees remaining after harvesting, assuming that their uptake was not affected by the harvesting, was insignificant compared to what it had been by the whole stand prior to clear-cutting. More living trees were left on our site than for the average Finnish clear-cut areas (3.8 m$^3$ ha$^{-1}$, Eeronheimo et al., 1997). If all the trees had been cut and harvested at our site, as would have been the practice earlier, it would have increased the biomass losses of C by 6% and that of N by 11%, resulting in total losses from the forest of 35 and 3.5%, respectively. If the proportion of deciduous trees left on site had been higher, the losses, of N in particular, would have been lower. If whole-tree harvesting, in which the foliage and branches are also removed from site, had been practised at our site the C and N losses would have increased to 46 and 9.7% of the total ecosystem pools, respectively.

In addition to the immediate losses of C and N associated with the removal of stemwood, clear-cutting affects many biogeochemical processes that bring about further changes in the C and N pools and fluxes. Once the trees are removed there is no longer a regular supply of litterfall, but there is a large, single flux of logging residues to the forest floor. The average C/N ratio, a factor known to control the initiation of net N mineralisation, was 80 in annual litterfall (Table 3). Before harvesting the average C/N ratio of the fine dead vegetation fraction was 65 and that of the coarse fraction was 256 and in the logging residues the respective ratios were 84 and 199 (calculated from the data presented in Fig. 1). A threshold C/N ratio of 60–110 for the initiation of net N mineralisation of Scots pine needle litter has been reported in Swedish studies (Berg and Staaf, 1980; Berg and Ekbohm, 1983). The high C/N ratios at our site, especially in the coarse dead vegetation fractions, therefore would indicate low N mineralisation, and even immobilisation in logging residues (Berg and Staaf, 1980; Berg and Ekbohm, 1983; Titus and Malcolm, 1999; Hyvönen et al., 2000). However, the C/N ratio of different biomass compartments varies, being much lower in foliage than in branches and some net mineralisation may therefore, have occurred, as indicated by a small increase in mineral N leaching we observed after the clear-cutting (Piirainen et al., 2000). /2

The removal of trees at clear-cutting represents a sudden decrease in nutrient uptake from the soil, which when combined with a greater amount of water reaching the forest floor, increases the risk for mineral N leaching. In boreal coniferous forests, mineral N is growth limiting and is largely immobilised in the soil. The understorey vegetation is also affected by changes in light conditions (Kellomäki et al., 1977), nutrient availability, mechanical disturbance, and covering by logging residues at the time of cutting (Olsson and Staaf, 1995), and subsequently changes due to succession. A strong flush in the growth of ground vegetation immediately after clear-felling would help immobilise mineral N. Furthermore, much of the available N in the soil may also become rapidly immobilised, at least temporarily, in soil microbes or dead organic matter after clear-cutting, leaving little for leaching.

We consider the soil C and N pools at Kangasvaara unchanged immediately after harvesting. This is confirmed by our initial analysis of soil water fluxes of dissolved organic carbon (DOC) and N. Up to 3 years after clear-cutting there was only a small increase in the leaching flux of DOC and total N, dominated by organic N, from below the illuvial B-horizon (Piirainen et al., 2000). There are only a few studies on the long-term changes in soil C or N pools after clear-

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cutting of boreal forest. The results of Olsson et al. (1996), 15–16 years after clear-cutting, show that soil C pools decreased 17–22% and those of N 13–22% on sites with Norway spruce. A modelling study by Liski et al. (1998) indicated that clear-cutting of unmanaged *Myrtillus*-type forest decreases soil C pool by 14% and that by Karjalainen (1996b) suggests that the total C pool of forest would decrease by 3% within 100 years after clear-cutting. The soil pools can further be decreased by intensifying the harvesting method (Bengtsson and Wikström, 1993), by repeated cuttings (Bengtsson and Wikström, 1993; Karjalainen, 1996b; Liski et al., 1998) or by shortening the rotation length (Aber et al., 1978; Karjalainen, 1996b). In temperate forests observed decreases in forest floor mass and N content level-off after 50–60 years (Covington, 1981; Federer, 1984) or no clear decrease after clear-cutting has been observed (Yanai et al., 2000). In the future we intend to evaluate the long-term effects of harvesting on soil C and N pools and leaching losses from the stand on the stream water draining the catchment.

## 5. Conclusions

We made a detailed study of the ecosystem C and N pools in an old-growth forest and showed that the vegetation is the major pool of C, whereas soil is the major pool of N. The understorey vegetation pools were very small (<2%). Stem removals after clear-cutting reduced the ecosystem pool of C by one-third while the reduction for N was only 3%. It will be necessary to continue monitoring in order to determine the long-term effect of clear-cutting on the pools of C and N in the soil and vegetation as the regenerating stand develops.

## Acknowledgements

We thank the motivated and skilful staff of the Finnish Forest Research Institute, Joensuu Research Centre and the Laboratory of Vantaa Research Centre for technical help in carrying out the field work and laboratory analyses. We also thank the Ministry of Agriculture and Forestry for partially funding the project and Finnish Forest and Park Service for pro-

viding access to the study site and carrying out the harvesting operations.

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METSÄNTUTKIMUSLAITOKSEN TIEDONANTOJA 859, 2002
THE FINNISH FOREST RESEARCH INSTITUTE, RESEARCH PAPERS 859, 2002

ISBN 951-40-1839-7, ISSN 0358-4283