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Agricultural University of Norway

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Modelling nutrient cycling in forest ecosystems

Modellering av næringssyklus i skogøkosystemer

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Errata

Introduction, p.24, line 38: *1934* changed to 2034

Paper 1, p.14, Throughfall, line 2: *2* changed to 3

Paper 1, p.14, Throughfall, line 9: *date* changed to data

Paper 1, p15, Throughfall, line 2: *troughfall* changed to throughfall

Paper 3, p.8, Table 3: *frlch (2)*^{3} changed to frlch(2)^{3,4}

Paper 3, p.23, Conclusion, line 14: *NO3- concentration simulated* changed to NO3- concentration in throughfall simulated

Paper 4, p.3, Method, line 37: *(1986-1934)* changed to (1986-2034)

DISCLAIMER

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Contents

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NISK, Ås,17.12.1997

Sheila H.S.B. Kvindesland

Abstract

KVINDESLAND, S.H.S.B., 1997. The modelling of nutrient cycling in forest ecosystems. Doctor Scientiarum Theses 35. Agricultural University of Norway, Ås.

The effects of nutrient cycling on soil, soil solution and streamwater chemistry have been studied in Norway spruce ecosystems at Birkenes and Nordmoen, S. Norway. The Nutrient Cycling Model (NuCM) has been documented, calibrated and tested.

At Birkenes, S. Norway, the long-term catchment base cation budget implied an annual loss of soil exchangeable (Ca^{2+} + Mg^{2+}) of 0 to 2.5 %, and a gain in K^+. No evidence of decreasing exchangeable Ca^{2+} or Mg^{2+} on podzols was found with time. Of the atmospheric input of nitrogen, amounting to 26 kg ha^{-1} $year^{-1}$, only 2 kg N ha^{-1} $year^{-1}$ occurs as runoff. Of the nitrogen retained in the catchment, 45 % was attributed to net tree uptake and 43 % was assumed to build up other living/dead organic matter.

Nutrient cycling was compared at a 30-year old (P30) and 40-year old (P40) Norway spruce stand at Nordmoen. Soil water fluxes could not be accurately determined from Cl^- concentrations in throughfall and soil layers over a 4-year period. Both canopies retained approximately half of the deposited inorganic N, while organic N was released. Total deposition and Net Canopy Exchange of most ions as well as litterfall fluxes of Mg^{2+}, K^+ and N were greater in P40 than P30, causing higher nutrient fluxes of most elements in throughfall and the forest floor. In mineral horizons in P40 Al^{3+} became the dominant cation in soil solution, while Ca^{2+} was the dominant cation throughout P30. Base cation budgets showed that the most important input to soil was litterfall for Ca^{2+} and weathering for Mg^{2+}. Cycling of K^+ was tight, with throughfall as the dominant input to the soil. The weathering rate of K^+ was estimated very low.

The hydrology sub-model in NuCM was replaced with the SOIL hydrology simulation, due to problems simulating snow. NuCM simulated the concentrations of K^+ and Mg^{2+} well in throughfall, but Ca^{2+} and pH were simulated poorly. Canopy process description was too complex to calibrate effectively, while the large disappearance of inorganic N was not included. Nitrate concentrations in soil solution were simulated poorly due to high simulated concentrations of NO_3^- in throughfall, no description of N immobilisation by soil microbes and inflexibility in nitrification parameters. Sulfate and total monomeric Al were successfully simulated in all horizons. The cycling fluxes of NH_4^+ and Mg^{2+} were simulated well, but Ca^{2+} and K^+ were simulated poorly due to no description of the translocation/accumulation in needles. NuCM's predictions of soil exchangeable pools and critical loads in 2034 under scenarios of reduced S deposition and net tree uptake were far larger than MAGIC's predictions. This was attributed to major differences in the simulated K^+ uptake and Ca^{2+} weathering flux.

Key words: *Nutrient cycling, modelling, NuCM, Norway spruce, Ca, Mg, K, N.*

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Norwegian summary

KVINDESLAND, S.H.S.B., 1997. Modellering av næringssyklus i skogøkosystemer.. Doctor Scientarium Thesis 35. Norges landbrukshøgskole, Ås.

Effektene av næringssyklus på jord, jordvann og bekkevanns kjemiske sammensetning ble undersøkt i granskog-økosystemer i Birkenes og på Nordmoen. En næringssyklus-modell (NuCM) ble dokumentert , kalibrert og testet.

En langsiktig beregning av basekation-budsjettet for Birkenesfeltet antydet et årlig tap for utbyttbart (Ca^{2+} + Mg^{2+}) mellom 0 og 2,5 % og en gevinst av K^+. Det kunne ikke påvises at mengden med utbyttbart Ca^{2+} og Mg^{2+} i podzoler har avtatt med tiden. Av de 26 kg N ha^{-1} $år^{-1}$ som ble tilført i total deposisjon, var bare 2 kg ha^{-1} $år^{-1}$ å finne i avrenning. Av det N som ble holdt tilbake i feltet var 45 % knyttet til netto opptak i trærne, mens 43 % var antatt bundet i annet levende og dødt organisk materiale.

Næringssyklusene ble sammenlignet i en 30 år gammel (P30) og en 40 år gammel (P40) granbestand på Nordmoen. Vannflukser gjennom en 4 års periode kunne ikke bestemmes nøyaktig ved hjelp av Cl^- konsentrasjoner i kronedrypp og jordvann i ulike jordhorisonter. I begge bestand ble ca. halvparten av den årlige tilførsel med uorganisk N forbrukt i trekronene mens organisk N ble produsert. Total deposisjon og netto kroneutveksling for de fleste ioner samt flukser av Mg^{2+}, K^+ og N med strøfall var større i P40 enn i P30, og dette førte til større flukser for de fleste ioner i kronedrypp og i skogbunnen. I mineraljorda under P40 var Al^{3+} det dominerende kation i jordvann, mens Ca^{2+} var dominerende under hele P30 bestanden. Basekation-budsjettene viste at strøfall har størst betydning for Ca^{2+} tilførselen til jorda, mens forvitring har størst betydning for Mg^{2+}. Kalium har et tett kretsløp hvor kronedrypp har størst betydning, og forvitrings hastighet var estimert lav.

Den hydrologiske sub-modellen i NuCM ble erstattet med hydrologi simuleringen fra SOIL, på grunn av problemer med snøsimulering i NuCM. NuCM simulerte konsentrasjonene av K^+ og Mg^{2+} bra for kronedrypp, mens Ca^{2+} og pH ble simulert dårlig. Beskrivelsen av krone-prosessene ble for komplisert til en effektiv kalibrering, mens de store N-forbrukene i trekronene ikke var tatt med. Det ble en dårlig simulering av nitrat konsentrasjonene i jordvannet på grunn av høye simulerte nitrat konsentrasjoner i kronedrypp, manglende beskrivelse av jord-mikrobenes N-forbruk og manglende fleksibilitet ved parametrene for nitrifikasjon. Sulfat og total monomerisk Al ble simulert bra i alle jordhorisonter. Sykliske flukser med NH_4^+ og Mg^{2+} ble bra simulert, mens K^+ og Ca^{2+} flukser ble dårlig simulert på grunn av manglende beskrivelse av translokering i nåler. NuCM`s forutsigelse av jordas lagre med næringsstoffer og tålegrenser i år 2034, ved forskjellige antagelser om tilført svovel og næringsopptak i trær, var mye større enn det som beregnes med MAGIC. Det skyldes hovedsakelig store forskjeller i de simulerte K^+ opptak og Ca^{2+} forvitring.

Nøkkelord: Næringsyklus, modellering, NuCM, gran, Ca, Mg, K, N.

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Introduction

Definitions

Forest ecosystems, nutrient cycling, models and atmospheric deposition are four commonly used terms, which are central to this work. For this reason, I begin by defining and placing them in the context of this work.

The forest ecosystem can be idealistically defined according to Odum (1971) as 'the unit that includes all of the organisms (i.e., the community) in a given area interacting with the physical environment so that a flow of energy leads to a clearly defined trophic structure, biotic diversity, and material cycles (i.e., exchange of materials between living and non-living parts within the system).' In the Nutrient Cycling Model (NuCM) (Munson et al., 1992), the forest ecosystem is vastly simplified to a mature tree stand, four upper soil horizons and the nutrient solution travelling through this system. Hence the community of thousands of different plants, insects and microbes are reduced to one tree species and certain soil processes, e.g. coarse litter decay and nitrification, where these organisms are assumed implicitly to be present.

Nutrient cycling can be best viewed in a diagram (Fig. 1). This work concentrates on the biogeochemical cycle, where chemicals are exchanged within the forest ecosystem. However both the geochemical cycle, which is exchange of chemicals between ecosystems, e.g. atmospheric deposition onto the forest ecosystem or loss of nutrients to groundwater/stream, and the biochemical cycle, the redistribution of chemical within a living organism e.g. translocation in trees, are touched upon in this work.

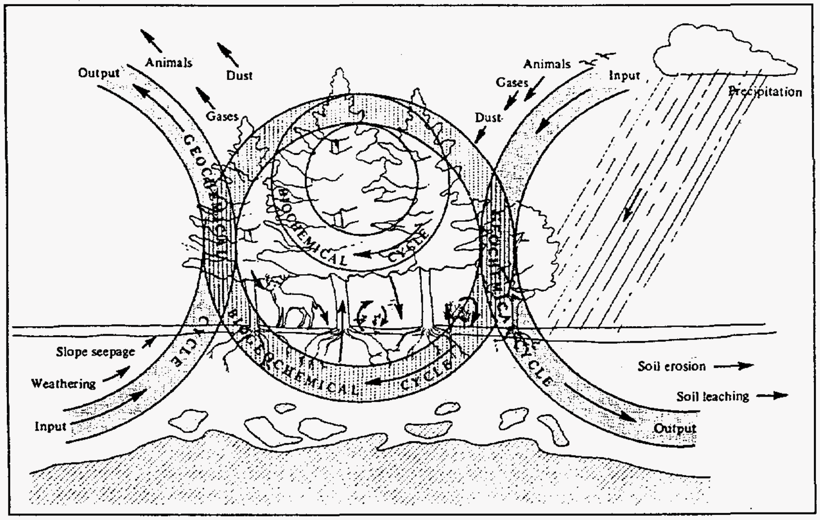

Fig.1. The three major types of nutrient cycle: geochemical (between ecosystems), biogeochemical (within an ecosystem), and biochemical (within an organism). (From Kimmins, 1987)

A model is a representation of reality. A simple descriptive model could be the first stickman or matchstick man that most children draw to represent a person. A simple mathematical model could be an equation representing how forest litter (L) decays with time dependent on the amount of litter present, where k is the rate constant.

$$-d[L] / dt = k [L]$$

The incredible spread of computers into all types of science in the last twenty years has caused a likewise spread in the use and complexity of mathematical modelling in these fields. Modelling is the mathematical description of a whole string of logically related equations, describing e.g. processes in the simplified forest ecosystem. These equations can be written as computer code, enabling a machine to do the computing process. The model interface with the user demands input, parameters and starting conditions supplied by the keyboard. The computer calculates, stores and displays the results. Because my background is soil and water science rather than computer programming, I have not attempted to create and programme a new model. This can be a very lengthy process. Instead I chose to calibrate and test a pre-existing but little tested nutrient cycling model.

Atmospheric substances can be deposited onto land surfaces in two ways according to Fowler (1980) : the direct transfer to and absorption of gases and particles by natural surfaces (dry deposition) and the indirect transfer by the substance being within or on the surface of rain, hail and snow (wet deposition). Acid precipitation or rain denotes precipitation with high amounts of the hydronium ion and increased concentrations of the anions sulfate, SO_4^{2-} and nitrate NO_3^- and the cation, ammonium, NH_4^+ (Overrein et al., 1980). Acid precipitation can also contain heavy metals, trace metals and organic micropollutants. The sulfur and nitrogen stem from both natural sources, e.g natural biological decay processes, seaspray and volcanoes (S) and man-made sources. These manmade sources are primarily due to combustion of coal and petroleum thus producing sulfur and nitrogen oxides, while ammonia is released from livestock, manure and fertilizers. The trends of SO_2 and NO_x emission in Europe are a steady increase since the industrial revolution, with a slight depression during the 1930s recession. When oil consumption increased strongly after 1950, SO_2 emissions doubled between 1950 and 1970. With the international agreements on S reduction in Europe in 1979, 1985 and 1994, the S content in acid deposition has sunk by 40% in Southern Norway since 1980 (Statens forurensingstilsyn, 1997). However the N content continues to stay at a steady level. S emissions have been easier to reduce as they come from point sources, e.g. 57 % due to power plants and industry. As over 70% of the N emission is due to road and ship transport, reducing millions of point sources (cars in Europe) is not an easy task. Although acid precipitation is now on the decline in Europe and N. America, it still causes damage as recently pointed out in Canada (Anon., 1997). Acid precipitation is on the increase in areas of high industrial growth in Asia. This work focuses on the fate of deposited NO_3^- and NH_4^+ in particular and SO_4^{2-} and H^+ in the forest ecosystem.

Brief history of forest nutrient cycling and atmospheric deposition research and modelling

Nutrient cycling in forests has long been a research topic for forest scientists and ecologists. In the 1960s, research focused on fertilisation and nutrient deficiencies or establishing the fluxes through different forest types (Carlisle et al., 1967). Models used in forestry in the 1960s were mainly concerned with yield and forest management techniques. In the 1970s, the International Biological program provided a major impetus to modelling of ecosystem processes (Kimmins, 1987). A report on integrated research in the coniferous forest biome (Waring and Edmonds, 1974) included a section on modelling the forest ecosystem (Sollins et al., 1974).

Acid deposition's threat to fresh water and forest environments became an issue in the late 1960s. Acid deposition and forest nutrient cycling then began to be researched in greater co-operation (Wood and Bormann, 1975). In Norway, as in N. America, sulfur deposition was the chief concern throughout the 1970s and early 1980s (Overrein et al., 1980; Mason, 1990). Understanding stream and lake water chemistry was the main focus of research as declining fish stocks were the most obvious damage linked to acid deposition in Norway. Although S is an essential plant nutrient, its transport through the forest ecosystem to fresh water was possible to model with only minimal involvement of forest nutrient cycling. Models began to be produced in the late 1970s (Reuss, 1980) and in profusion during the 1980s, the Birkenes model (Christophersen et al., 1982), the ILWAS model (Gherini et al., 1985) and the MAGIC model (Cosby et al., 1985a,b). With the exception of ILWAS, most of these models were used to simulate and predict the effects of acid deposition on streams and lakes having passed through forest soil with minimal description of forest nutrient cycling processes e.g. a curve of net uptake in trees over time in MAGIC. Instead the emphasis was on inorganic soil processes such as cation exchange, sulfate adsorption and mineral weathering.

With the reduction of S deposition in Europe, but the still increasing or steady N deposition, together with increased signs of N leaching at various sites (Ågren and Bosatta, 1988; Aber et al., 1989; Dise and Wright, 1995), more focus was placed on modelling nitrogen. It was then that nutrient cycling and acid deposition modelling really began to join forces. At least two models, NICCCE (van Dam and van Breemen, 1995) and MERLIN (Cosby et al., 1997) have evolved to interpret whole forested catchment experiments on nitrogen saturation. The model MERLIN produced by the developers of MAGIC stands for modelling of ecosystem retention and loss of inorganic nitrogen.

Now with the huge interest in global C cycles and the role forests play in this, the cycling of C and N in forests is even more in focus. The new buzz term is landscape modelling where ecosystem processes like decomposition are coupled to a Geographic Information System (GIS) and applied to, e.g.10 ha cells in the White Mountain National Forest (Currie and Aber, 1997). Several conferences have taken place on this topic in 1997 and will soon be taking place (Ninth North American Forest Soils Conference, 1998). Biosphere and global ecosystem models are also being produced, e.g. the model Hybrid v3.0, which treats the daily cycling of carbon, nitrogen and water within the biosphere and between the biosphere and the atmosphere (Friend et al., 1997).

Background

During the first three years of this PhD work, I worked in a joint project involving the Norwegian Forest Research Institute (NISK), the Department of Soil and Water Science at the Agricultural University of Norway (NLH) and the Department of Computer Science at the University of Oslo (UiO). The main aim of the project, 'Modelling of Soil Processes' was to test and develop process-oriented models for nutrient cycling in forest ecosystems. The project was divided into four sections,

(1) Modelling,
(2) Soil chemistry/mineralogy,
(3) Use of existing data and
(4) Transfer old and new data to G.I.S.

Although I have worked with all four sections, my main emphasis has been on sections (1) and (3), Modelling using existing data.

The sub-aims of Modelling (1) were (a) to select useful models for nutrient cycling in forest ecosystems, with the main emphasis on soil and soil water, (b) to investigate the model's data requirements against field data available, (c) to evaluate the field data's quality and representativeness for modelling and thus suggest improvements to future field data collection. The overall aim was to select, calibrate, test and improve forest nutrient cycling model(s).

Using existing data is a necessity, when modelling with data demanding models. Long data series with all imaginable measured data at a site are an unusual commodity in forest research in Norway. Only two sites partly fulfilled these demands, the forested catchment Birkenes and the forest plots at Nordmoen, both situated in Southern Norway. The overall aim was to combine existing data from a number of different research projects, institutes, etc. in order to calculate nutrient fluxes within the two areas (papers 1 and 2). This was particularly important at Birkenes, where research centred at the University of Oslo had for two decades concentrated on understanding and modelling the stream chemistry. Soil and soil water chemistry had been studied in detail, yet forest processes had been incorporated in this research to a minimal degree. As the Forest Monitoring Programme (OPS) had established an intensive forest plot at Birkenes in 1986, the time seemed ripe to combine this great wealth of atmospheric, forest, soil and stream data to obtain better estimates of nutrient fluxes and pools in the Birkenes catchment.

The Sites

Although the choice of these two sites was dictated by previous intensive research on acid deposition, the sites were useful in being opposites of each other. Birkenes represents a more typical piece of Norwegian landscape, a small forested catchment with soils evolving on moraine till, bare granite rock and peaty deposits. Nordmoen in contrast is a flat site with water draining to groundwater and the more homogenous forest is situated on thick glaciofluvial deposits. The forest at Nordmoen is representative of the more economically important forests in the 'Østlandet' region of Norway, where the flatter landscape allows more efficient forest management techniques.

The critical loads for soil in relation to damage to vegetation (molar Ca:Al ratio in the top 50 cm of soil) and in relation to fish (water alkalinity > 0 µeq l-1) have been calculated for both these sites (Wright et al., 1990; Wright et al., 1991). In regions with thin and patchy soils e.g. Birkenes which are characteristic for large areas of Norway, the alkalinity (fish) criterion is usually the more stringent. At Nordmoen however the soil and forest are more sensitive to acid deposition than the surface water.

Birkenes

The Birkenes watershed (58°15'N, 08°15'E) lies 20 km inland and north west of Lillesand. It covers an area of 41.6 ha and consists of 2 main valleys. The lower valley with the main stream being Vestre Tveitdalen and the higher valley with the main bog and a slower flowing tributary stream being Langemyrdalen.

Birkenes has a humid climate with an average precipitation of 1473 mm year^{-1} of which 1147 mm year^{-1} is discharged as stream water for the period 1980-1993 (Statens forurensingstilsyn, 1995). Dry summers with low discharge and wet autumns with high discharge are a typical weather pattern (Christophersen et al., 1982). The average temperature is 6.6 °C.

In 1996, 8.4 kg ha^{-1} of total S and 14 kg ha^{-1} of total N was deposited at Birkenes (Statens forurensingstilsyn,1996). There has been a marked decrease in average annual SO_2 concentrations at Birkenes since 1978, while the sum of total N concentrations shows no marked tendency (Statens forurensingstilsyn, 1996).

The catchment lies within the Precambrian Birkenes granite and the surface materials represented within the catchment are bare rock: 9 %, till: 23%, bogs: 13% and thin humus soils: 55 %. The last glaciation caused their distribution with sandy till being deposited on the stoss sides, where the best forest soils, podzols and brown earths have developed, thicker till is found in the central valley, while the rock was scoured clean on the slopes facing south-east, where bare rock and thin humus soils dominate.

The most important tree species are Norway spruce (79%) and Scot's pine (11%), while deciduous trees such as aspen, birch, oak and rowan tend to dominate in the bogs or on higher slopes. The Norway spruce is on average 80 years-old. The major ground vegetation is a bilberry-bracken community. Greater details can be found in Dale et al., (1974). Details of the history of land use at Birkenes are scarce, but with farms nearby, a combination of forestry and grazing has presumably occurred over the centuries. Historic fish data (Christophersen, 1983) show that brown trout had high spawning activity until around 1950. Older trout were still caught in the small lake below the Birkenes stream in the 1950s and trout disappeared in a larger lake 500 m upstream in the 1960s.

Nordmoen

The study site Nordmoen (60°16'N, 11°06'E) is situated 45 km north of Oslo at an elevation of 200 m. The meteorological station Gardemoen is situated 10 km south of Nordmoen and the annual standard normals (1961-90) show a mean temperature of 3.8 °C (174 days with temperature below 0 °C) and mean precipitation of 862 mm. The snow pack usually remains from December to April.

The average annual total sulphur and nitrogen deposition for 1986-1990 is 9.3 and 10.4 kg ha^{-1} respectively (Statens forurensingstilsyn 1988, 1989, 1990, 1991, 1992), while in 1996 total S and N deposition was 4.5 kg ha^{-1} and 7.9 kg ha^{-1} respectively (Statens forurensingstilsyn, 1997).

It lies on a flat plain of thick glaciofluvial sand deposits (60 m deep) and the groundwater table lies 1-3 m below the surface. The soil is classified as a Cambic Arenosol (FAO) (Stuanes and Sveistrup, 1979).

The main tree species is Norway spruce (*Picea abies* (L.) Karst.) with 3% Scots pine (*Pinus sylvestris* (L.)). The dominant species in the field layer are *Vaccinium myrtillus* (L.) and *Deschampsia flexuosa* (L.) Trin., while moss covers the ground layer. Greater details on vegetation can be found in Røsberg, (1991). The forest history includes selective cutting during the 19th and early 20th century and clearcutting in 1947, followed with replanting of Norway spruce.

Selection of Models

At the time I began this work, two models were widely used in Norway to simulate the effects of acid deposition on forest soils and freshwater. They were the static critical load model, PROFILE (Sverdrup and Warfvinge, 1993) and the dynamic model, MAGIC (Cosby et al., 1985a,b) In addition the Birkenes model (Christophersen et al., 1982) was undergoing reformulations (Taugbøl, 1993) and EMMA (Christophersen et al., 1990) was being developed at certain catchments, including Birkenes.

Researchers at NISK, using the Nordmoen site, had recently partaken in the American Integrated Forest Study (IFS) programme (Johnsen and Lindberg, 1992), which evaluated the effects of atmospheric deposition on nutrient cycling in forest ecosystems. The **Nu**trient **C**ycling **M**odel NuCM (Munson et al., 1992) was the resulting synthesis of hypotheses and processes from this research programme. A dynamic model was necessary to study the effects of acid precipitation on nutrient cycling and soil water, and therefore the initial choice was reduced to the well-tested MAGIC model and the very fresh NuCM model. It is always preferable to join a group of experienced modellers and model users than to start out on one's own.

A workshop on the Comparison of Forest-Soil-Atmosphere Models was to be held in the Netherlands, May 1993 (Van Grinsven et al., 1995). This seemed an ideal event to gain more knowledge on other potential models and the final selection of the model(s) could be made after the workshop. As an introductory exercise, the dynamic model MAGIC was therefore chosen for application to the Solling dataset (Tiktak et al., 1995) used for comparison at the workshop. MAGIC stands for the '**M**odel of **A**cidification of **G**roundwater **I**n **C**atchments'. It is a process-oriented model which quantitatively predicts the effects of acid deposition on catchment soil water and stream water chemistry. The model uses lumped average parameters of soils to represent the whole catchment. The soil equilibrium reactions modelled include dissolved CO_2, a solid phase of aluminium hydroxide, aluminium and base cations in soil water and aluminium and base cations adsorbed on the soil. Long-term changes in deposition, sulfate adsorption, base cation weathering and a more detailed hydrology description are included in the dynamic side of the model. The forestry's involvement in MAGIC is confined to net uptake and release of base cations against time. Nitrogen is modelled to a minimal degree, input as total deposition and channelled either to tree uptake or soil leachate. The developers of MAGIC have chosen to concentrate on predicting annual changes in stream water chemistry and vastly simplify process description occurring in the catchment. Cutting out details is one of the necessary arts of modelling and computer programming. However lumping all the nutrient cycling processes of translocation, foliar and root exudation, litterfall, litter decomposition and mineralisation and gross uptake into this net uptake time curve either involves a formidable modelling task in itself or a great deal of assumption making. Therefore due to the exclusion of most descriptions of soil processes affecting nitrogen e.g. nitrification and the vast oversimplification of nutrient cycling processes, it became clear early on that MAGIC should not be selected to research nutrient cycling hypotheses and process formulation in the future.

Sixteen models participated in the comparison workshop (Tiktak and van Grinsven, 1995). However only four of these were interesting for our group (Roger Røren working with plant physiology models, Trine Sogn working with soil-water models and myself) as the others were either regional models, plant physiology models, forest succession models or purely C and N models. These four models were NUCSAM (Groenenberg et al., 1995), SOILVEG (Van Heerden et al., 1995), NAP (Van Oene and Ågren, 1995b) and ForM-S (Oja et al., 1995).

NUCSAM, simulating the major hydrological and biogeochemical processes in the forest canopy, litter layer and mineral soil was planned linked to the very detailed forest growth model FORGRO (Mohren and ven de Veen, 1995). Linked, these two models would be the most detailed complex process-oriented forest nutrient cycling model yet produced. However it became clear that both these models and SOILVEG, were still under development in the Netherlands and no documentation had yet been produced for them. Therefore due to the time and distance factors, these models could not be selected.

The NAP model was in fact selected by Roger Røren for further testing in Norway. However its main goal is predicting forest growth over a long time span and both hydrologic and soil processes are simplified to a minimum. Therefore it was a less interesting model for both Sogn and myself.

ForM-S stood for the forest modelling series and included four different modules, water fluxes (1), soil temperature (2), ion fluxes (3) and primary production and nutrient cycling in upland stands (4). It was the only model programmed in the easier to programme "STELLA" language, which allows a transparent display of model assumptions and modal structure. Although the model was not yet freely available, documentation for some of the modules was available. One major disadvantage was the operating system - Macintosh.

We therefore arrived home from the workshop enriched in our knowledge of the different models, but still uncertain which model to choose. Despite our desire to choose NUCSAM or ForM-S, the former was still under development and the latter would force investment in an otherwise unnecessary computer. In contrast we already had the running version of NuCM, we had documentation (the original NuCM manual; Munson et al, 1992) and it was possible to view the programme files. NuCM has evolved from the ILWAS model (Goldstein et al., 1984; Gherini et al., 1985) and in comparison has simplified hydrological processes, more complex canopy interactions and similar soil chemical processes. ILWAS, although a very advanced model for its time (Johnson, 1990) has been applied very little by other users due to a poor screen interface with the user. In the development of NuCM, a user friendly interface was stressed and achieved. The processes included in NuCM compared favourably with the processes included in NUCSAM and SOILVEG. Therefore despite the small group of users world-wide of NuCM, the decision was taken to use NuCM in further research.

For research purposes it is important to have process-oriented models, which by the very nature of research - new ideas, understanding, hypotheses, tend towards the complex. For management purposes and regional applications simple process-oriented or even empirical models are preferred as this reduces input demand and makes the model more accessible for new users. The very process of transferring qualitative hypotheses about the forest ecosystem, to quantitative hypotheses, to a string of related mathematical expressions to be programmed in a computer language will also tend towards simplifying the complex.

I feel very strongly that the assumptions lying behind the more simple models, e.g. MAGIC, PROFILE, which are used to predict critical loads and hence affect the S and N reductions agreed by governments should be tested. This is what I am attempting to do in this thesis. These assumptions often include the over-simplification of nutrient cycling into net uptake v. time curves (MAGIC) or constant nutrient cycling (SAFE- Jönsson et al., 1995). In the Netherlands, the **RE**gional **S**oil **A**cidification **M**odel RESAM (De Vries et al., 1995) was developed precisely to include nutrient cycling aspects often left out of the more simple process-oriented models. NUCSAM (Groenenberg et al., 1995) was later derived from RESAM to apply at forest stand level and include inter annual variability.

Objectives of thesis

There have been two main goals for this thesis.

1. To study nutrient cycling processes in Norway spruce forests with the emphasis on the effects on soil chemical properties, soil solution chemistry and streamwater chemistry. To investigate the effects of different aged stands on nutrient cycling. To construct nutrient budgets of the base cations and nitrogen at both the Birkenes and Nordmoen site.
2. To select, document, calibrate, test and improve nutrient cycling models for use in Norwegian forests. In this task, it has also been important to study the past and existing methods of data collection in relation to the requirements of NuCM (the selected model) for data input and suggest improvements. It has also been important to compare NuCM with MAGIC, which until now has been the most commonly used model to predict critical loads in forest ecosystems in Norway.

List of papers

1. Kvindesland, S., Jørgensen, P., Frogner, T. and Aamlid, D, 1994. Hydrogeochemical Processes in a Forested Watershed in Southern Norway. Aktuelt fra Skogforsk, 10/94.
2. Kvindesland, S. and Røsberg, I, (Submitted). A Comparison of Nutrient Cycling in 30 and 40 year-old Norway spruce plots at Nordmoen, south-east Norway. Submitted, Water, Air and Soil Pollution.
3. Kvindesland, S. (Submitted). Calibration of the Nutrient Cycling Model (NuCM) at Nordmoen, a Norway spruce site in south-east Norway. Submitted, Ecological Modelling.
4. Kvindesland, S. (Submitted). Testing the Nutrient Cycling Model (NuCM) at Nordmoen, a Norway spruce site in south-east Norway. Submitted, Ecological Modelling.

Results and Discussion

Paper 1. Hydrogeochemical Processes in a Forested Watershed in Southern Norway.

Since the 1970s, Birkenes has been a centre for research on the effects of acid precipitation on stream water chemistry and between 50-100 scientific articles based on its data have been written until 1992. In 1986, the Norwegian Monitoring programme for forest damage (OPS) established an intensive forest plot in the Birkenes catchment. Although soil, soil solution chemistry and hydrological flow paths have been much in focus to explain the resulting streamwater chemistry, forestry data has been integrated to a minimal degree. This work aims to rectify this by quantifying and investigating forest processes that affect the catchment budget. Here we quantify the most important processes such as weathering, tree uptake and loss/gain of base cations from soil exchange sites in more detail than in previous work, thus allowing the calculation of element budgets. We focus in particular on (1) Output - Input budget for a 10-year period for the main ions, including dry deposition and seasalts; (2) Base cation budget, for the whole catchment and for a till site, including uncertainties; (3) Nitrogen budget. Two sub-objectives associated with the base cation budget were (1) to improve the original surface material map of the catchment (Lundquist, 1976) and (2) to find time trends in the chemical characteristics of podzols from 1974-1992.

A steady decline in Ca (2.3 %) and Mg (2.1 %) concentrations in streamwater has been observed at Birkenes for the period 1972-1987 (Christophersen et al., 1990b). This was interpreted as due to decreasing base saturation in the catchment's soil, assuming that all input and output fluxes remained constant. This finding will be discussed in light of results from the base cation budget and soil studies.

The Birkenes catchment has already been described (see Introduction, The Sites). Precipitation was collected daily by the Norwegian Institute for Air Research (NILU), while streamwater was collected weekly by the Norwegian Institute for Water Research (NIVA). Precipitation, throughfall and lysimeter water collected weekly by OPS at the intensive forest plot were stored and analysed according to the routine methods used at NISK (Ogner et al., 1991). Dry deposition was estimated from bulk deposition and throughfall measurements. Despite the availability of air concentrations of the major ions and estimations of their deposition rates (Statens forurensingstilsyn, 1997), it was both simpler, easier and we suspect more accurate to use our method at a windy site like Birkenes, where the filter effect of the tree canopies (Hasselrot and Grennfelt, 1987) is the most important factor.

A series of 13 soil profiles first dug in 1974, was resampled in Summer 1992. The resulting soil database allowed statistical analysis (Duncan's multiple range test) of the changes in chemical characteristics of the podzols with time, although too little data existed to do likewise with bog and thin humus soils. Mineral weathering was calculated for till, thin soil (thin humus and bare rock) and the whole catchment based on the Si flux in discharge. Using ratios of Si to base cations found in the various minerals present in Birkenes granite and assuming congruent dissolution, weathering rates for the base cations were calculated.

A forest survey was performed in spring 1993, where 85 circular plots (radius 8 m) spread regularly over the whole catchment were surveyed. All trees were measured for breast height diameter over 5 cm and every third tree for height. On dominant trees, age and width of the 10 last annual rings and the last 5 years height increment were measured. In addition soil samples were taken and the soil classified as podzol, bog, thin humus or bare rock. Using this data, a bold attempt was made to calculate rather than guess the net uptake of base cations and nitrogen by trees in the Birkenes catchment using the 10- year increment in biomass calculated from Marklund biomass functions (Marklund, 1988) and concentrations of these elements in the trunk, branches, needles and stump and roots. Net uptake on the different soil types was also calculated.

Throughfall data was analysed to show the effect of tree canopy processes on incoming total deposition. The filter effect of different sized birch, pine and spruce trees was clearly seen, with increasing SO_4^{2-} concentrations the taller and denser the tree canopy (Bjor et al, 1974). A large amount of inorganic N was consumed in the tree canopy and a smaller amount of organic N released. Direct uptake by the tree and/or epiphytes are the suspected cause.

The improved surface material map showed that Birkenes is covered by 9% bare rock, 23 % till, 13 % bog and 55 % thin humus soils. The chemical characteristics of podzols showed no clear trends in CEC, exchangeable base cations, exchangeable acidity or pH with time. Both Loss On Ignition (LOI) and total-N increased (not significantly) in most horizons between 1974-1992. The problems in relocating the exact soil profile, differences in weather conditions and time of the year could have hidden any small temporal changes in podzol chemical characteristics. The concentrations of exchangeable base cations in podzols appear to vary more in space than in time over this time span at Birkenes.

Mineral weathering calculations for the whole catchment suggest an annual release of 11 $mmol_c\ m^{-2}\ Na^+$, 12 $mmol_c\ m^{-2}\ K^+$ and 46 $mmol_c\ m^{-2}$ (Ca^{2+} + Mg^{2+}). Tills were estimated to release 65 $mmol_c\ m^{-2}$ (Ca^{2+} + Mg^{2+}) annually while thin soils were estimated to release 39 $mmol_c\ m^{-2}$.

Our studies on the catchment hydrology included an estimation of the annual storage and replacement frequency of 'drainable water' in the three main soil types, where 'drainable water' is the difference between maximum saturation and field capacity in each soil type. For an annual rainfall of 1100 mm, water stored in the catchment is estimated replaced 25 times in thin humus and 10 times in the till and top layer of the bogs, principally in the spring and autumn.

The median net nutrient uptake by trees for the catchment (in $mmol_cm^{-2}$ $year^{-1}$) was 74 for N, 36 for Ca^{2+}, 11 for Mg^{2+} and 18 for K^+. Possible errors in the method are listed which give a maximum uptake error between 27-36 % for Ca^{2+}, K^+ and N and 86 % for Mg^{2+}. Despite the possible errors, calculated net uptake fitted well with other comparable uptake studies.

Base cation budgets for the catchment and for the till areas were calculated both in the short-term (including annual net uptake for the entire tree biomass) and in the long-term (including annual net uptake for wood only). Uncertainties in each line of the budget; exchangeable base cation capital, weathering, net uptake by trees and the output-input budget were included. The data indicate an annual long-term catchment loss of soil exchangeable (Ca^{2+} + Mg^{2+}) varying between -2.5 to 0 %, a gain of K^+ between +2.7 to +4.8 % and a loss/gain of Na^+ between -2.6 to +1.8 %. The annual loss of exchangeable cations from the till was lower, -0.3 % (Ca^{2+} + Mg^{2+}) in the long-term budget.

The decline in Ca^{2+} and Mg^{2+} in streamwater observed between 1972-1989 was interpreted as due to decreasing base saturation in the catchment soil, assuming all input and output fluxes remained constant. This assumption is not true, as the input and output of SO_4^{2-} have been declining since 1980 (Statens forurensingstilsyn, 1996). Although this is clearly one reason for the registered decline in Ca^{2+} and Mg^{2+} in streamwater, the annual long-term base cation budget (1980-1990) suggests a slight loss of exchangeable (Ca^{2+} + Mg^{2+}) between 2.5 - 0 %. The smaller decrease in exchangeable (Ca^{2+} + Mg^{2+}), calculated for tills could not be observed in the time trend analysis of podzol soils. Although no historic data exists to prove this theory, it is hypothesised that thin humus soils could be the main soil site for loss of exchangeable Ca^{2+} and Mg^{2+} in the catchment.

The estimated annual total deposition of NH_4-N and NO_3-N was 26 kg ha^{-1}, which is higher than that estimated by NILU in 1990 viz. 20 kg ha^{-1} using air concentrations and deposition velocities for summer and winter (Statens forurensingstilsyn, 1991). The output-input budget showed that 24 kg N ha^{-1} was retained in the catchment each year, of which 45 % was attributed to net tree uptake, 9 % left the catchment in stream water as organic N, 3 % was estimated lost in net gas exchange and 43 % was unaccounted for and probably used to build up other living/dead organic matter.

The use of forest data at Birkenes allowed a new estimation of dry deposition, which suggests higher total N deposition than previously estimated. However 16 % of total N (allowing for exudation of organic N) is estimated to disappear in the tree canopies, either to tree/epiphyte uptake or lost to the atmosphere. The net tree uptake calculation for N indicated that 45 % of the 24 kg N $ha^{-1}year^{-1}$ retained in the catchment can be attributed to tree uptake. The base cation budgets and podzol studies indicated that the good forest soils on till deposits may not be leaching as many base cations as previously thought. Certainly after timber harvest, when all the branch and needle biomass is returned to the forest soil, the Ca^{2+} and Mg^{2+} losses for the whole catchment are small (0 - 2.5 %) and K^{+} is accumulating. However the short-term catchment base cation budget indicated a loss of soil exchangeable (Ca^{2+} + Mg^{2+}) between 0 to 5.4 %, which we hypothesise is mainly lost from the thin humus soils. Although trees situated in draining furrows in this soil may do well, loss of exchangeable cations will result in a decrease in the quality class of thin humus soils for forests. This might mean less economically as these sites are harder to manage and harvest.

Paper 2. A Comparison of Nutrient Cycling in 30 and 40 year-old Norway spruce plots at Nordmoen, south-east Norway.

The aim of this work was to find the similarities and differences in solute fluxes in precipitation, throughfall, and soil water between a 30 year-old (P30) and 40 year-old (P40) stand of Norway spruce at Nordmoen, south-east Norway. With additional data of dry deposition, litterfall, gross nutrient uptake of trees and estimates of weathering rates, nutrient cycling budgets for the 30 and 40 year-old stands were constructed for Ca, Mg, K and N. The P30 stand in this work was used in the modelling work with NuCM (Papers 3 and 4). Two sub-objectives which partly arose as a result of this modelling work were: (1) The qualitative study and discussion around the spatial variability of Cl^- in soil water, as this affects both estimation of soil water fluxes in this paper and calibration of the hydrological sub-model of NuCM. (2) The testing of several published estimates of weathering rates against field observations, as weathering rates are such an important input in models like MAGIC (Cosby et al., 1985a,b) and NuCM.

A general description of the Nordmoen site has already been given. The two square plots P30 and P40 (area 1000 m^2), which lie practically adjacent, have similar former land use and soil chemical properties. They were established in 1986 as part of the Integrated Forest Study (IFS) programme (Johnson and Lindberg, 1992). Bulk precipitation, throughfall, litterfall and soil water under four different horizons (O, E, Bs and BC) were monitored over a four year period (October 1986- September 1990). In the growing season, chemical analysis of these samples was monthly, while in the dormant season, analysis was performed when sampling would permit. The water, soil and plant samples were stored, prepared and analysed at NISK as described in Ogner et al., (1991). The water fluxes were calculated on a four year basis (1986-1990) by assuming that Cl^- is conservative in the soil. The fluxes of dry deposition and Net Canopy Exchange (NCE) were estimated from precipitation and throughfall data, assuming Na^+ and SO_4^{2-} behave conservatively through the tree canopy. Gross nutrient uptake of trees was estimated from growth increments and biomass compartment contents (Johnson and Lindberg, 1992).

According to Miller (1995), nutrient cycling in the development of an even-aged stand is controlled by the development of the 'nutrient-demanding' crown and fine root biomass. Until canopy closure, the trees ensure maximum production of nutrient-rich leaves, twigs and fine roots to achieve early dominance of the light and soil resources. After canopy closure, growth is concentrated in stem wood and structural roots, while the leaf biomass is simply maintained. Superimposed on this cycle is the effect that the older the tree, the larger its surface area and this often increases the amount of total deposition onto a forest (Horntvedt and Joranger, 1974; Stevens et al., 1992; Beier et al., 1993). At Nordmoen, canopy closure occurs at between 25- 30 years. One of our objectives was to test Miller's theory by comparing nutrient cycling between the 30 and 40-year old stands.

The estimated annual water fluxes for P30 and P40 were very similar, with interception at 26 to 27 % and a similar flux of drainage water. As often interception and evapotranspiration are found to be larger under older and more widely-spaced trees (Kittridge, 1948), this is a surprising result. However the soil water fluxes could not be accurately determined from Cl^- concentrations in throughfall and in the different soil layers, even on a four year scale (Manderscheid et al., 1995) due to the high spatial variability of average Cl^- concentration in soil water. In the forest floor, the Cl^- concentration appeared to be influenced by the spatial variability of Cl^- in throughfall. In the mineral horizons, this spatial variability in Cl^- concentration and in water flow may be amplified by roots.

The solute fluxes at P30 and P40 plots shared the following similarities: Total deposition was dominated by seasalt, H^+, SO_4^{2-}, NO_3^- and NH_4^+. Both canopies retained approximately half of the annual inorganic N, while organic N was released. Uptake and exudation by trees and epiphytes could explain this but more research is urgently needed. The inorganic nitrogen largely disappeared after passage through the forest floor. Sulfate was the dominant anion from precipitation to the BC horizon, except in the forest floor and E horizon, where organic anions were most abundant. The dominant cation in precipitation and throughfall was H^+, but its flux diminished down the soil profile and Ca^{2+} became the dominant cation in forest floor.

The following differences were found in total deposition and solute fluxes between P30 and P40: Total deposition of non-marine components and Net Canopy Exchange (NCE) of all ions except Ca^{2+} and organic N were greater in P40 than P30. Litterfall fluxes of Mg^{2+}, K^+ and N were also greater in P40 than P30. This caused higher nutrient fluxes of most elements in throughfall and through the forest floor. From the E horizon and downwards Al^{3+} became the dominant cation in soil solution in P40, while Ca^{2+} was the dominant cation in P30.

Estimated weathering fluxes from a 6- year soil column study (Teveldal and Jørgensen, 1996) and modelled by MAGIC (Wright et al., 1991) were compared to the soil water fluxes in the B horizons and input-output budgets in the base cation cycles. The K^+ and Mg^{2+} weathering rates of both methods fitted well with the field data. For Ca^{2+}, the low MAGIC rate (8 $mol_c ha^{-1} year^{-1}$) leads to an annual loss of 6% exchangeable Ca, while the high rate estimated from soil column experiments (360 $mol_c ha^{-1} year^{-1}$) leads to balance in the input-output budget. However the high rate from the soil column experiment could have been overestimated due to the problem of separating Ca^{2+} generated by mineralisation of organic matter, from Ca^{2+} generated by mineral weathering.

Using the weathering rates estimated by the soil column experiments, base cation cycles for both P30 and P40 plots showed that for Ca, litterfall was the most important input to the soil while for Mg, weathering was the largest input to the soil. Cycling of K was tight, with throughfall as the dominant input to the soil and the weathering rate for K was estimated very low. The general trends for both P30 and P40 are a Ca and Mg cycle showing balance for the input-output budgets for soil, while the K cycle shows a large annual loss of exchangeable K^+. Looking at the finer differences between P30 and P40, the Ca and Mg cycle show that P40 is slightly gaining exchangeable Ca^{2+} and is at balance for exchangeable Mg^{2+}, while P30 shows a slight annual loss of both exchangeable pools. In contrast the K cycle shows the loss of exchangeable K^+ is greater in P40. In the N cycle, P30 can cover annual N uptake from the flux of inorganic N in throughfall and mineralisation of annual litterfall, leaving a small amount of N to be immobilised in the soil. In P40, the throughfall flux and mineralisation of all annual litterfall would still leave a deficit of 105 mol $ha^{-1}year^{-1}$, which could be supplied by mining N from the large pools of existing soil N (Emmett et al., 1997).

The higher total deposition in P40 compared to P30 is assumed due to the stand's 5m extra height and greater surface and confirms the trend found by others (Horntvedt and Joranger, 1974; Stevens et al., 1992; Beier et al., 1993). The greater NCE and litterfall flux of most elements, except Ca^{2+} on P40 compared to P30 fits with Miller's hypothesis that biomass cycling fluxes like throughfall and litterfall will be greater in an older stand than a stand only just after canopy closure. In addition the slight difference between a gain of exchangeable cations in P40 and a loss of exchangeable cations in P30 in both the Ca and Mg cycle fits Miller's hypothesis that following canopy closure there can be a recharge of soil exchangeable nutrients provided the soil mineralogy is adequate (Miller, 1986). However the K cycle, with a greater annual loss of exchangeable K^+ in P40 than P30 does not fit this hypothesis, although the low weathering rate of K^+ could explain this.

Paper 3. and Paper 4. Calibration and Testing of the Nutrient Cycling Model (NuCM) at Nordmoen, a Norway spruce site in south-east Norway.

Acid deposition constitutes a major environmental problem over large areas of Europe and N. America. Anthropogenic emissions of sulfur and nitrogen compounds lead to accelerated acidification of soil, surface water and groundwater (Zak et al., 1997). Model predictions provide a fundamental basis for emission abatement policy decisions in Europe. In Norway, the models MAGIC (Cosby et al., 1985a,b) and PROFILE (Sverdrup and Warfvinge, 1993) have been most commonly used to predict the future effects of S and N deposition on forest ecosystems (Wright et al., 1991; Frogner et al., 1992). Although these models include many important soil processes, the forest's involvement is simplified to net tree uptake. Nutrient cycling processes such as leaf exudation, litterfall and mineralisation are not included.

The Nutrient Cycling Model (NuCM) is a process oriented model, which simulates the effects of atmospheric deposition on forest nutrient cycling. NuCM (Munson et al., 1992) was developed as part of the Integrated Forest Study (IFS) (Johnson and Lindberg, 1992) and has been used to simulate S deposition scenarios (Liu et al., 1991; Johnson et al., 1993), base cation leaching (Johnson, 1995) and the effects of harvesting and tree species change on nutrient cycling (Johnson et al., 1995). NuCM was also tested on an indoor bucket lysimeter experiment with small Scots pine trees receiving solutions with different N levels (Sogn et al, 1995b ; Sogn and Abrahamsen, 1997). However no thorough documentation of the calibration and testing of the NuCM model at the forest stand level under ambient conditions yet exists.

The main objectives of the two papers and appendix were:

1) To calibrate the NuCM model using one year of data for a Norway spruce stand at Nordmoen, where a comprehensive list of the used parameters and how they were obtained is presented and an initial evaluation of the strengths and weaknesses of process description is given.
2) To test the calibrated NuCM model on three new years of data at the same site, where throughfall, soil solution and nutrient cycling fluxes are compared to observed and independently estimated values. NuCM was also used to predict the effect of a 40 and 90% reduction in S deposition for a 48-year period, so the results could be compared with existing MAGIC predictions.
3) To improve documentation of the NuCM model (User manual - Appendix 1)

NuCM is a dynamic process oriented model describing the atmospheric, hydrologic, vegetation and soil processes which alter the chemical composition of water as it moves through the tree canopy and down the soil profile. NuCM simulates the concentration of 14 solutes; Ca^{2+}, Mg^{2+}, K^+, Na^+, Ala^{3+} (total monomeric Al), Alo (organic monomeric Al), NH_4^+, NO_3^-, SO_4^{2-}, Cl^-, DOC, PO_4^-, SiO_2 and DIC and the two variables pH and ANC. These components are tracked through the following pools: canopy, bole and roots, leaf litter, snowpack, soil solution, soil minerals, soil exchange surfaces and soil organic matter. Many processes are included to simulate the fluxes in and out of these pools. Full details of these processes can be found in Munson et al., (1992) and Kvindesland (1997). Tree growth is modelled more empirically as a function of a user-defined development stage and the availability of nutrients and water.

The four year dataset (Oct. 1986 - Sept. 1990) from the 30-year old Norway spruce stand (plot P30) at Nordmoen, south-east Norway was used for modelling. The site, measurements and methods are fully described in Kvindesland and Røsberg, (in review). The second year (1 Oct. 1987 - 30 Sept. 1988) of this period was used for calibration to avoid lysimeter installation effects, while the remaining three years were used to test the calibrated model.

NuCM separates the hydrology sub-model, which has a daily timestep and must always be run first, from the biogeochemical sub-model, which I ran at a monthly timestep. A meteorology file contains daily precipitation , temperature , relative humidity, wind speed and cloudiness and a deposition file (AIR file) contains precipitation and ambient air concentrations. These files are required to run the hydrology sub-model, which produces a result file (H2O file) needed to run the biogeochemistry sub-model. To circumvent problems associated with modelling N immobilisation in the canopy, a special construction of the AIR and H2O files was used to model throughfall concentrations. This means that only the base cations and pH are simulated in throughfall (i.e. using wet and dry deposition and canopy processes). During the calibration of NuCM's hydrology, it became clear that although NuCM simulated reasonable soil water percolation during the snow-free months, poor simulation of percolation occurred in the winter months due to poor snow simulation and problems simulating a frost layer. For this reason the hydrology results from the SOIL model (Jansson, 1991) were used instead. NuCM demands an extensive parameter list, which is described comprehensively in the calibration paper. These 94 parameters according to my evaluation can be described as measurable (44), estimable from measurements (17) and calibration/guess parameters (33). To statistically evaluate the calibration and test simulations, the normalised mean absolute error (NMAE) was used and time plots of simulated v. observed were plotted. To compare against existing MAGIC predictions for Nordmoen (Wright et al., 1991), the calibrated model was run for 48 years (1986 - 1934), where the existing four years of hydrology and deposition data were repeated twelve times. Three scenarios were simulated: a 40 % reduction in total S deposition and a 25 % reduction in nutrient uptake (40/25 sc.), a 90 % reduction in total S deposition and a 25 % reduction in nutrient uptake

(90/25 sc.) and a 90% reduction in total S deposition and a 100 % reduction in nutrient uptake (90/100 sc.).

The calibrated model (calibration paper) was meant to be tested directly on the other dataset, however a minor recalibration was necessary prior to testing. Root data , the coarse litter and humus decay rates and the uptake distribution of NH_4 between horizons had to be altered. Ideally many years should be used for calibration (which normally are not available). Although solute concentrations in throughfall and soil solution and soil exchangeable pools were checked in a 10 year run during the calibration, cycle fluxes were not. Therefore if one is using 1-2 years for calibration, it is recommended to run NuCM 5-10 years ahead and check in particular the cycle fluxes.

The dry summers of 1989 and 1990 caused SOIL to simulate low moisture contents and water fluxes in the lower B horizons, causing high simulated concentrations of NO_3^- and Cl^-, which in turn led to high simulated concentrations of Ca^{2+}, Mg^{2+}, and Ala in these horizons during this period.

Although Ca^{2+}, Mg^{2+} and K^+ concentrations in throughfall could be calibrated adequately, the test showed that Ca^{2+} could not be simulated adequately on a monthly basis as the exudation pattern appears to vary widely between years. Bulk deposition and throughfall are the two canopy variables normally measured at Norwegian forest sites, so the detailed canopy interactions of NuCM, including wet and dry deposition, foliar nitrification, exudation and leaching processes, are impossible to parameterise at the present data level. Yet this detailed description of canopy processes does not include the sorption/uptake of inorganic N, nor the production of organic N which was observed at Nordmoen (Kvindesland and Røsberg, in review). NuCM balances the exudation of positive base cations by exuding OH^- ions, which causes substantial and unrealistic increases in simulated pH in throughfall. The production of organic anions and uptake of NO_3^- could instead balance the exudation of cations, but before any process-oriented description can be incorporated in NuCM, research is needed to identify and quantify N consumption and release by these canopy sinks. Despite trying to circumvent problems associated with modelling N immobilisation in the canopy by using observed fluxes of throughfall as far as possible, the concentrations of NO_3^- and NH_4^+ simulated in throughfall during the winter were very high compared to the observed concentrations (one collected throughfall sample for each winter period). However the practical problems of collecting snow throughfall in winter, do add uncertainty to these observations.

In soil solution, the concentrations of SO_4^{2-} and Ala (total monomeric Al) were simulated successfully in all horizons, while pH and the concentrations of Ca^{2+}, Mg^{2+} and Na^+ were simulated reasonably in the O and E horizons. The concentration of NO_3^- in soil solution was simulated poorly in the dormant season due to probably overestimated concentrations of NO_3^- simulated in throughfall and no description of immobilisation of N by soil microbes. In addition, the nitrification description is inflexible. Organic and inorganic monomeric Al (Alo and Ali respectively) were simulated poorly due to limiting the definition of organic Al in NuCM to the complex, $Al^{3+}R^{3-}$, when in reality Al^{3+} may also complex with partly dissociated organic acids. Although DOC was calibrated successfully in the O and E horizons (calibration paper), the recalibration of decay rates caused the average concentration of DOC to be simulated too low, although the dynamics were still simulated well (test paper). In both the calibration and testing, DOC was simulated poorly in the B horizons. This low simulation of DOC can be explained by another source of DOC not described in NuCM, e.g. root exudation and decay of other organic matter and/or poor process description of DOC production from litter.

The nutrient cycling of the cations, NH_4^+ and Mg^{2+} were simulated successfully. The nutrient cycling of Ca^{2+}, especially exudation, uptake and litterfall was simulated poorly due to the lack of a process describing the accumulation of Ca^{2+} in needles as they age. It is vital to describe at least simple accumulation and translocation in needles, where the net difference between element content in current and newly fallen needles is included, so both uptake and litterfall can be simulated better in NuCM.

Future predictions by NuCM were compared to the predictions by MAGIC under different scenarios. For the 40/25sc., MAGIC and NuCM predicted very different soil exchangeable pools of Ca^{2+} and K^+ in 2034. While NuCM predicted 590 and 280 $mol_c\ m^{-2}$ of Ca^{2+} and K^+ respectively, MAGIC predicted 0 in both pools. While MAGIC, with continued tree uptake, predicted a critical load of $< 0\ mmol_c m^{-2} year^{-1}\ SO_4$ to ensure a molar Ca/Al ratio > 1 in the top 60 cm of soil, NuCM in the 0/0sc. (no change scenario) predicted a Ca/Al ratio of 1.2 and 0.9 in the O and Bs layer respectively in 2034. The reasons for the large difference in predictions lies mainly in simulation of the important fluxes, uptake and weathering. The K^+ net uptake is underestimated in NuCM because K^+ in litterfall is overestimated due to no description of translocation. While MAGIC simulates 1, NuCM simulates 32 for the Ca^{2+} weathering rate (in $mmol_c\ m^{-2}\ year^{-1}$). Although the real Ca^{2+} weathering rate is still unknown, the MAGIC rate appears far too low as soil sampled 10 years later in 1996 suggested no decrease in exchangeable Ca in the forest floor compared to the 1986 measurements. Therefore predictions for critical loads and exchangeable pools in 2034 are likely to lie nearer NuCM's predictions than MAGIC's predictions. However one must conclude that neither MAGIC nor NuCM could produce good weathering estimates on their own and independent field estimates of weathering remain important for realistic future predictions. To compare how the inclusion of nutrient cycling processes affects predictions, NuCM and MAGIC should be compared in the future with similar uptake and weathering fluxes.

Conclusions

When including forestry data in the catchment studies at Birkenes, it can be concluded that:

- The long-term base cation budget for the catchment (with annual net uptake for wood) implies a net annual loss of exchangeable soil (Ca^{2+} +Mg^{2+}) of 0 to 2.5 % and a gain in K^+ of 2.7 to 4.8 %.
- The long-term base cation budget for till soils showed an average annual loss of exchangeable (Ca^{2+} +Mg^{2+}) of 0.3 %.
- In a statistical analysis of the chemical characteristics of podzols no evidence of decreasing exchangeable Ca^{2+} and Mg^{2+} between 1974 -1992 was found, suggesting that podzols are not losing as many base cations as previously thought.
- The annual total N deposition was estimated at 26 kg $ha^{-1}year^{-1}$, while only 2 kg ha^{-1} $year^{-1}$ occurs as inorganic N (mostly NO_3^-) runoff.
- Of the N consumed in the catchment, 45 % was attributed to net tree uptake, 9 % left the catchment in stream water as organic N, 3 % was estimated lost in net gas exchange and 43 % was unaccounted for and probably used to build up living/dead organic matter.

Nutrient cycling was compared at a 30-year old (P30) and 40-year old (P40) Norway spruce stand at Nordmoen. In both plots:

- Soil water fluxes could not be accurately determined from Cl^- concentrations in throughfall and soil layers over a 4-year period due to the high spatial variability of average Cl^- concentration in soil water.
- The canopies retained approximately half of the annual inorganic N, while organic N was released.
- The remainder of inorganic N input largely disappeared in the forest floor.

The following differences in nutrient fluxes were observed between P30 and P40.

- Total deposition of non-marine components and Net Canopy Exchange (NCE) of all ions except Ca^{2+} and organic N were greater in P40 than P30. The higher total deposition in P40 was assumed due to the stand's 5m extra height and greater surface area.
- Litterfall fluxes of Mg^{2+}, K^+ and N were also greater in P40 than P30, causing higher nutrient fluxes of most elements in throughfall and through the forest floor.
- From the E horizon and downwards Al^{3+} became the dominant cation in soil solution in P40, while Ca^{2+} was the dominant cation in P30.

- The greater NCE and litterfall flux of most elements, except Ca^{2+} in P40 compared to P30 fits with Miller's hypothesis that biomass cycling fluxes like throughfall and litterfall will be greater in an older stand than in a stand just after canopy closure.

Base cation budgets were constructed, using weathering rates estimated in soil column experiments.

- For Ca, litterfall was the most important input to the soil while for Mg, weathering was the largest input to the soil.
- Cycling of K was tight, with throughfall as the dominant input to the soil. Weathering rates of K were estimated very low.

Despite the availability of an existing user manual, it was necessary to improve documentation of the hydrological and biogeochemical processes described in the Nutrient Cycling Model (NuCM) (Appendix 1). Calibration (using one year of data) and testing (using three years of data) of NuCM was performed on the 4-year dataset from the 30-year old Norway spruce stand (P30) at Nordmoen.

In the hydrology simulation:

- The hydrology sub-model in NuCM was replaced with the SOIL hydrology simulation, due to problems with snow simulation in NuCM, leading to poor winter hydrology simulation.
- SOIL simulated adequate soil hydrology, except during very dry summers, when low moisture contents and water fluxes in the B horizons, caused too high simulated concentrations of NO_3^- and Cl^-, in turn causing too high simulated concentrations of Ca^{2+}, Mg^{2+} and Ala (total monomeric Al).

In the throughfall simulation:

- the concentrations of K^+ and Mg^{2+} were simulated successfully, but Ca^{2+} and pH were simulated poorly.
- NuCM describes many canopy processes, which are difficult to calibrate when only bulk deposition and throughfall are routinely measured at Norwegian forest sites.
- However the large disappearance of inorganic N and production of organic N observed at Nordmoen was not described in NuCM.

In the soil solution simulation:

- NO_3^- concentration was simulated poorly, especially during the winter due to the high simulated throughfall concentrations of NO_3^- and the absence of a description of immobilisation of N by soil microbes. Inflexibility of the nitrification description led to general poor simulation of NO_3^- in soil solution.

- The concentrations of SO_4^{2-} and Ala were simulated successfully in all horizons, while pH and the concentrations of Ca^{2+}, Mg^{2+} and Na^+ were simulated reasonably in the O and E horizons.
- Organic and inorganic monomeric Al (Alo and Ali respectively) were poorly simulated due to limiting the definition of organic Al to the complex, $Al^{3+}R^{3-}$, when in reality Al^{3+} may also complex with partly dissociated organic acids.
- The average concentration of DOC was simulated too low, although the dynamics were simulated well. This low simulation of DOC can be explained by another source of DOC not described in NuCM, e.g. root exudation and decay of other organic matter and/or poor process description of DOC production from litter.

In the nutrient cycling simulation:

- The cycling fluxes of NH_4^+ and Mg^{2+} were simulated successfully.
- The nutrient cycling of Ca^{2+}, especially exudation, uptake and litterfall was simulated poorly due to the lack of a process describing the accumulation of Ca^{2+} in needles as they age.
- It is necessary to describe simple accumulation and translocation in needles, where the net difference between element content in current and newly fallen needles is included, so both uptake and litterfall can be simulated better in NuCM.

The comparison of NuCM with MAGIC predictions under a scenario with 40 % reduction in S deposition and 25 % reduction in net tree uptake, showed very different results for the soil exchangeable pools and critical loads in 2034.

- NuCM predicted 590 and 280 $mol_c\ m^{-2}$ of Ca^{2+} and K^+ respectively, while MAGIC predicted 0 in both pools.
- MAGIC predicted a critical load of < 0 $mmol_c m^{-2} year^{-1}$ SO_4 to ensure a molar Ca/Al ratio > 1, while NuCM in the no change scenario (i.e. 68 $mmol_c m^{-2} year^{-1}$ SO_4) predicted a Ca/Al ratio of 1.2 and 0.9 in the O and Bs layer respectively.

The very different predictions are mainly caused by differences in simulation of the uptake and weathering fluxes.

- The K^+ net uptake is underestimated in NuCM because K^+ in litterfall is overestimated due to no description of translocation.
- While MAGIC simulates 1, NuCM simulates 32 for the Ca^{2+} weathering rate (in $mmol_c\ m^{-2}\ year^{-1}$). Although the real Ca^{2+} weathering rate is still unknown, the MAGIC rate appears far too low as soil sampled 10 years later in 1996 suggested no decrease in exchangeable Ca in the forest floor compared to the 1986 measurements.

Probably predictions for critical loads and exchangeable pools in 2034 are nearer NuCM's than MAGIC's predictions. However one must conclude that neither MAGIC nor NuCM could produce good weathering estimates on their own and independent field estimates of weathering remain important for realistic future predictions. To study how the inclusion of nutrient cycling processes affects predictions, NuCM and MAGIC should be compared in the future with similar uptake and weathering fluxes.

Concluding remarks

Too much of my time on this thesis was spent trying to understand the programme coding of NuCM. Being untrained in programming was a handicap, but so was the poor documentation of programme code, which sadly is rife among scientific models (Tiktak and van Grinsven, 1995). I therefore applaud the attempts by other scientists to make the model design and internal workings more transparent to non-developers. Here the call for a common model documentation etiquette (Benz and Knorrenschild, 1997) and encouragement of the use of object-oriented programming to create modular and generic ecosystem models (Acock and Reynolds, 1997) are important for a smooth transfer of knowledge.

Modellers are always confronted with a choice between complexity and functionality in model design. A complex process-oriented model can be created, which contains the latest findings of research, but is only possible to parameterise from laboratory data or at an intensive research site. By simplifying processes, a more functional management model can be produced, which demands less data and is therefore more portable. In Norway, the models MAGIC (Cosby et al., 1985a,b) and PROFILE (Sverdrup and Warfvinge, 1993) have been most commonly used to predict the future effects of S and N deposition on forest ecosystems (Wright et al., 1991; Frogner et al., 1992.). These models include important soil processes but simplify nutrient cycling processes to a net uptake versus time graph (MAGIC). The models have been run on over 100 different plots and a critical load map for the whole of Norway created, even if input data is quite roughly estimated on some plots (Solberg et al., in prep.). Clearly it would be impossible to perform the same task with NuCM, yet NuCM simulates nutrient cycling processes and keeps track of important pools and fluxes that both MAGIC and PROFILE ignore.

Although NuCM is a more detailed model, many areas of newer research e.g rhizosphere and mycorrhiza processes (Courchesne and Gobran, 1997; Jongmans et al., 1997; Simard et al., 1997; Clothier and Green, 1997), which could prove of immense importance in understanding forest nutrient cycling, are not included. This emphasises the need to have research models to test out the effects and importance of new findings on future predictions of for example critical loads.

Van Oene and Ågren (1995a) investigated models describing the effects of S and N deposition on forest, where the effects of differences in complexity in models originating from soil science and tree physiology were studied. They concluded that increased complexity is only warranted in a model when it entails strong feed-back mechanisms. NuCM, although it uses a simple predefined growth rate, allows nutrient limitation in the soil to increase or decrease this growth rate during the simulation . This feed-back mechanism is a clear improvement to the fixed and unchangeable net nutrient uptake rate used in MAGIC.

NuCM, although successfully simulating several important parameters, clearly needs improvement in process description. Canopy process description can be simplified to match existing data collection, while more research is needed before a process description of the N dynamics in the canopy can be incorporated in NuCM. To improve simulation of NO_3^- in soil solution, increased user's control over the nitrification parameters and inclusion of the immobilisation of N by soil microbes is required. Redefining Alo (organic monomeric Al) to also include complexes with partly dissociated organic acids will improve both Alo and Ali (inorganic monomeric Al) simulation in soil solution. Too low DOC simulation indicates either a missing DOC source or that the decomposition process should be reconsidered.

NuCM and MAGIC under future scenarios of reductions in S deposition and net tree uptake at Nordmoen gave very different predictions for soil exchangeable pools of Ca^{2+} and K^+ and critical loads. These differences were mainly due to the very different simulations of the important fluxes, K^+ uptake and Ca^{2+} weathering. Probably critical loads and exchangeable pools in 2034 are nearer NuCM's than MAGIC's predictions. However this is due to choosing to calibrate the weathering rates in NuCM towards estimates, which I believe are more realistic. Therefore one must conclude that neither MAGIC nor NuCM could produce good weathering estimates on their own and independent field estimates of weathering remain important for realistic future predictions. To study how the inclusion of nutrient cycling processes affects predictions, NuCM and MAGIC should be compared in the future with similar uptake and weathering fluxes.

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PAPER 1

HYDROGEOCHEMICAL PROCESSES IN A FORESTED WATERSHED IN SOUTHERN NORWAY

Sheila Kvindesland, Per Jørgensen, Tore Frogner and Dan Aamlid

Contents

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Introduction

Much research has been done in and written on the Birkenes catchment (Lundquist, 1976; Christophersen et al., 1982, 1990 a,b; Christophersen and Neal, 1990; Mulder et al., 1990, 1991). In this article, we want to integrate some of the previous research done with new measurements and quantify important processes both for the first time and in more detail than in previous work.

Fig. 1 shows a typical section of either of the 2 main valleys in the catchment.

Our aim is to compare the input of atmospheric deposition, which is part measured and part theoretically calculated, with the output stream solution measured at the tip of the catchment over a 10 year period. We then want to quantify some of the main processes shown in Fig. 1, such as tree uptake, weathering and loss from exchange surfaces. For this purpose we have gathered together as much existing atmospheric deposition, stream water, soil solution, soil and tree data as we could find in addition to measuring new data. It is particularly the base cations and nitrogen that interests us and these two catchment budgets are calculated at the end of the article. It is interesting, when we have such an extensive data set on base cations for a catchment of this size to find the uncertainty in the base cation budget calculation. The purpose was therefore to find if it is possible to quantify the loss of base cations from the exchange complexes through this approach.

Study area

The Birkenes watershed covers an area of 41.6 ha and consists of 2 main valleys. The lower valley with the main stream being Vestre Tveitdalen and the higher valley with the main bog and a slower flowing tributary stream being Langemyrdalen (see Fig. 2). Fig. 2 (commentated more fully in soil section) shows the distribution of the various soil types within the watershed. The areal coverage in % for the different types are:

Bare rock: 8.6 % Till: 23.2 % Bogs: 12.6 % Thin humus soils: 55.6 %.

Most of the flat area of Langemyrdalen is covered with peat soils, which were more extensive in the past, before forest drainage ditches were cut. The main stream in Vestre Tveitdalen has peat deposits along its course, which widen out into a small bog in its south west end. The catchment lies within the Precambrian Birkenes granite. The best forest soil is found on the till deposits where podzols and brown earths have developed. The 2 main areas of till deposits are situated on the lower south east slopes of the 2 valleys. The distribution reflects the glacial activities during the last glaciation. The stoss sides have a cover with sandy tills. Thicker hummocky till deposits are found in the central part of the valleys. On the higher north west slopes, one finds thin humus soils with bare rock dominating the summits and steepest slopes.

Materials and Methods

Water

The precipitation was collected daily by the Norwegian Institute for Air research (NILU). In-between each sampling, the container was rinsed thoroughly, thereby eliminating dry deposition build-up. The throughfall and precipitation was collected weekly by the Norwegian Monitoring Programme for Forest Damage (OPS) and dry deposition therefore had a chance to build up in the precipitation containers. The NILU precipitation data is therefore used in the input- output budget (Table 8). Lysimeter water from the forest monitoring station at Birkenes was collected weekly by the Norwegian Forest Research Institute (NISK) (Frogner, 1993). The precipitation and throughfall water (OPS), the lysimeter water and the water samples collected in Summer 1993 were stored and analysed according to the routine methods used at NISK (Ogner et al., 1991). Stream water was collected weekly by the Norwegian Institute for Water research (NIVA).

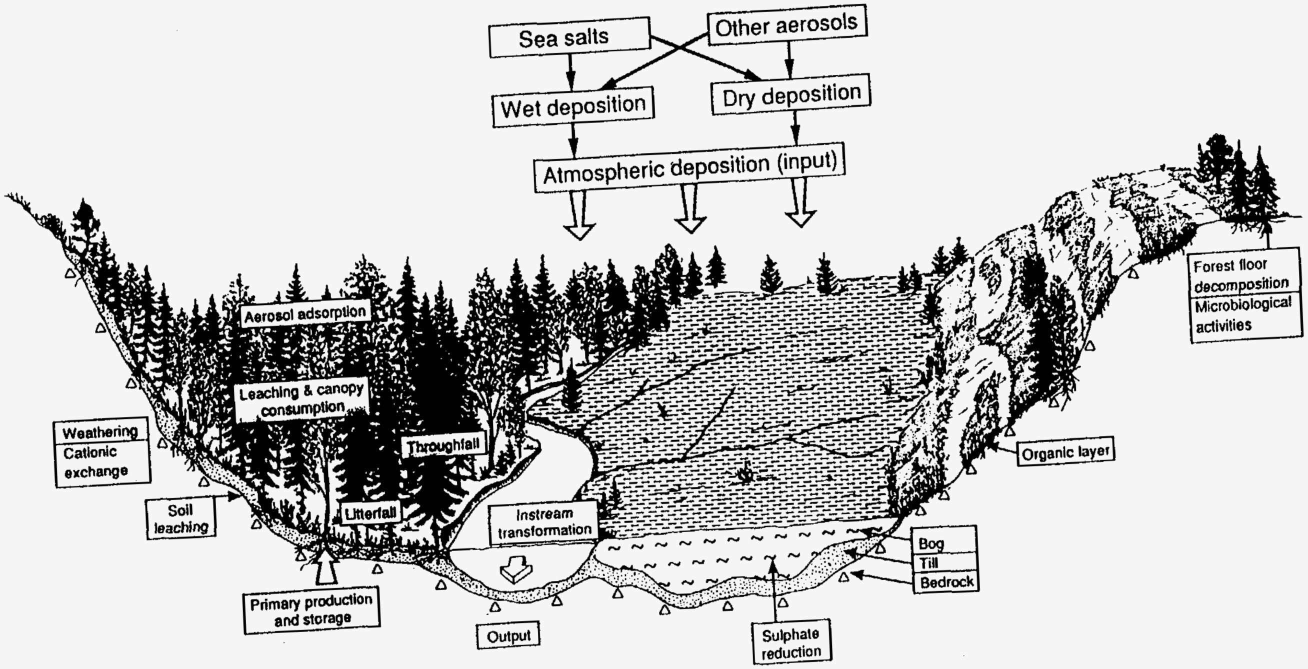

Fig. 1. The main hydrogeochemical and biological processes occurring in a typical valley section at Birkenes.

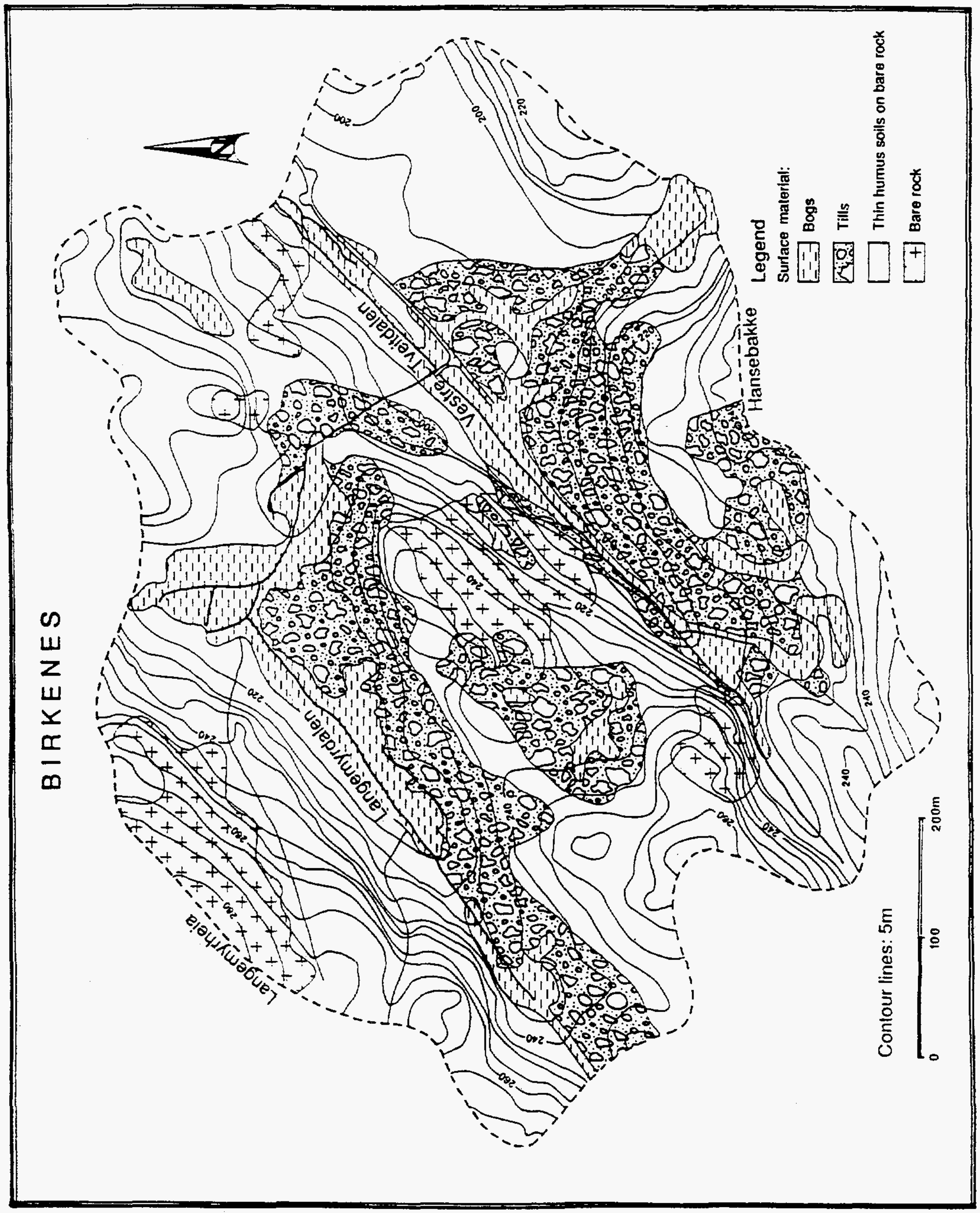

Fig. 2. A simplified map of surface materials in the Birkenes catchment.

Soil

The soil data came from a number of sources. Table 1 shows details of the data origin and the 3 laboratory methods used for cation extraction. All methods gave similar values for the actual concentrations of base cations. Buffered and unbuffered extractions naturally gave different values for base saturation (BS), cation exchange capacity (CEC) and exchangeable acidity. The methods used to find other chemical properties are described in the references marked with (M) in the table. The following definitions are used for CEC and BS.

$\mathrm{CEC} = \mathrm{Ca_{ads}} + \mathrm{Mg_{ads}} + \mathrm{K_{ads}} + \mathrm{Na_{ads}}$ + exchangeable acidity $(\mathrm{Al_{ads}} + \mathrm{H_{ads}})$

$\mathrm{BS} = (\mathrm{Ca_{ads}} + \mathrm{Mg_{ads}} + \mathrm{K_{ads}} + \mathrm{Na_{ads}})/\mathrm{CEC}$

Table 1. Details on the data origin and the extraction method used for cations

Point series	Years measured	Data Reference	Extraction method	Extraction Reference
1-13 1-13 62-71	1974 1992	Frank, 1980. (M) New data.	Buffered extraction (1M Ammonium acetate, pH 7) for base cations.	Page, Miller & Keeney, 1982.
14-17 72-75 72-75	1984 1984 1992	Statens forurensingstilsyn, 1985. Statens forurensingstilsyn, 1993.	Unbuffered extraction (1M Ammonium nitrate) for base cations.	Ogner et al., 1991.(M)
18 19-26 19-26 19-26	1986 1986 1987 1991	Overvåkningsprogram for skogskader, NISK.	Unbuffered extraction (1M Ammonium nitrate) for base cations.	Ogner et al., 1991.(M)
27-61	1990	Fjeldal, 1992. (M)	Unbuffered extraction (0.1M Barium chloride) for base cations.	Hendershot & Duquette, 1986.
	1993	New data (85 forest survey plots)	Unbuffered extraction (1M Ammonium nitrate) for base cations.	Ogner et al., 1991.(M)

Forest vegetation

A forest survey of Birkenes was carried out in late April/early May (Øyen, 1993). The 41.6 ha catchment was divided into a 50 by 100 m grid and at each of the 85 intersections, a circular area of 201 m^2 (radius 8 m) was surveyed. All trees with breast height diameter (D1.3) over 5 cm were measured for D1.3, and every third tree was measured for height (H). The dominant trees (by diameter) were bored at breast height. Age and width of the 10 last annual rings were measured. In addition the last 5 years height increment was measured in dm.

Biomass increments

In order to find the biomass in kg DM (Dry Matter) per tree, the biomass increment was calculated as the difference between 2 calculations 10 years apart. The Marklund biomass functions were used for the calculation (Marklund, 1988). Litterfall data is not used in this calculation as one is calculating the net accumulation of biomass within 10 years. Stem over bark (G2 and T2), living branches with needles (G12 and T14), needles (G16 and F18) and stump and root system (G23 and T25) biomasses were calculated for spruce and pine respectively using the named functions. The following birch functions, stem over bark (B2) and living branches (B11) were used to calculate biomass for all 4 deciduous species. All functions generally required D1.3 (cm) and H (m) as independent variables, giving the dependent variable ln [biomass (kg DM)] per tree, where ln denotes the natural logarithm.

A regression of D1.3 versus H for each tree species was used to estimate heights in those trees which were not measured. The 1983 values of D1.3 and H (by means of the 10 last years radius and the 5 last years annual shoot length) were only available for a few trees from each plot and only for spruce and pine. A regression of D1.3 1993 versus D1.3 1983 and H 1993 versus H 1983 for pine and spruce gave the following best fits.

Spruce, D1.3: $y = 1.04x - 4.49$, $r^2 = 0.98$, N = 80
Spruce, H: $y = 1.10x - 5.94$, $r^2 = 0.85$, N = 80
Pine, D1.3: $y = 0.98x - 1.35$, $r^2 = 0.99$, N = 14
Pine, H: $y = 0.97x - 1.18$, $r^2 = 0.96$, N = 14

where y = 1983 value of D1.3 or H, and x = 1993 value of D1.3 or H.

The pine regressions were also used to find D1.3 and H for deciduous trees, as it was assumed that pine, like the deciduous trees tended to dominate on lower quality sites.

As the Spruce H regression had a relatively large negative intercept, low heights in 1993 would give unrealistically low or negative heights in 1983 for small trees. A new regression of heights <10 m in 1993 versus heights in 1983 was forced through the origin and 1983 heights were calculated ($H83 = 0.42H93$). The difference between biomass in 1993 and 1983 divided by 10 gave the biomass increment (kg DM per tree per year). The sum of biomass increments for all trees on one plot area was then recalculated as a biomass increment in kg DM/ha.

Element uptake

The contents of the elements (N, Ca, Mg and K) in % of DM in stem over bark, branches and needles for spruce and pine were taken from a Swedish regression of biomass element content versus latitude (Rosen, pers. comm.). If the Swedish regression using Birkenes latitude gave unreasonable values, the element contents were adjusted towards those measured in spruce at Nordmoen (Johnson and Lindberg, 1992). Nutrient contents of current year, last year and 2 years old spruce needles were measured in Birkenes in 1986 (data from the Norwegian Forest Monitoring Programme, NISK). An average of these values was used to include retranslocation in the net uptake calculation. The element concentrations for spruce and pine stump and root system were taken from total below ground root concentrations of a 55 year Norway Spruce forest in Southern Sweden (Nihlgård, 1972). No element contents are available for the deciduous trees, so the pine concentrations are used. The tree volume distribution of 79% spruce and

21% pine and deciduous trees was used to calculate N, Ca, Mg and K uptake in kg/ha/year from the whole catchment.

Hydrology

Water pathways

Figure 3 illustrates the hypothesised distribution of precipitation among the soil types and the subsequent flow paths. The resulting stream discharge is obviously a complex mixture.

As seen from the topography of the area (Fig. 1) all the snow melting and rain falling on bare rock will rapidly flow into areas covered with thin humus soils and mix with the rain falling here. This mixture comprises 64 % of the input during a rainstorm. This water will either:

1. enter an area covered by bogs
2. enter an area covered by tills
3. run directly into a brook.

The precipitation falling directly on the tills will mix with water which has run through thin humus soils or over bare rock. This mixed water will then either:

4. enter an area covered by bogs.
5. run directly into a brook.

The rain falling directly on the bogs will mix with water of type 1 and 4 before it runs into the brook. Put simply, practically all water must travel through either a thin strip or large expanse of bog before entering the stream.

All researchers at Birkenes agree that water flow paths are of the utmost importance for the resulting stream water chemistry (Christophersen et al., 1982, 1990a).

End Member Mixing Analysis (EMMA) has been suggested as a new way of modelling stream water from a mixture of defined soil water end members (Christophersen and Neal, 1990; Christophersen et al., 1990a; Hooper et al., 1990). EMMA has been performed on stream and lysimeter data from Birkenes (Christophersen et al., 1990a; Easthouse et al., 1992). In the first reference, EMMA picked 3 soil end members; the O/H, the B/C and deep bogs "the valley bottom layers". Only 4 chemical species were modelled: Al_i (inorganic monomeric Al fraction), H^+, Si and Ca. Their concentration range is well defined within a horizon but differs between the selected soil end members. The plot for Al_i against H^+ shows how median concentrations from lysimeters in the valley bottom, B/C and O/H layers in the podzols/organic soils span the range of stream concentrations. At high discharge, Al_i and H^+ concentrations are high, which EMMA interprets as a large contribution from the O/H layers, while at low discharge the model interprets that the main contribution to the mixing comes from the deeper soils low in both ions. However similar plots of Ca versus H^+ and Si versus H^+ could not span the range of stream concentrations. Although Si was not expected to behave conservatively (precipitation and biological consumption), Ca was, and it was therefore concluded that the observed end member set was insufficient to explain stream water chemistry. The second EMMA reference used 3 similar soil end members, the podzol O, podzol B plus surface bog layers, and deep bog layers and 4 chemical species: DOC, Al_i, H^+ and Ca. A diagram showed the estimated end-member contribution to the stream during the 6 day autumn storm period studied. Surprizingly the contribution from the B horizon always dominated the peak stream discharge, even if the O horizon contribution became more significant during secondary storm events (35% contribution).

The combination of surface bog with B podzol profile may make sense chemically in the small area of the catchment intensively studied with lysimeters and soil samples (Mulder et al., 1991), but cannot make sense in the whole catchment. Firstly Figure 2 shows that the slopes north of Langemyrdalen are mainly thin humus and there is no or minimal contact between mineral B horizon and the surface bog. This must also be the case in other areas of the catchment. Secondly we suspect larger expanses of surface bog (Langemyrdalen) to be progressively less affected by B horizon flow as one approaches the bog's centre. The fact that so much water, around 60% of discharge, flows through the B horizon so quickly producing peak stream flow after one day seems unlikely from simple Darcian flow calculations (Wels et al., 1991). Water pathways have been investigated in a catchment with similar thin till cover and bedrock in Ontario, Canada (Renzetti et al., 1992). From trench and tensiometer studies during 6 autumn rain events, they concluded that subsurface flow (between the mineral/organic layer and the bedrock) was the largest contributor to stream discharge (42-85% during rain events with antecedent moist conditions). This could be the case in Birkenes too and what EMMA labels B horizon flow could well be mainly subsurface bedrock flow. Figure 4 shows possible water flow paths for the thin humus/bare rock to till interface on slopes.

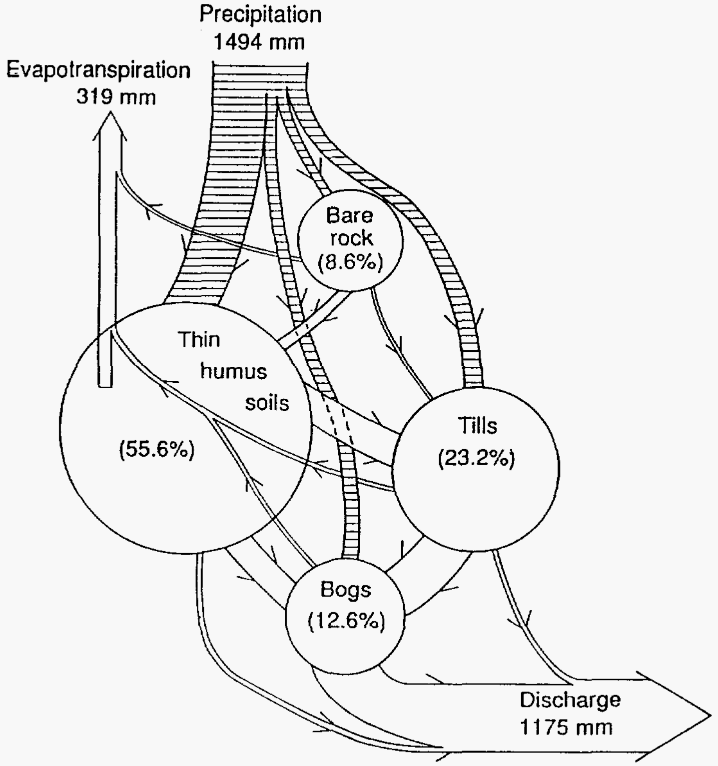

Fig. 3. The distribution of precipitation among the soil types and the major water fluxes at Birkenes.

Wet periods at Birkenes

During snow melt or periods with heavy precipitation, the areas covered with thin humus become saturated and excess water lead by a series of furrows and fissures moves down to the areas covered with bogs and till. In this dense rock, there are very few deep fissures, so only a very small amount of water will move down to the ground water level in the rocks. The furrows can be thought of as the macropores shown in fig. 4, allowing quick subsurface flow. If thin humus had direct contact with the stream, a peak stream discharge would occur. However instead the subsurface flow moves into the till or bogs. In the till, this subsurface flow can continue and a rising saturated layer can cause flow through the B horizon. At the same time flow in the B horizon can be caused by percolation through the O and E horizons or through macropores like root channels. At the Canadian watershed (Renzetti et al., 1992), they felt both mechanisms could cause flow in the B horizon but pointed out the subsurface flow was twice the magnitude of the B horizon flow, even if they peaked at the same time.

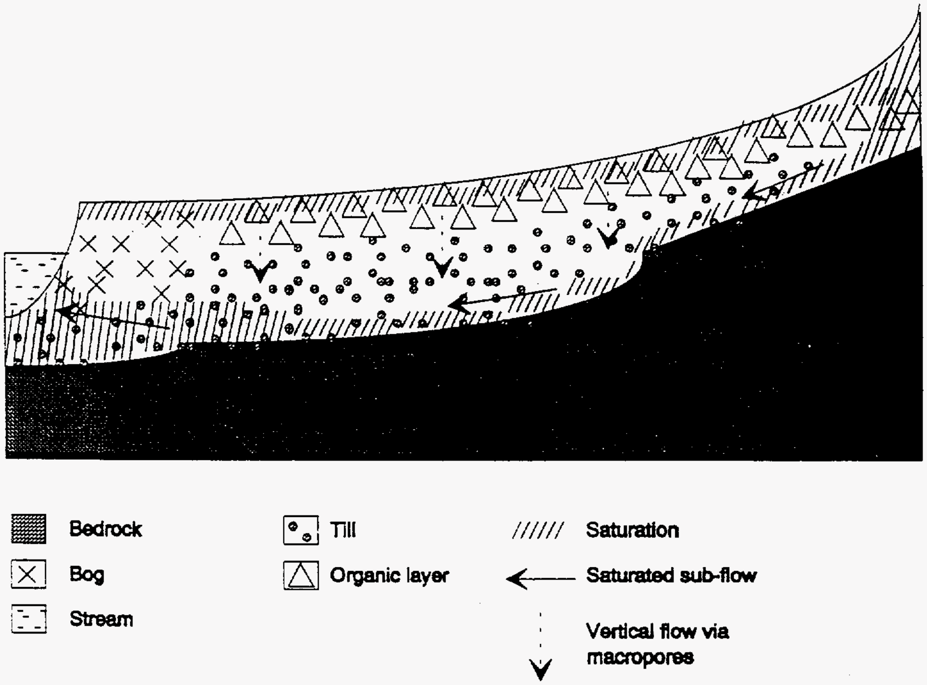

Fig. 4. A schematic representation of how subsurface flow may occur at Birkenes (adapted from Renzetti et al., 1992).

During wet periods, the ground water level in the bogs will be raised and water will flow quickest in the channels near the surface and through the more permeable surface layers (top 20 cm). In the effluent areas near the brook, the surface water will be mixed with ground water from the till. The overland flow will rapidly leave the area and the brook will then be fed with drainable water from the tills and the bogs. If there is no precipitation over a long period the discharge from the area will stop since the contribution from deep ground water is very small.

A sea-salt event at Birkenes (Mulder et al., 1990) showed that with wet antecedent conditions causing near saturated soils and high ground water tables intercepting organic soils, considerable lateral flow occurred through the top layers to the stream . The chemistry of this water reflects its passage through the similar organic layers of thin humus and podzol. This sea-salt event involved a lot more rain than the period simulated by the second EMMA run (Easthouse et al., 1992), 353 mm over 18 days compared to 168 mm over 15 days. Perhaps it is then not so surprising in the second EMMA run that most flow is from the B layer rather than from the O layers, which occurs during very rainy periods.

Tensiometer studies on hill slopes at Birkenes have shown that upward hydraulic gradients caused by pressure of upslope water can cause mixing of water through the layers (Hauhs, 1989). This shows that while we try to simplify flow paths between horizons and soil types, one must always remember the complex spatial distribution of soils and topography at Birkenes.

Water storage and replacement frequency

The difference between total input of rain and snow and the discharge is due to evapotranspiration. For the period 1974 to 1990 the average annual input was 1494 mm while the discharge was 1175 mm and consequently the evapotranspiration was 319 mm. Figure 3 illustrates the average annual hydrological budget. We have tried to estimate the amount of water stored in the area after snow melt and very heavy rain by looking at the hydrological budget for 8 selected years between 1974 and 1990. The amount of evapotranspiration in these dry years was 300 mm. We have distributed this evapotranspiration over the year by using the same distribution pattern as Jørgensen and Østmo (1990).

Figure 5 shows the average net input and discharge for the 8 selected years. The net input during June through August was negative due to high evapotranspiration, and the discharge stopped almost completely in July and August. We will define "drainable water" in this section as the difference between maximum saturation and field capacity. This means that all drainable water has left the area and the water content is falling below field capacity due to evapotranspiration.

During the first rainfall in September the net input was 183 mm while only 91 mm was leaving the area. This means that 92 mm was stored in the field. Since these 92 mm comprise the drainable water as well as the deficit below field capacity, the amount of drainable water must be less than 92 mm. During October and November there was almost equilibrium between net input and discharge. Accumulation of snow explains why input is higher than discharge during December, January and February (1977, 1982 and 1984 had cold winters). This "excess water" was discharged at the end of February and beginning of March. We have calculated that 84 mm of drainable water was removed from the soils before the discharge stopped (Fig.5). Consequently we believe that the amount of drainable water is close to 84 mm.

The drainable water will mainly be stored in the tills and the bogs. Figure 6 illustrates how the ground water level in the largest bog in the area (Langmyra) changed during 1974 and 1975. Clearly the bog was fully saturated during January and February and there was overland flow.

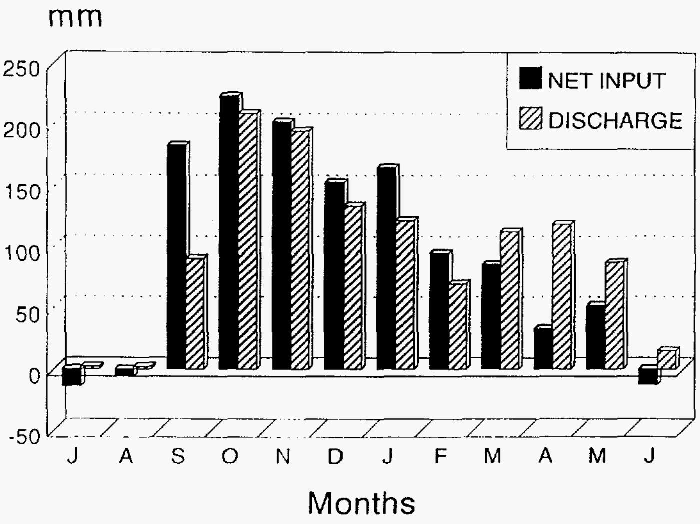

Fig. 5. The average net input and discharge of water for 8 selected years with dry summers at Birkenes.

The discharge in the brook stopped when the ground water level was at 20 cm depth. We calculated, using data from Brække (1981b) and Paivanen (1973) that the upper part of the bog, zone A on Fig. 6, contains when it is fully saturated nearly 100 mm of drainable water.

The area covered by bogs is 12.5 % of the total watershed. If 100 mm drainable water is representative for all the bogs, the amount of drainable water will be 12.5 mm when calculated for the whole drainage area. Due to evapotranspiration the ground water level in the bog was lowered another 40 cm during summertime. This amount must be filled up before any drainage from the bogs can start. The total contribution from the bogs to drainage and evapotranspiration this year was close to 300 mm or 37 mm for the whole drainage area.

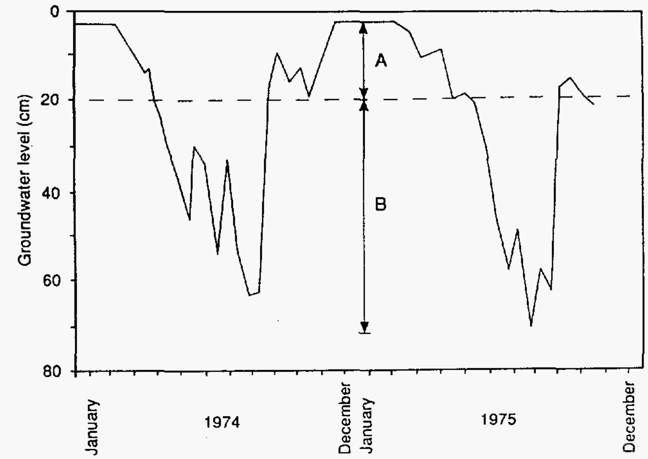

Fig. 6. The ground water level for Langemyrdalen, Birkenes, 1974-75. (Brække, 1981b) Zone A = upper 20 cm of bog, Zone B = lower bog, when bog is fully saturated.

Bedrock with a thin layer of humus soils cover 55.6% of the area. This humic soil can store about 40 mm with drainable water, which is equivalent to 22 mm for the whole drainage area. The remaining amount must be stored in the tills which are covering 23.2 % of the area. This means that the tills will contain when fully saturated 215 mm with drainable water.

The water contents in podzols at full saturation and different pF values are given in table 2.

Table 2. Water content (vol%) in a podzol at different pF values (data from the Norwegian Forest Monitoring Programme, NISK). Last column shows the amount of drainable water (mm) at pF=0.1bar.

Layer	Thickness (cm)	Saturated	0.02bar	0.1bar	mm H_2O 0.02bar	mm H_2O 0.1bar
FH	11	80.4	52.9	45.2	30.2	38.6
Ahe	5	49.1	37.4	24.9	5.9	12.1
Aeg	11	50.5	36.2	24.8	15.7	28.3
Bhfg	8	65.6	49.3	34.3	13	25
Sum	35				64.8	104

For the soils in this area the field capacity will be reached at a pF value between 0.02 and 0.1 bars. This means that the amount of drainable water stored in the upper 35 cm of the soil profiles will be between 65 and 104 mm and probably close to 100 mm. The remaining amount can be stored in the till if the average thickness is 1.5 m or more assuming an effective porosity (drainable) of 0.15. Even if there are considerable uncertainties in these calculations it will be very difficult to find storage capacity for more than 84 mm with drainable water. With an annual input of nearly 1100 mm, the drainable water stored in the field is replaced around 25 times in thin humus soils, and 10 times in the till and top layer of the bogs during autumn and early spring.

Water chemistry

We have calculated an Output-Input budget for the watershed using wet and dry deposition and discharge.

Wet deposition

The mean volume weighted (monthly data) composition of precipitation (wet only) is given in Table 3

Table 3. Mean volume weighted composition of precipitation (wet only) for the period 1980-1990 (µmol$_c$ l^{-1}) (NILU data from NIVA's monitoring programme data bank).

H^+	Ca^{2+}	Mg^{2+}	Na^-	K^+	NH_4^+	Cl^-	NO_3^-	SO_4^{2-}
57.7	9.26	12.93	54.32	4.29	42.28	64.05	39.89	68.11

The concentrations of Cl^- versus Mg^{2+} and Na^+ for different samples is illustrated in Fig. 7C and 7D. The lines in the diagrams show the ratios of these ions in sea water. We conclude from this that the precipitation contains a mixture of sea salts and other non-marine constituents.

The composition of these non-marine constituents is calculated by subtracting the contribution from sea salts, using chlorine as an internal standard. (Table 4).

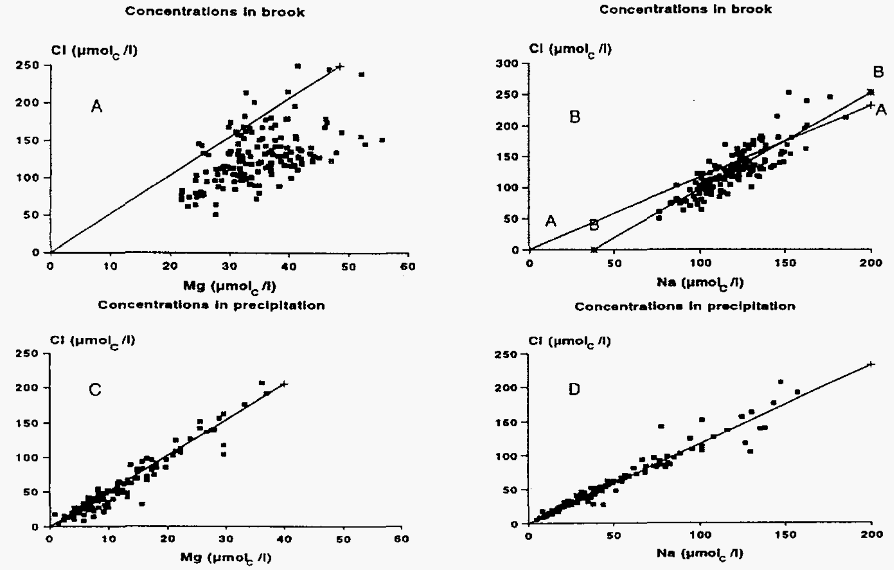

Fig. 7. The concentration of Na and Mg ions against Cl ions in precipitation and discharge.

Table 4. Composition of non-marine constituents in the precipitation ($\mu mol_c\ l^{-1}$).

H^+	Ca^{2+}	Mg^{2+}	K^+	NH_4^+	NO_3^-	SO_4^{2-}
57.5	6.97	0.57	0.05	42.28	39.89	61.7

The table shows that the non-marine contribution is mainly composed of "ammonium nitrate" and "sulphuric acid". The marine sulphate makes up 9.4% of the total sulphate.

Discharge

The average composition of the discharge (monthly data) is given in Table 5.

Table 5. Mean volume weighted composition of discharge for the period 1980-1990 ($\mu mol_c\ l^{-1}$) *(NIVA's monitoring programme databank).*

H^+	Ca^{2+}	Mg^{2+}	Na^+	K^+	NH_4^+	Cl^-	NO_3^-	SO_4^{2-}	Al_i	HCO_3^-
30.01	55.01	33.82	118.57	5.9	4.09	127.71	9.09	134.91	47.94	0.71

Al_i denotes inorganic monomeric Aluminium

The concentration of Cl^- versus Mg^{2+} and Na^+ for different samples is illustrated in Fig. 7A and 7B. The ratio of Mg/Cl is generally higher than for precipitation which reflects the net contribution from weathering, ionic exchange, mineralization minus nutrient uptake, taking place along the various flow paths. The Na/Cl ratios are not very different from what is found in precipitation. Line B-B is the best linear correlation, while line A-A represents the ratio in sea water. The average Na/Cl ratio in discharge is higher than the ratio in precipitation. With high concentrations (e.g. seastorms), this ratio is lower than in precipitation, probably reflecting a retention of Na^+ in ion exchange complexes. With low concentrations, the ratio is higher, probably reflecting a supply of Na^+ from weathering of alkaline feldspars and ion exchange processes.

Dry deposition of marine salts

The annual flux of Cl^- from the watershed is 145.97 $mmol_c\ m^{-2}$ while the input with wet precipitation is 95.69 $mmol_c\ m^{-2}$. We assume that Cl^- is conservative for this period of ten years, and that the difference, 50.28 $mmol_c\ m^{-2}$, must come from dry deposition. Again we assume that Cl^- has a marine origin and we calculate the following input as dry deposition (Table 6).

Table 6. Dry deposition of marine salts ($mmol_c\ m^{-2}$).

Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-	SO_4^{2-}	HCO_3^-
1.89	9.75	43.09	0.94	50.28	5.18	0.22

Dry deposition of other constituents

The soils have for long periods been flushed with high and relatively constant concentrations of sulphate. We therefore assume that there will be little sulphate adsorption/desorption in the soils. This was also found by Christophersen and Wright (1981) and by Eriksson et. al. (1992) for watersheds along the S.W. coast of Sweden. The sulphate reduction in the bogs will be small due to the very slow water movement in the deeper parts of these bogs, and the sulphate reduction in the upper part is balanced by sulphide oxidation. We therefore assume sulphate behaves conservatively. The relative composition of other constituents is assumed the same as in wet deposition and we calculate the remaining non-marine dry deposition (Table 7).

Table 7. Dry deposition of other constituents than marine sea salts. (mmol$_c$ m^{-2}).

H^+	Ca^{2+}	Mg^{2+}	K^+	NH_4^+	NO_3^-	SO_4^{2-}	Org.An.
44.05	5.34	0.43	3.83	32.38	30.53	47.27	8.23

Output - input budgets

All these input and output data are summarised in Table 8. This budget includes both sea salts and dry deposition and so differs from the input/output budgets shown in SFT reports (Statens forurensingstilsyn, 1993).

These calculations show:

- The input of sulphate with dry deposits was 34% of the total sulphate deposition.
- 95 % of ammonium and 89% of the nitrate has been consumed.
- A large amount of protons were consumed in the watershed while Ca, Mg, Na and Al were released.
- There has been an annual loss from the watershed of nearly 51 mmol$_c$ m^{-2} with Ca^{2+} and Mg^{2+}.
- There has been a small annual gain to the watershed of K^+.

Table 8. Output - input budget for the Birkenes catchment averaged for the years 1980-1990 (mmol$_c$ m^{-2} year^{-1}).

	Output	Wet dep.	Sea salts Dry dep.	Other Dry dep.	Output-input
H^+	34.3	85.91	0	44.05	-95.41
Ca^{2+}	62.88	13.83	1.89	5.34	41.81
Mg^{2+}	38.66	19.32	9.75	0.43	9.15
Na^+	135.53	81.15	43.12	0	11.25
K^+	6.74	6.41	0.94	3.83	-4.43
NH_4^+	4.67	63.17	0	32.38	-90.87
Cl^-	145.97	95.69	50.28	0	0
NO_3^-	10.39	59.6	0	30.53	-79.74
SO_4^{2-}	154.2	101.76	5.18	47.27	0
HCO_3^-	0.81	0	0.22	0	0.59
Al^{3+}	54.8	0	0	0	54.8
Org.An.	26.2	12.74	0	8.23	5.23

Throughfall

Wet deposition to the forest has been measured as a part of the Norwegian Monitoring Programme for Forest Damage (OPS) (Horntvedt et al., 1992). The data given here (Table 9) was collected during a 2 year period (Aamlid, 1992; Aamlid, 1993).

The fluxes show great annual variation. The differences are mainly due to variation in the precipitation amount (Table 9) as there are only small differences in the annual mean concentration. The amount of throughfall precipitation varied from 14.5 % to 21 % of the open area precipitation (mean = 17.5 %). The variation is probably within the expected range. However, due to the relatively open forest type, the interception at the Birkenes plot is less than found in other spruce forests in Norway; 22-24% (Aamlid, 1992; Aamlid 1993). The date clearly demonstrate that most elements (H, Na, Mg, Mn,

SO_4-S, Cl, K and Ca) are enriched in the troughfall precipitation related to open area precipitation. Only ammonium and nitrate ions are reduced, while the rest-N (total-N - NO_3-N + NH_4-N), which is mostly organic N is enriched. The enrichment of all these elements could either be due to dry deposition or foliar leakage. The open precipitation collects wet and to some extent dry deposition, but this is no good measure of the total deposition to the forest surface. Therefore it is necessary to look at some older detailed measurements from the same plot. Table 10 shows the input with rain as well as the throughfall from different trees during the period July - November 1972. The values are given as $mmol_c\ m^{-2}$ for the whole period, while the amount of water is given in mm ($l\ m^{-2}$) for the same time interval.

Table 9. Throughfall- and open field precipitation fluxes at the forest monitoring station at the Birkenes catchment 1990-1992.

	Year	Weeks	Precip. (mm)	H^+	Na^+	Mg^{2+}	Mn^{2+}	NH_4^+	NO_3^-	SO_4^{2-}	Cl^-	tot-S	K^+	Ca^{2+}	rest-N
K	90	41	1587	104.53	203.18	24.44	1.15	42.76	56.69	77.26	238.4	78.45	45.7	21.13	31.63
K	91	41	1084	63.67	125.79	17.2	0.96	27.2	40.12	55.12	153.81	57.99	35.14	14.62	27.63
K	92	43	1209	61.69	113.4	15.51	1.11	21.35	35.77	53.52	140.27	56.67	37.31	14.05	22.34
K	Mean	42	1293	76.63	147.46	19.05	1.07	30.48	44.19	61.98	177.5	64.38	39.38	16.59	27.21
N	90	41	1856	80.7	140.54	12.92	0.35	54.47	58.54	49.69	161.26	49.59	8.03	5.94	23.56
N	91	42	1375	56.79	98.87	9.18	0.25	49.4	57.83	37.27	116.55	38.8	5.55	4.67	13.57
N	92	43	1470	62	79.21	7.12	0.27	51.19	61.83	40.24	93.22	41.83	5.78	4.17	0.99
N	Mean	42	1567	66.5	106.18	9.75	0.29	51.69	59.4	42.39	123.68	43.42	6.44	4.94	12.71

K = throughfall, N = open field. Chemical fluxes are in $mmol_c\ m^{-2}$.

Table 10. Amounts in $mmol_c\ m^{-2}$ for open precipitation and throughfall from different trees. The period studied was July to November 1972 (data from Bjor et al., 1974).

	mm	H^+	Ca^{2+}	Mg^{2+}	Na^+	K^+	NH_4^+	Cl^-	NO_3^-	SO_4^{2-}
Precip.	388	21.13	2.68	3.74	18.46	1.23	9.43	18.13	9.43	27.13
Birch 15 m	318	12.1	10.35	11	28.93	12.09	4.17	31.32	6.83	30.4
Birch 20 m	362	14.45	18.96	12.92	37.48	19.37	1.4	46.3	2.39	40.09
Pine 15 m	359	28.55	17.15	14.84	55.14	17.87	2.36	56.15	4.25	47.07
Pine 20m	295	28.93	19.13	15.7	62.55	16.76	1.4	66.98	2.72	45.81
Spruce 5 m	318	17.38	9.43	7.52	26.73	12.8	3.16	31.31	6.2	33.22
Spruce 15 m	252	30.28	18.87	12.45	46.77	19.28	1.04	49.77	2.34	51.62
Spruce 20 m	246	35.72	20.33	14.06	56.78	18.87	0.61	60.98	1.73	56.16

This table clearly illustrates that the fluxes of chloride and sulphate with throughfall are much larger than the input with rain. This can either be due to leakage from the canopy or adsorption of aerosols (dry deposition).

Leakage of sulphur from the canopy or uptake of sulphate is limited in this type of coniferous forest (Lindberg et. al. 1986). Leakage of chloride must be small compared to the values found here. Consequently we again make the assumption that chloride and sulphate are conservative components. We also assume that all chloride comes from sea salts. If these assumptions are correct, it is clear that the amount of aerosols adsorbed varies with type of trees and size of the trees. This generally means that the taller the tree and the denser the canopy, the more dry deposition a forest will adsorb. Since most of the forest in Birkenes is old (on average 80 years), we are using data for the large spruce (most representative for the catchment) to calculate the total input as well as leaching from the canopy (table 11). The dry deposition is calculated in the same way as described previously in the water chemistry section, with the chloride and sulphate assumptions mentioned above. We believe this is a more realistic method than using air concentrations and deposition rates, because the filter effect of trees is the most important factor at a windy site like Birkenes (Hasselrot and Grennfelt, 1987).

Table 11. Chemical budget for the largest spruce studied (Units mmol$_c$ m^{-2}) (data from Bjor et al., 1974).

	H^+	Ca^{2+}	Mg^{2+}	Na^+	K^+	NH_4^+	Cl^-	NO_3^-	SO_4^{2-}	Σcations -Σanions	Σcations
Precip.	21.13	2.68	3.74	18.46	1.23	9.43	18.13	9.43	27.13	1.98	56.67
Dry sea salts	0	1.61	8.31	36.74	0.80	0	42.85	0	4.41	0.2	47.46
Dry other	22.94	2.78	0.22	0	1.25	16.86	0	14.42	24.62	5.01	44.05
Total dep.	44.07	7.07	12.27	55.2	3.28	26.29	60.98	23.85	56.16	7.19	148.18
Through-fall	35.72	20.33	14.06	56.78	18.87	0.61	60.98	1.73	56.16	27.5	146.37
Canopy leach.	-8.35	13.26	1.79	1.58	15.59	-25.68	0	-22.12	0	20.31	-1.81

These calculations, for one large spruce, illustrates that more than 97 % of the ammonium and 92 % of the nitrate deposited (wet and dry) is consumed in the canopy.

The reduced amounts of N-components may be caused by at least two main pathways. Since N is known as the main limiting factor for forest growth in Norway (Brantzæg et al., 1970) it is reasonable to believe that the trees take up as much as possible of available nitrogen. The demonstrated difference between open field precipitation and throughfall precipitation may be caused by a direct uptake through needles (Granat and Hällgren, 1992; Eilers et al., 1992; Brumme et al., 1992; Rosen and Lundmark-Thellin, 1985). Another explanation may be uptake by other organisms, e.g., epiphytic lichens (Friedland et al., 1991; Lang et al., 1976; Lang et al., 1980). As pointed out by Bjor, Horntvedt and Joranger (1974) there are different types of lichens on these trees which will greatly enhance the uptake of ammonium and nitrate (Eldhuset and Røsberg, in prep.). Enrichment of nitrogen in throughfall precipitation is reported in the Norwegian forest damage monitoring at just those sites where lichens are sparse or lacking in the crowns. This lack of lichens may be associated with rapid growth of air pollution (SO_2) (Aamlid 1992, 1993). The lichens and algae release part of the nitrogen as organic-N, which is often greatly enriched in throughfall (Alenäs and Skärby, 1988).

It must be remembered that these measurements were done during the maximum growing season, and the conditions are different during wintertime. These measurements only demonstrate that a considerable part of the nitrogen components are consumed on the trees.

Ca, Mg and Na to a small extent, K to a large extent and organic anions are all released by the canopy. As the Na content of needles is very low, Na probably does not leach from the canopy and the error may be due to assuming all Cl comes with sea salts.

It is interesting to notice that the total flux of cations with throughfall is almost identical to the sum of the inputs. This means that the nitrogen consumption is mainly an "ion exchange process".

The dry deposition of sulphate is about 50% of the total deposition under this tree. Even if the pH of the throughfall is lower than for the wet precipitation, the flux of H^+ is lower and thereby a net consumption of protons takes place in the canopy.

The soils

A surface material map (Fig. 2) was created using Lundquists surface geology map (Lundquist, 1976) as a base and 157 soil sampling points. These points represent profiles sampled by layer as well as averages of composite bore samples. The sampling period spans from 1974 to 1992 and some points have been resampled once or twice in different years.

The soils have been grouped into 3 types:

- Bogs.
- Tills. The most common soil type on the tills is podzol, with some brunisols and gleys. Micro-podzolisation has occurred in many acid brunisols (Pipjers and Mulder, 1990), making it hard to separate different soil types. The profiles were divided into the 4 basic horizons of a podzol: the O or organic layer, the E or eluvial (leached) layer, the B or illuvial (enriched) layer and the C or base rock layer.
- Thin humus soils on bare rock. There are 3 types of this soil, showing increasing depth and complexity. The first is a mixture of moss and needles, directly on rock. The second is identical, but has a thin layer of crumbling rock above the intact rock. The third type is found in crevices and depressions in the rock and consists of dead organic matter, a leached layer and crumbling parent rock.
- Bare rock. Purely bare rock with lichen.

The map is a very simplified version of the real complexity found in the catchment. This is clearly seen when comparing it with the detailed soil surveys of sub-areas done by Pijpers and Mulder (1990) and Fjeldal (1992). However 3 groups of soil type, divide the catchment into manageable segments for modelling, both hydrologically and chemically.

Table 12 shows some average chemical characteristics of the soils. The pH tends to increase with depth and is lowest in the organic podzol horizon. CEC and BS are greatest in the organic podzol horizon, followed by thin humus and bog and then decrease with depth in the mineral podzol horizons. With base cations, both the thin humus and the organic podzol horizon have roughly the ratios 20Ca:6Mg:3K:1Na. The illuvial and eluvial layers and bog soil follow this ranking, although the ratios are different.

Table 12. Average physical and chemical properties of the soil groups at Birkenes. Number of samples in parentheses.

Soil type	Depth (cm)	pH (H_2O)	Kjeldahl total-N (mmol kg^{-1})	CEC (mmol$_c$ kg^{-1})	BS (%)	Ca (mmol$_c$ kg^{-1})	Mg (mmol$_c$ kg^{-1})	K (mmol$_c$ kg^{-1})	Na (mmol$_c$ kg^{-1})
BOG	20.0 (34)	4.6 (2)	1390 (1)	235 (34)	21.5 (34)	27.5 (34)	10.3 (34)	5.6 (34)	3.2 (34)
THIN HUMUS	6.9 (49)	4.2 (13)	867 (3)	202 (34)	44.7 (34)	51.2 (49)	17.9 (49)	10.2 (49)	2.9 (49)
TILL O-horizon	9.1 (97)	3.9 (25)	1100 (16)	257 (38)	54.7 (38)	84.4 (97)	26.7 (97)	12.8 (97)	4.0 (97)
TILL E-horizon	9.6 (34)	4.3 (11)	46.3 (2)	25.9 (16)	18.7 (16)	2.2 (34)	1.1 (34)	0.9 (34)	0.5 (34)
TILL B-horizon	20.9 (54)	4.5 (25)	204 (15)	44.5 (37)	8.0 (37)	1.4 (54)	0.7 (54)	0.7 (54)	0.5 (54)
TILL C-horizon	20.8 (6)	4.7 (3)	33 (3)	6.3 (3)	7.6 (3)	0.9 (6)	0.2 (6)	0.4 (6)	0.3 (6)

Bog soils

There are 2 series of bog soil measurements, one from Langemyra in 1974 and the other from Vestre Tveitdalen in 1989, plus a few scattered measurements from Tveitdalen in different years (Table 13). If one compares the exchangeable base cations, it is clear that Tveitdalen in 1989 has less than half the concentration of Langemyra in 1974. However the difference could be caused by spatial variability, time or both. The other measurements tend to suggest that Vestre Tveitdalen has lower base cation concentrations. It would be worthwhile resampling Langemyra to see if base saturation is still as great here.

Table 13. Average exchangeable base cations at 2 bog localities. Number of samples in parentheses.

	Ca ($\text{mmol}_c\ \text{kg}^{-1}$)	Mg ($\text{mmol}_c\ \text{kg}^{-1}$)	K ($\text{mmol}_c\ \text{kg}^{-1}$)	Na ($\text{mmol}_c\ \text{kg}^{-1}$)
Langemyra, 1974, 20cm, (5)	40.2	20.3	15.1	7.6
V. Tveitdalen, 1989, 20cm, (25)	26.7	8.6	4.5	2.5

Changes in chemical characteristics of the podzols with time

The podzol samples were divided into 3 time series: 1974, 1984 and 1986-1992. Thin humus and bog soils were not included due to lack of samples. The time periods are unequally spaced but was the best solution considering the assorted collection of soil samples. The samples at each time period are from points spread all over the catchment, although the sample locations are different for each time period. However despite this heterogeneity in sampling, it is hoped that the mean from each sampling period can give an overall picture of changes in the podzol with time. The Duncan's multiple range test (a parametric multivariate test) was used to test the significance (a = 0.05).

Table 14. Results of the Duncan's multiple range test between the 3 time periods.

	Buffered extraction			Unbuffered extraction										
	CEC	BS	exch. acidity	CEC	BS	exch. acidity	Na^+	K^+	Mg^{2+}	Ca^{2+}	pH H_2O	pH $CaCl_2$	total-N	LOI (%)
O 74-84	ns	ns	ns	ns	ns	nd	ns	*	ns	ns	ns	ns	ns	ns
O 74-92	ns	ns	ns	ns	ns	nd	ns	ns	ns	ns	ns	ns	ns	ns
O 84-92	ns	ns	*	ns	*	*	ns	ns	ns	ns	ns	ns	ns	ns
E 74-84	ns	ns	ns	ns	ns	nd	ns	ns	ns	ns	ns	ns	ns	ns
E 74-92	ns	ns	*	ns	ns	nd	ns	ns	ns	ns	ns	ns	ns	ns
E 84-92	ns	ns	*	ns	ns	*	ns	ns	ns	ns	ns	ns	ns	ns
B 74-84	ns	ns	ns	ns	ns	nd	ns	ns	ns	ns	*	ns	ns	ns
B 74-92	ns	ns	ns	ns	ns	nd	ns	ns	ns	ns	*	ns	ns	ns
B 84-92	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
C 74-84	ns	ns	*	ns	ns	nd	ns	ns	ns	ns	ns	ns	ns	ns
C 74-92	ns	ns	*	ns	ns	nd	ns	*	ns	ns	*	ns	ns	ns
C 84-92	ns	ns	*	ns	ns	ns	ns	*	ns	ns	ns	ns	ns	ns

ns denotes not significant, * denotes significance at 5 % level and nd denotes not determined.

Analysis of Base cations, Aluminium, Iron, Manganese and exchangeable acidity

The only base cation to show significant changes was K in the O horizon, which dropped significantly from 1974 to 1984 and increased in 1992. No significant changes occurred in the E and B horizons. In the C horizon, K dropped significantly from 1974 to 1982 and then increased significantly from 1984 to 1992. No obvious trend appeared in the base cation data.

Exchangeable Mn, Fe and Al were measured in 1984 and 1986-1992 (not shown in table 14). Highest values occurred in the O horizon, followed by the B horizon and smallest values in the E horizon. All horizons for all 3 cations showed higher values in 1984 compared to 1992, except for Mn in the O horizon which had its highest value in 1992.

Buffered exchangeable acidity (Total acidity) was measured at all 3 time points For O, E and B horizons acidity increased from 1974 to 1984 and then dropped to a lower level than 1974 in the 1986-92 period. In the C horizon, acidity decreased with time. Unbuffered acidity was only measured in 1984 and 1986-92 and in all horizons decreased from 1984 to 1992.

BS was always at its lowest in 1984. With unbuffered BS in the O horizon, there was a significant rise from 1984-1992. In the O and C horizon it returned in 1986-92 to the same level as in 1974. In the E and B horizons, it returned to a higher level than that in 1974.

CEC in each horizon was highest in 1984. In 1992 with the O and B horizon, it returned to a higher value than in 1974. With the E and C horizons it returned to a lower value than in 1974. No significant differences were found between the CEC in any horizon with time.

Analysis of pH

pH was measured in water and $CaCl_2$. The pH was always lower with $CaCl_2$ and mimicked the changes in pH(H_2O) but with smaller oscillations. The pH in the O horizon has decreased 0.1 unit (ns) in the last period. In the E, B(1974-1984 and 1974-1992 *) and C(1974-1992 *) horizons, the pH was lower (0.4 units (0.8 units in C)) in 1984 and then rose slightly or stayed at the same level.

Analysis of organic matter

Loss on ignition (LOI) and total-N has increased in each horizon from 1974 -1992 (ns), except for in the E horizon, where although LOI and total-N increased in 1984, both decreased in the last measurement.

Results of other soil comparisons at Birkenes

A comparison of the soil status after 5 years (1986-1991) at the Birkenes intensive forest research plot showed few significant changes (Jensen, 1993). Here composite bore samples from the same location were compared. The only significant changes were an increase in Ca in the Ae horizon, while Ca decreased in the B horizon. The 13 profiles dug in scattered locations over the entire catchment in 1974 (Frank, 1980) were resampled as close as we could locate them in 1992. The only significant change was an increase in the O horizon in pH. However the trend with time was decreased Ca and Mg and therefore BS in the O horizon, while pH increased in the top horizons and decreased in the lower mineral horizons.

Conclusions on soil comparisons

The resampling of soil at the same site and horizon with time at Birkenes does not give any clear trend in changes of chemical characteristics with time (Statens forurensingstilsyn, 1993). There could be several reasons for this. Firstly we may not be managing to resample at the same site or exact horizon and soil heterogeneity is so great that difference in space is greater than difference in time. Secondly the time of year should be the same at each sampling as plant uptake causing acidity in the top soil horizons is greatest in Spring and Summer. Thirdly weather conditions before and during sampling should be as similar as possible. All 3 factors occurred while resampling the original 13 profiles (Frank, 1980). It was hard to relocate profiles after 19 years and we resampled in dry conditions in June compared to soils at field capacity in October, 1974. Fourthly sampling horizons precisely with a soil borer is difficult practically and humus/mineral components can pollute horizon samples.

The results of our comparison show 1984 as the time when base saturation in the O horizon, pH in mineral horizons are at their lowest, while exchangeable Mn, Al, Fe and acidity were at their highest. These soil samples were taken in June 1984 from 4 locations where soil and vegetation type were as homogenous as possible. Yet the same soils sampled in 1992 showed no significant trends (Statens forurensingstilsyn, 1993). We see no reason why 1984 or these locations should give this result. It shows the great variation in space rather than time among podzols at Birkenes.

Weathering

Chemical weathering is defined in this study as the release of base cations from minerals either to solution, to an exchange site, or to the biota. There are a number of reasons for studying chemical weathering. From an ecological perspective, chemical weathering of minerals is the major long-term source of nutrient cations (Ca, Mg, K, Na) to the exchangeable pool, and is the only major process by which acidic deposition is neutralised on a time-scale of decades or even centuries. Therefore, if the losses (output-input) of base cations are larger than that released due to weathering, the excess lost must be taken from the exchange complexes. The element losses from this watershed are given in Table 8.

The release of different base cations from weathering depends upon composition and amount of various minerals in different size fractions (surface area), the minerals' relative weathering rate, temperature, pH in water, pathways for the water and water fluxes. The composition of different soils and soil horizons depends upon previous weathering history. Thin soils have been depleted of the most easily weathered minerals which are still found in deeper parts of other soils. Obviously the weathering reactions taking place in a 5 cm thick strongly weathered soil are different from the reactions taking place in the different layers of a 2 m deep podzol profile developed on till, since the relative percentages of the more resistant minerals will increase upon weathering. As a result it is very difficult to calculate the contribution from weathering.

Paces (1986) assumed that silica released by weathering is nearly conservative. We have modified his weathering model slightly. Since there is no input of water soluble silica with total deposition the amount leaving the watershed with discharge is due to weathering. The average Si content in the discharge is 65 µmol l^{-1}(Christophersen et. al. 1990a). In contrast to aluminium the amount of silica is decreasing with increasing discharge (and increasing proton concentrations). An average annual discharge of 1175 mm results in the removal of 76.4 mmol m^{-2} with Si.

Water analyses showed that there is considerable weathering on bedrock covered with thin soils (Table 23), while there is no weathering in the surface of the bogs covering 12.6% of the area and the contribution from the bedrock under the bogs must be very small due to the very low permeability of these organic soils. During warm periods with low discharge, there is a diatom production and water leaving the area has a very low Si content. The amount of water is normally between 150 and 200 mm. Probably there is a small precipitation of Si in the soil profiles. When we make corrections for the bog area (12.6%) and diatom production (175/1175 should contain average concentration of Si), we find that the release of Si from tills and areas covered with no or very thin soils must be close to 98 mmol m^{-2}.

Till is covering 23.2% of the area while thin soils (thin humus soils on bare rock and bare rock) are covering 64.2%. If we use an average weathering rate for all tills (R_{TILL}) and another average rate for thin soils ($R_{T.SOIL}$) we can write the following relation:

$$R_{AV} \times 87.4 = R_{TILL} \times 23.2 + R_{T.SOIL} \times 64.2 \qquad \text{(equation 1)}$$

where R_{AV} is the average release of Si. The value for $R_{AV} = 98$ and consequently we find that:

$$R_{TILL} = 369 - R_{T.SOIL} \times 2.77 \qquad \text{(equation 2)}$$

The average Si content in soilwater collected from lysimeters in tills is 109 µmol l^{-1}. As these lysimeters were of the PRENART type, there could be a small leakage of Si from the lysimeter apparatus, but we have chosen to ignore this possible error source. If we assume that water draining tills contains 109 µmol l^{-1}, the contribution from thin soil weathering must be 62 µmol l^{-1}since this will give a discharge containing 65 µmol l^{-1}. This means that the release of silica from tills is 1.68 times higher than from thin soils.

When we substitute this into equation 2 we find that:

$$R_{TILL} = 139 \text{ and } R_{T.SOIL} = 83 \text{ mmol m}^{-2} \text{ with Si.}$$

Base cations released due to weathering.

The main minerals in the till as well as the in situ derived weathering products are quartz, alkaline feldspars, amphibole, pyroxene, epidote and muscovite. Microprobe showed that the feldspar is a perthite with the composition $(Na_{0.48}K_{0.52})AlSi_3O_8$ derived from the Birkenes granite. The dark minerals had a ratio of (Ca + Mg) to silica near 0.8 but the ratio as well as type of ions varied.

If we assume that all cations came from the alkaline feldspar the annual release of base cations for the whole area would be 33 $mmol_c\ m^{-2}$. If the cations came from the dark minerals it would be 157 $mmol_c\ m^{-2}$, since the ratio of base cations ($mmol_c$) to Si (mmol) is close to 1.6 in these minerals. These values define the possible range, but the actual release must be closer to the minimum than the maximum value as the relative percentages of the more resistant minerals (alkaline feldspars) will increase upon weathering in the thin soils and top layers of till.

According to our Output - Input budget the loss of Na is close to 11 $mmol_c\ m^{-2}$. We assume that this was released from the alkaline feldspar and consequently 12 $mmol_c\ m^{-2}$ with potassium would be released from the same mineral. This weathering would release 69 mmol with silica. If the remaining 29 mmol with silica comes from dark minerals, these would release 46 $mmol_c$ with Ca and Mg. Consequently the total annual release of base cations would be 69 $mmol_c\ m^{-2}$.

There are certainly other minerals contributing to the weathering but including these minerals makes it necessary to reduce the contribution from the minerals mentioned.

Using equation 1 and the same release ratio for Si, namely 1.68 more cation is released from till than thin soils, one can calculate the following release of base cations.

Table 15. Annual release of base cations and silica from thin soils and tills.

	Na ($mmol_c\ m^{-2}$)	K ($mmol_c\ m^{-2}$)	Ca+Mg ($mmol_c\ m^{-2}$)	Si ($mmol\ m^{-2}$)
Catchment	11	12	46	98
Thin soil	9	10	39	83
Till	16	17	65	139

Even if there are several assumptions made in these calculations, it must be remembered that the annual release of silica is controlling the calculated release of base cations.

The high contribution from thin soils is important, since this is the dominating soil type in many areas with a heavy load of acid precipitation.

Jacks (1990) has presented a review of weathering studies in Scandinavia where he states that rocky areas with very thin soil covers are likely to have annual weathering rates, measured as release of basic cations, of 5 - 20 $mmol_c\ m^{-2}$. The value depends upon the mineralogical composition of the soil. Till areas with a thick soil cover (1-5 m) have weathering rates of 20 -50 $mmol_c\ m^{-2}$. The rate is increasing with increasing content of dark, easily weatherable minerals like hornblende and biotite. Since the composition of the Birkenes granite and the granite derived soils have fairly high contents of easily weatherable minerals we would expect the weathering rate to be fairly high in thin soils as well as in tills.

The forest

Results of catchment forest survey

The site index for Norway spruce (*Picea abies* (L.) Karst.) was estimated according to the H_{40} system (Tveite, 1977). 5% of the plots had a site index less than $H_{40} = 5$ and were described as unproductive. These were situated on bare rock. The other plots could be divided into a low quality class (H_{40} = 5-10.9) 23.4%, a medium quality class (H_{40} = 11-16.9) 66.7% and a high quality class ($H_{40} > 17$) 9.9%. The latter was mostly found on tills.

The following tree species distribution was found according to volume over bark: Norway spruce 78.5%, Scots pine (*Pinus sylvestris* L.) 10.7%, Birch (*Betula pubescens* Ehr.) 4%, Aspen (*Populus tremula* L.) 3.6%, Pedunculate Oak (*Quercus robur* L.) 2.7% and Rowan (*Sorbus aucuparia* L.) 0.5%. The deciduous trees tend to dominate on some bog and thin humus sites, particularly on the higher slopes. The average standing volume at Birkenes is 177m³/ha.

Results of Biomass Increment calculation

The biomass increments of the 85 plots were not normally distributed, so the median biomass increment of each tree part is used. The stem over bark increment varies between 80 - 8498 kgDM/ha/year and the total biomass increment gives a N uptake varying between 0.1- 22 kg/ha/year (0.8-156 mmol/m²/year). The median net uptake by trees at Birkenes is presented in Table 16, while Table 17 shows the median net uptake from the 4 different soil types at Birkenes.

Table 16. Median net uptake by trees in the Birkenes catchment (kg/ha/year)

	Stem over bark	Branches	Needles	Root and stump	Total above ground net uptake	Total net uptake
N	3.32	1.9	2.05	2.68	7.27	9.95
Ca	3.32	1.48	0.69	1.34	5.49	6.83
Mg	0.45	0.26	0.2	0.36	0.91	1.27
K	2.86	0.8	0.94	1.97	4.6	6.57

Table 17. The median net uptake from the 4 soil types and from the whole catchment weighted according to soil type area (mmol$_c$ m^{-2} year^{-1})

	bog	till	thin humus	bare rock	catchment
N	84.11	90.96	73.54	16.08	74.01
Ca	40.83	45.21	35.61	7.99	36.12
Mg	12.55	13.63	10.97	2.42	11.05
K	19.75	21.98	17.53	3.93	17.67

There are many error sources in this biomass increment method of calculating net uptake by trees (see Materials and Methods section). σ is the standard error of regression.

1. Regression was used to find 66% of heights in 1993.
   σ = 3.62 m, spruce. σ = 4.77 m, pine.
2. Most of the heights and diameters in 1983 are found by regression .
   σ = 1.67 cm, D1.3, 1983, spruce. σ = 0.67 cm, D1.3, 1983, pine.
   σ = 2.99 m, H, 1983, spruce. σ = 1.04 m, H, 1983, pine.

3. The maximum and minimum values of the 95% prediction intervals for H 1993, D1.3 1983 and H 1983 were used in the Marklund functions. This gave a total above ground biomass increment varying by +/- 12.6%.
4. The accuracy of the biomass functions for trees in Birkenes could not be tested, because of lack of real biomass increment measurements. However others have tested Norwegian material against Marklund's functions (Brække, Lunnan and Rørstad, 1994). They found that Norway spruce stands gave satisfactory results, while the Marklund functions underestimated aboveground biomass of the pine by about 20%. The use of Marklund functions in Norwegian forests has also been investigated by Blingsmo, 1992. He concludes that the functions give reasonable results with possibly an overestimate in biomass for large spruce volumes.
5. The nutrient contents of all parts of the trees, except the spruce needles, were determined from trees in Southern Sweden and Norway. Estimated standard deviations are shown in parenthesis in table 18.

Table 18. Element concentrations in %DM of different tree compartments.

Norway spruce	Stem over bark	Branches	Needles	Stump and roots
N	0.14 (0.04)	0.44 (0.11)	1.09	0.3 (0.01)
Ca	0.14 (0.05)	0.36 (0.04)	0.39	0.15 (0.02)
Mg	0.02 (0.03)	0.06 (0.02)	0.11	0.04 (0.01)
K	0.13 (0.02)	0.17 (0.09)	0.51	0.22 (0.02)
Scot's pine	Stem over bark	Branches	Needles	Stump and roots
N	0.1 (0.02)	0.35 (0.05)	1.32 (0.15)	0.3 (0.01)
Ca	0.1 (0.02)	0.2 (0.03)	0.3 (0.09)	0.15 (0.02)
Mg	0.01 (0.01)	0.05 (0.01)	0.1 (0.01)	0.04 (0.01)
K	0.05 (0.02)	0.2 (0.03)	0.55 (0.09)	0.22 (0.02)

6. Although the forest surveying represents 85 plots evenly distributed over the catchment, it is possible that some forest type is over or under represented in the survey compared to the real situation.

Adding the standard deviation for the biomass increment (3) and for the element concentrations give these maximum possible uptake errors:

N +/-34%, Ca +/- 36%, Mg +/-86%, K +/-27%.

These error sources could cause one to be sceptical to the result, however other studies show the result is reasonable. 1-15 kg N/ha/year is said to be net accumulated by Scandinavian forests over the commercial rotation period (Løfgren, 1991). Unfertilised Scots pine growing in a nutrient poor bogsite accumulates 5.3 kg/ha/year in the first 31 years (Brække, 1990). At Nordmoen, soil uptake of N is calculated as 15.1 and 21.6 kg/ha/year on 2 different plots (Johnsen and Lindberg, 1992). Root production as a percentage of total production (Nihlgård, 1972) is 19%. If one calculates the Marklund median stump and root increment/ total Marklund tree increment, it is 21%. This gives some support to the root biomass calculation.

Base cation budgets

Two base cation budgets are calculated; one for the whole catchment (Table 19) and one for the till area (Table 20). The till budget is interesting as it shows the nutrient state for the more fertile forest sites in the catchment.

The exchangeable base cation capital for the whole catchment was calculated by weighting the base cation values for each soil type by the % coverage of till (podzol), thin humus, bog and bare rock. The O, E and B horizons were included in the till capital, while the bog capital was calculated with a 20 cm depth, as it is this horizon that normally contributes to stream water (see Fig. 6). For the organic horizons, the new 1993 soil data (table 1) was used. For the bog, 1989 -1990 data for the lower 15 cm depth and for the podzol E and B horizons, 1984-1992 data was used. The calculations for the weathering rates were explained in the weathering chapter. As it was too difficult to find separate release rates of Ca and Mg from dark minerals, (Ca + Mg) are given as a lump sum for weathering. This forced us to also lump (Ca + Mg) in the base cation budget as weathering is such an important process. The net uptake by trees is given in Table 17. Table 8 gives the loss to stream values for the catchment. For the till budget, we must assume that the base cations lost from an average catchment m^2 of soil is the correct amount lost from a m^2 of podzol. The podzol probably contributes more base cations than other soil types because of its larger CEC and therefore buffering ability, but in contrast probably has less water flowing through it to carry the cations away than the thin humus soils.

Table 19. The catchment base cation budget (units $mmol_c\ m^{-2}\ year^{-1}$).

	(Ca + Mg)	K	Na
Exch. base cation capital	1655	263	130
Catchment weathering	46	12	11
Net uptake by trees	-47	-18	
Taken out at harvest	-15	-6	
Catchment change (Output - input)	-51	4	-11
% annual change on CE complex (short-term)	-3.1	-0.8	0
% annual change on CE complex (long-term)	-1.2	3.8	0

Table 20. The till base cation budget (units $mmol_c\ m^{-2}\ year^{-1}$).

	(Ca + Mg)	K	Na
Exch. base cation capital	2747	506	258
Weathering from podzol	65	17	16
Net uptake by trees	-59	-22	
Taken out at harvest	-21	-7	
Catchment change (Output - input)	-51	4	-11
% annual change on CE complex (short-term)	-1.6	-0.2	1.9
% annual change on CE complex (long-term)	-0.3	2.8	1.9

The annual short-term catchment budget shows that most (Ca + Mg), (3.1%) are being lost from the exchange sites, K is being lost to a small extent (0.8%), while Na is at equilibrium due to our method of calculating weathering. The annual long-term catchment budget, which only includes the base cation loss in timber transported from the site shows lower losses for (Ca + Mg) and an actual gain for K.

The loss of so much Ca and Mg in the short-term annual catchment budget seems unrealistic, for if one extrapolated this loss linearly forwards or backwards in time, the exchange surfaces would be emptied in 30 years or should have been empty now. Of course one cannot really extrapolate the exchange process linearly, as the lower the base saturation, the better the base cations cling to the exchange sites (Wiklander and Andersson, 1972). K is a cation which tends to accumulate in top soils with time due to biological activity and demand. It also tends to be held strongest by exchange surfaces (Teveldal, 1993). A gain in K is therefore likely.

The till budget shows the same patterns in loss and gains as the catchment budget, except the scale of loss and gain is smaller. Na appears to be accumulating in both short-term and long-term budgets.

The till budget appears more realistic as Ca and Mg supplies would last at least 60 years (short-term) and 400 years (long-term) by linear extrapolation. The gains in Na are unrealistic. Na is the only base cation with a large deposition input and small errors here can cause larger errors in this calculation when the Na exchange capital is so small. Although there can be a temporary gain in Na on exchange surfaces after a seastorm (Mulder et al., 1990), this should be leached out fairly quickly as Na has the lowest selectivity coefficient of the 4 base cations (Teveldal, 1993).

An uncertainty analysis of the catchment budget is useful as there are potential errors in each line of the budget. The net tree uptake calculation has many potential errors as already mentioned and is larger than what has previously been guessed for the Birkenes catchment (Christophersen et al., 1990c). The maximum and minimum uptake values for Ca, Mg and K are used, shown at the end of Results of Biomass Increment section. The weathering uncertainty is set to +/- 20%. The base cation capital at the moment is near a minimum figure. Although it is the top 20 cm of bog that most water travel through, in drier periods EMMA demonstrates that deeper bog layers are contributors to the stream. On the surface material map, podzols may be under-represented and thin humus over represented. A maximum capital is then calculated for 60 cm deep bog and 20% greater area of podzol, taken from the thin humus area. The output - input budget is calculated on 10 years data, which should smooth out annual extremes. However there are errors in rain measurements (10%), a stream gage error (5%), chemical analysis (5%) and dry deposition calculation (10%). In total we estimate a +/-10% error on this budget. The maximum and minimum % changes of base cations on the exchange surfaces in the catchment are presented in table 21.

Table 21. Maximum and minimum % annual base cation change on soil exchange surfaces in the catchment using errors mentioned in text.

	(Ca +Mg)	K	Na
Maximum, short-term uptake	-5.4	-3.2	-2.6
Maximum, long-term uptake	-2.5	2.7	-2.6
Minimum, short-term uptake	-0.7	1.7	1.8
Minimum, long-term uptake	0	4.8	1.8

The table shows how Ca and Mg with minimum capital and weathering and maximum uptake will decrease both annually and after harvest. However maximum capital and weathering and minimum uptake values cause near to zero change for Ca and Mg on the exchange complexes.

K is only decreasing in the worst scenario, minimum capital and weathering and annual maximum uptake, but increasing in the 3 other situations. Na is decreasing under maximum change and increasing under minimum change by quite similar amounts. In summary, (Ca + Mg) show a tendency to decrease, K to increase and Na to remain the same on the catchment cation exchange complexes in the long-term.

In the catchment budget, we attribute the greatest stripping of exchange complexes to the thin humus soils. These cover over half the catchment and are flushed with the greatest amount of water in the course of a year (their drainable water is calculated replaced about 25 times).

We have only 3 measurements comparing base saturation in 1974 and 1992 on thin humus soils. No real change could be observed. Thin humus soil is in the process of building itself up to a thicker more podzol like soil. Moss and lichens take up base cations from solution flowing around them during precipitation. When these die back and become the organic layer, an important question is how easily are the base cations then leached. A leached layer at the bottom and below the organic layer indicates loss of elements. Bog areas cover only 13% of the catchment, however they have a high base cation capital and great fluxes of water flow through some areas. Such bogs could be another main area of base cation stripping.

As the till exchangeable base cations reflect the nutrients available in the rooting depth of the better forest sites, the base cation supply for these soils looks quite healthy. However forest growing on thin humus will be more exposed to problems with base cation supply in the future.

The Nitrogen budget

Total deposition minus loss with the stream (table 8) show that 171 $\text{mmol}_c\ m^{-2}\ \text{year}^{-1}$ of NO_3 and NH_4 disappear in the catchment. Of this, 78 $\text{mmol}_c\ m^{-2}\ \text{year}^{-1}$ are taken up by the trees. We want to discuss where the remaining 93 $\text{mmol}_c\ m^{-2}\ \text{year}^{-1}$ of N ends up.

Total-N in the discharge began to be measured in 1988. In 1990 the measurement of NH_4 in the discharge was stopped, as concentrations were so low. Rest-N is defined as (Total-N) - (NH_4 + NO_3) in 1988, and as (Total-N) - (NO_3) from 1989 to 1992. This gives a volume weighted annual loss of 16 $\text{mmol}_c\ m^{-2}$ rest-N in the discharge. The input of rest-N to the system through deposition is very low when bulk precipitation is measured daily, as shown in Fig. 9. Growth of algae, droplets of throughfall, pollen and other organic small particles can cause larger concentrations of rest-N in bulk precipitation which is collected over longer time periods. We therefore assume the net loss from the catchment is 16 $\text{mmol}_c\ m^{-2}$ of rest-N and the amount of N we need to account for can be reduced to 77 $\text{mmol}_c\ m^{-2}\ \text{year}^{-1}$.

Nitrogen gas exchange must be estimated and included in the budget. N fixation and denitrification both occur in Birkenes. Very little research has been done on quantifying these processes. In the N budget for Klosterhede (Løfgren, 1991), N fixation is put at < 1 kg $ha^{-1}\ \text{year}^{-1}$ and this number is often used in N critical load calculations. However some studies have found 2 - 5 kg $ha^{-1}\ \text{year}^{-1}$ (Gundersen, 1992). Bog Myrtle (*Myrica gale* L.) growing on bogs can increase N fixation, but at Birkenes this plant is found on only a small part of the total bog area. We will use 1 kg $ha^{-1}\ \text{year}^{-1}$ for N fixation at Birkenes.

Lysimeter experiments using podzol soil from a region close to Birkenes show that 0.04-1.5 kg N_2O-N $ha^{-1}\ \text{year}^{-1}$ escape from soil, mainly during the nitrification process (Situala, Bakken and Abrahamsen, 1993). In the UK a sitka spruce plantation on peat soil lost 3.2 kg $ha^{-1}\ \text{year}^{-1}$ by denitrification (Ineson et al., 1991), and a poorly drained oak-beech forest in the Netherlands lost 20 kg $ha^{-1}\ \text{year}^{-1}$ (Tietema et al., 1992). We put denitrification at 5 kg $ha^{-1}\ \text{year}^{-1}$ for bogs, 1.25 kg $ha^{-1}\ \text{year}^{-1}$ for till and thin humus soils and 0 kg $ha^{-1}\ \text{year}^{-1}$ for bare rock. This gives a total N gas loss of 1.6 kg $ha^{-1}\ \text{year}^{-1}$. The total output - input gas balance is 0.6 kg $ha^{-1}\ \text{year}^{-1}$ or 4.3 $\text{mmol}_c\ m^{-2}\ \text{year}^{-1}$.

The tree canopies are the first interactive surface the total deposition hits. The Net Canopy Exchange (NCE) which is (Throughfall + Stem flow) - Total deposition, is negative for inorganic N for the one tree calculation at Birkenes, meaning it is either taken up by the canopy surfaces or lost to the atmosphere. Results from mainly American sites (Johnson and Lindberg, 1992) show negative NCEs for inorganic N and a regression analysis suggested that 40% of inorganic N is taken up by the canopy surfaces or lost to the atmosphere. For all sites measured, rest-N had a positive NCE. A regression of total N deposition versus total N out suggested that 16% of total N is taken up by the canopy surfaces or lost to the atmosphere. However there seemed to be a trend with Spruce sites to have greater canopy uptake in response to greater N deposition. This could be due to some spruce characteristic that aided N uptake in needles and /or due to epiphytic lichens/algae, which have high biomass on spruce. Epiphytic coatings on Norway spruce needles in Norway were found to be positively correlated to the wet deposition of N and to mean annual temperature (Thomsen, 1992). Lichen hanging from branches could also take up a lot of N (Lang et al., 1976). An experiment on branches from Nordmoen with and without lichen showed that four times as much NO_3^- and 10 times as much NH_4^+ was taken up on lichen covered branches compared to branches without lichen (Eldhuset and Røsberg, in prep.). In the large spruce calculation (Table 11), 97% of NH_4^+ and 92% of NO_3^- disappeared in the canopy. However rest-N was not measured and measurements were taken during the growing season. If we say that 15% of total N deposition is lost in canopy during the year, this leaves 158 $mmol_c\ m^{-2}$ still reaching the forest floor. We attribute 90% of N loss to lichen and algae uptake and 10% to direct uptake by needles.

Litterfall from old spruce on a good quality site in Birkenes has an N content of 29.6 kg ha^{-1} $year^{-1}$. If we estimate that the average N content of litter for the whole of Birkenes is 12.5 kg ha^{-1} $year^{-1}$ or 89 $mmol_c\ m^{-2}$, how much of this litter will mineralise, giving NH_4^+ and NO_3^- that will actually leach out of the soil to the stream? As there is such a large N pool in the soil already, it is very hard to register the slight increase in this pool which maybe quite large in terms of the annual disappearance of N in the catchment. Fertilisation experiments can show high accumulation rates, while long-term natural studies show low N accumulation (Gundersen, 1992). If we hypothesise that 50% of the average N litterfall is mineralised and leached per year, 44.5 $mmol_c\ m^{-2}$ would remain in the soil. The N mineralisation is more likely to be from older decomposed litter as new litter often immobilises N during the first 2 years (Prescott et al., 1993).

Field vegetation in the catchment varies greatly. Bogs are dominated with sedges and moss, thin humus by moss and grass, and till deposits by moss, grass, ferns and bilberry (*Vaccinium myrtillus* (L.)). A dense Norway spruce stand had a field layer that took up < 1 kg N ha^{-1} $year^{-1}$ (Nihlgård, 1972). A birch study in Finland (Malkönen, 1977) showed that mosses can bind 1.5 kg N ha^{-1} $year^{-1}$ while grasses and herbs can bind 6 and 8 kg N ha^{-1} $year^{-1}$. Clearly a lot of the field layer is produced annually, but how long does it take for the N to be released? If we hypothesise that bogs have 6, thin humus 1 and till 2 kgN ha^{-1} $year^{-1}$ bound in the field layer and 75 % decomposes a year, then catchment soils retain 3.2 $mmol_c\ m^{-2}$.

Algae both in the stream and in other moist areas of the catchments will take up nitrogen, especially during summer when water is at a standstill in the drainage ditches and streams. Phosphorus is the normal limiting nutrient for algae growth, so some N will always remain in the solution.

Table 22 shows both the calculated nitrogen budget based on Birkenes data previously mentioned in this section and a hypothetical nitrogen budget, which is a guestimate of how the unaccounted for nitrogen in the calculated budget might be used in the catchment.

Table 22. Nitrogen budgets for the Birkenes catchment.

Calculated budget	kg ha^{-1} $year^{-1}$	$mmol_c\ m^{-2}$ $year^{-1}$
Input - output	24	171
Net tree uptake	10.9 (0.4 direct uptake in canopy)	78 (2.8 direct uptake in canopy)
Net rest-N flux	2.2	16
Denitrification - N fixation	0.6	4.3
Unaccounted for (soil/living organic matter)	10.2	72.7
Hypothesised budget		
tree litter increase	6.2	44.5
field layer litter increase	0.4	3.2
lichen/algae in trees	3.5	25
Algae/organisms in stream	2.5 (taken up from mineralised litter N)	18 (taken up from mineralised litter N)

A 3 year study on N additions to a red pine forest in Massachusetts, USA suggested that the soil organic matter pool was the main sink for added N (Aber et al., 1993). 67% of added N could not be accounted for after measuring all other potential N sinks.

Water types

So far in this article we have been looking at data mainly on an annual time step. It is perhaps useful to remind ourselves and others how water types can vary through the catchment on an event basis.

Fig. 8 illustrates the composition of different water types collected shortly after a small summer shower following a long period without rain. The different water types are:

1. Rainwater.
2. Surface water collected on the bedrock surface near the summit.
3. Surface water which has flowed a short distance (< 20 m) on a bedrock surface covered with a thin humic layer.
4. Ground water in the Langemyrdalen bog.
5. Water in the brook in the S.W. part of Langemyrdalen.
6. Water in the brook entering Vestre Tveitdalen.
7. Water leaving the dam in Vestre Tveitdalen.

The figure illustrates how the water composition changed instantly when the rain hit the bedrock surface. This drastic increase in ionic strength must be due to dry deposits adsorbed on the lichens covering the bedrock surface and organic acids secreted by the lichens during the dry period, which are dissolved in a small amount of precipitation. Abrahamsen et al. (1979) also found that conductivity could increase on similar bedrock dependent on previous weather conditions and rain quality and they suggested this could be due to exudates or debris from the lichens and dry atmospheric deposits.

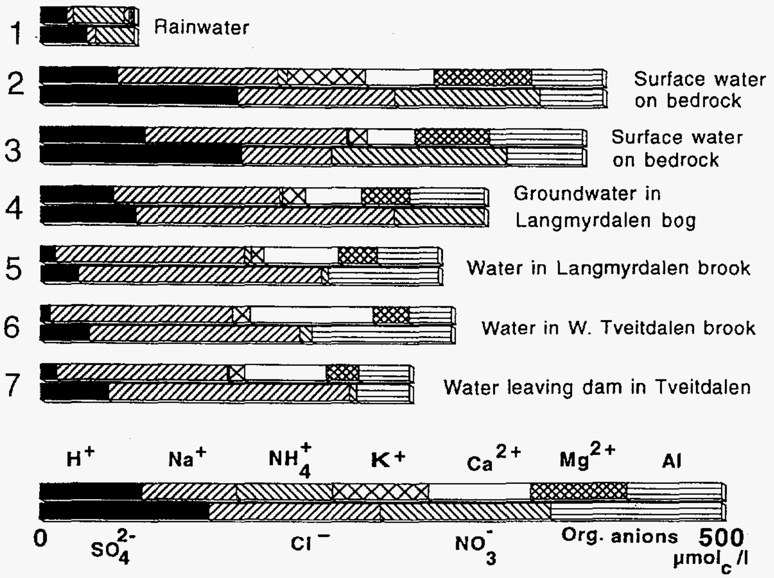

Fig. 8. The chemical composition of different solutions collected after a summer shower 1993 at Birkenes.

Brække (1981a) shows mean values of water types from different environments at Storgama. Barren bedrock had the highest concentrations of SO_4^{2-}, NO_3^-, NH_4^+ and has reasonable concentrations of base cations and Al^{3+} too. Cationic exchange and weathering will not increase the ionic strength but only change the water composition. On its way through the watershed the concentration is gradually decreasing due to dilution with water with lower ionic strength. This is also caused by the pattern of ionic strength in a rain shower, which is usually high in the first minutes and then drops exponentially. Even if water types 2 and 3 have a high proton content the ratios between base cations and protons will maintain a certain base saturation.

The Si content in different water types, illustrated in Fig. 8 are shown in table 23.

Table 23. Content of Si in water samples.

Sample	1	2	3	4	5	6	7
Si (mg/l)	0	1.1	2.6	0.94	1.9	2.6	0.14

Si reaches a high content in sample 3 after flowing 20m on a bedrock surface covered with thin humus. This could be due to Si weathering of the bedrock and Si leaching from the humus material. The very low Si content in the water leaving the watershed (sample 7) must be a result of diatom production in the almost stagnant water behind the dam during the warm period.

It is interesting to look in more detail at the nitrogen contents of the water types during this light shower. Fig. 9 shows the concentrations of NH_4^+, NO_3^-, Rest-N and Cl^- in the same water types as fig. 8 supplemented with 5 new water sample measurements.

Although we are unable to calculate fluxes, it is interesting to note the great variation in quantities and forms of N in the different water samples.

The precipitation has a low content of Cl^- which reflects a small contribution from sea salts. It contains only NH_4^+ and NO_3^- and no Rest-N. Total amount with nitrogen is 66.4 $\mu mol l^{-1}$.

The water layer flowing on the bedrock has a nitrogen content three times higher than the precipitation. Since it was sampled during the rainstorm, transpiration can only explain a little of the concentration rise. A major part of the NH_4^+ has disappeared while the amount of NO_3^- tripled and much organic nitrogen has dissolved in the water. After flowing another 10 m partly in contact with a thin humus layer all NH_4^+ has disappeared and the amount of NO_3^- has increased. Our explanation is that during the long dry period prior to this rain shower there was a dry deposition of sea salts as well as other components. This explains the high content of Cl^- in the water flowing on the bedrock surface. The high nitrate content can either be due to dry deposition or nitrification on exposed surfaces where the humus is heated in the sun. The removal of ammonium can be due to ionic exchange (and later consumption), biological uptake or oxidation to nitrate.

A young pine and spruce tree, both 2.5 m tall and free of lichen growth were selected for throughfall measurements. Water lying on the branches was collected in clean plastic pans by shaking the branches gently. The Cl^- content is 5 and 7 times higher than in precipitation due to previous dry deposition. The contents of NH_4^+ and NO_3^- are nearly the same as in precipitation on the spruce but somewhat higher in the throughfall from the pine. The most noticeable feature is the release of large amounts of rest-N which probably is organic components containing nitrogen (Alenäs and Skärby, 1988). Since the throughfall from spruce has nearly the same concentrations of NH_4^+ and NO_3^- as the rain this means that previously deposited N-components (dry deposition) were consumed during the dry period.

The ground water in both bogs contain large amounts of NO_3^-, some rest-N and small amounts of NH_4^+, while the water in the brooks contain mostly rest-N and the total content of nitrogen is smaller than in other water types except precipitation. This indicates that at this point in time, nitrification was an important process in the top layers of the bogs. Brække (1981a) however found very little NO_3^- in the ground water of peat from February to December. Ammonium and nitrate is effectively removed from the water in the brooks and only rest-N is leaving the watershed during periods with low discharge.

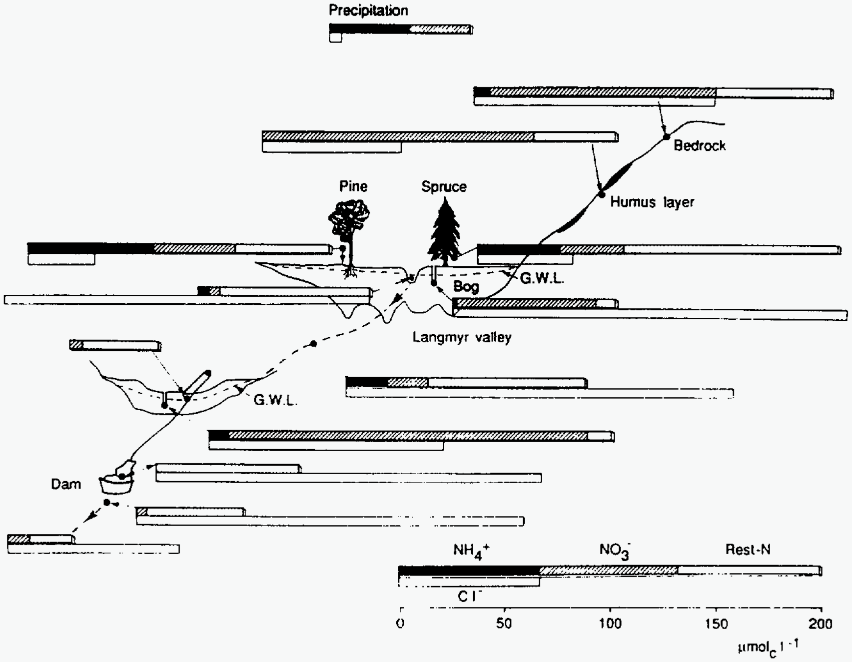

Fig. 9. The Nitrogen and chloride concentrations of different solutions collected after a summer shower 1993.

Discussion

The stream water shows for the period 1972 -1987 a steady decline in Ca (1.25 mmol$_c$ l^{-1} year^{-1}) and Mg (0.7 mmol$_c$ l^{-1} year^{-1}) (Christophersen et al., 1990b). This decline in percent is 2.27 % for Ca and 2.06 % for Mg. The B/C horizon soil water (1984-1987) also showed a decreasing trend for Ca and Mg, although Al and H were about constant. This was interpreted as due to a decreasing base saturation, assuming that all input fluxes (deposition and decomposition of organic matter), weathering and uptake fluxes remained constant.

At the time of writing of this article (Christophersen et al., 1990b), the SO_4^{2-} concentration had decreased only during the last two years (1986, 1987). The authors therefore ascribed the decline in Ca and Mg as due to soil acidification. However this trend in SO_4^{2-} decline has continued. The yearly average stream concentration of SO_4^{2-} (1991/1992) was significantly lower than the yearly average for the whole period (1974/75-1991/92), as were the concentrations of Ca and Mg (Statens forurensingstilsyn, 1993). The E horizon water from lysimeters situated to the South of the main stream also show a decreasing trend in Ca and Mg for this more recent period (1989-1992).

This decline in SO_4^{2-} mobile anions must either be compensated by an increase in concentration of other mobile anions or the concentrations of cations must decrease. This is according to the salt effect and the cation exchange theory (Reuss and Johnson, 1986). The results of the RAIN project at Risdalsheia, which can be compared to small patch of the top slopes at the Birkenes catchment, partly followed these theories (Wright et al., 1993). With "clean" rain applied for an 8 year period, SO_4^{2-} concentrations decreased gradually down to a low level and non marine base cations also declined. However concentrations of organic anions increased, even if the TOC concentrations remained roughly the same. This dissociation of organic anions in the soil and soil solution buffered any rise in pH, which only increased slightly from 4 to 4.2, again causing only modest decreases in inorganic labile Al concentrations. If one looks at the trends in H^+ and Al_i (biologically toxic Al) in the Birkenes stream, H^+ had a high concentration in 1991/92 but was not significantly different from the yearly average in the period (1974/75-1991/92). Al_i was at the highest measured concentration in 1991/92 and was significantly different from the yearly average in the period previously mentioned. TOC was at the highest concentration ever recorded in 1991/92 and the organic anion concentration calculated by charge balance was the highest recorded since 1975. These results to some extent mirror in a milder form what occurred in the RAIN project.

Even if the decline in base cations in the last 4 years is due partly to a decrease in mobile anions, our base cation budget is based on input-output data from 1980-1990. The soil and tree data used are more recent, but the soil section shows there should have been no great changes in base cation capital and tree uptake changes little over a 5 year period. For this earlier period, we also ascribe the reduction of Ca and Mg in the stream mainly to a reduction in base saturation with a corresponding increase in the exchangeable acidity.

If one thinks of the whole catchment's soils as one exchange pool and assumes that selectivity coefficients and the average incoming concentration to a soil remain constant over the years, then the 2% decrease in Ca and Mg in outgoing stream concentration should be directly comparable to a % decrease on the exchange sites. The short-term catchment base cation budget estimated a 3 % loss in Ca and Mg, which we think is a little high (as discussed previously) but compares favourably with the 2% decrease.

The podzol data compared here and previous comparisons of podzol properties over time have shown no real decreasing trend in base saturation. As already discussed, either the technical difficulties of soil resampling are too great to see a decrease or in fact there has been no real decrease. The fact that according to the base cation budget, Ca and Mg are decreasing, while Na and K are increasing would also keep base saturation stable. Also sea salt episodes have been more common at Birkenes in the period (1985/86-1991/92) than in the period (1974/75-1985/86) (Statens forurensingstilsyn, 1993). The decreasing base cation trends in soil solution from mineral horizons of the podzols, apart from being due to decreased base saturation in these soils, could also be due to decreased decomposition of organic matter or caused by trees cycling Ca and Mg more tightly.

The base saturation of thin humus soils could have decreased. The drainable water (max. saturation - field capacity) is replaced at least 25 times in the course of a year and Teveldal (1993) showed how soil columns receiving most rain were the first to attain a reduced and more stable base saturation. Unfortunately there are no time series of thin humus soils at Birkenes to test this theory.

The base saturation in the bog soils could also have decreased. H^+ and Al^{3+} from upslope mineral layers and surfaces may have replaced base cations on the exchange sites. The comparison of Langemyrdalen and Tveitdalen top layer bog samples showed very low base saturation in Tveitdalen, where large water fluxes traverse the bog to enter the stream. A resamping of Langemyrdalen samples is necessary to see if base saturation has dropped here since 1974.

As previously mentioned in the water pathway section, water flow paths are of the utmost importance for the resulting stream water chemistry. In the same section we discussed how EMMA simulates stream discharge as coming from 3 soil end-members. We discussed the water pathways simulated and can agree with EMMA to a certain extent, but feel the B horizon pathway is more likely to be a mixture mainly of sub-surface flow along the bedrock and B horizon water. However our theory on sub-surface/bedrock flow would need to be confirmed by a trench experiment.

If we move over to the soil and stream chemical assumptions in EMMA, one unresolved question is how the soil waters obtain their specific chemical signature. Is the stream water chemistry really a mixture of chemical fingerprints of the most important horizons the water has flown through or does the last soil horizon before the stream determine the stream water chemistry? (Christophersen and Neal, 1990). The last top horizon before the stream is generally bog deposits and at lower levels moraine deposits (Figs.1 and 2). Mulder et al (1991) addressed the problem of modelling stream water as mixtures of forest floor and B horizon water when there is no direct contact between these horizons and the stream. From detailed soil analysis in a small plot in the Southern valley, they found that surface peat exchange sites along the stream were significantly enriched in Al, probably due to return flow of Al-rich B horizon water. This means that the peat exchange composition has developed as a result of upslope soil water flow and has in the past buffered the stream from Al. Now that the surface peat has such a high relative Al saturation and substantial CEC, it can act as a source of Al to the stream. This means that the last soil horizon before the stream is in fact determining the stream chemistry, but its exchange composition has in turn been determined by upslope horizons, which have been included in the Birkenes model.

The proportions of each end member represent the soil layers where the water last had sufficient interaction to obtain its chemical signature (Hooper et al., 1990). For this interpretation to be correct, the source areas in the catchment (where elements are produced), the riparian zone (the stream banks) and the stream itself will only have to act as mixing zones and are thus chemically unimportant relative to the soils. For the 3 catchments analysed thus far by using EMMA, the soil horizons in riparian zones do not seem to have a soilwater chemistry that is distinct from comparable horizons upslope. However all lysimeters used so far are situated on the South East slopes of Tveitdalen and other bog areas could have more effect on water chemistry. Brække (1981a,b) showed that peat soils strongly impacted water during its residence time. Langmyra, the widest and largest bog in the catchment collects about half the catchment's runoff. The soil chemistry in the upper 20 cm must be important for the runoff from this side of the catchment. In rainy/snow melting periods the discharge from this stream could be chemically different from Tveitdalen and could be one of the missing "soil end-members" in the EMMA simulation. Yet in our summer water sampling, Fig.8 the stream water from Langemyra and Vestre Tveitdalen was quite similar in content. This was probably due to similar organisms and stream processes causing the same stream water chemistry in quite stationary water.

Organisms in the stream do affect element concentrations, for example Si can be taken up by algae in the summer. Generally in the summer, EMMA is more doubtful because of biological activity and longer residence time in the stream and stream banks. During periods with greater discharge, it could be more correct. However it was necessary in the second EMMA run (Easthouse et al., 1992) to divide the autumn storm period into 4 periods each with a different end member as the soil end member median concentration varied substantially during this transition from dry to wet conditions.

The nitrogen budget can account for about half of the disappearance of input N as tree uptake. The rest must be used to build up living organisms and the organic matter content of soil as shown in the hypothesised budget. The budget revealed how important rest-N is in both the stream and throughfall measurements. The summer water sample of the dam gives only rest-N and no NO_3^- or NH_4^+. Winter water samples are higher in NO_3^-. Throughfall calculations show that during the summer a lot of inorganic N is taken up, while rest-N is released. During the winter, more NO_3^- and less rest-N reaches the ground. On a yearly basis about 15% of input N disappears in the tree canopy, probably mostly to algae and lichen growth.

Although the S flux input to Birkenes has been decreasing since the mid seventies, the N flux has stayed the same, increasing slightly the last few years because of greater rainfall. The NO_3^- output increased in the stream in 1989 and 1991, while the ground water concentration of NO_3^- increased a lot in 1992. Although the catchment is not Nitrogen saturated in the sense that it is "an ecosystem where nitrogen losses approximate or exceed the inputs of nitrogen" (Ågren and Bosatta, 1988). It is close to an ecosystem where availability of inorganic N is in excess of total plant and microbial nutritional demand (Aber et al., 1989). The trees are certainly receiving more N than they can take up. Being an old forest (Ave. 80 years), dry deposition will increase with the canopy surface area while N demand will decrease with age (Emmett et al., 1993). The microbial N demand is large and algae can grow anywhere moist. However lack of P must limit this growth in the end. We would therefore expect rest-N to increase in the summer and NO_3^- to increase in the winter in the stream, before there develops a steady leaching of NO_3^- year round. A similar Swedish catchment, Gårdsjøen, which has been fertilised with N for 2 years showed increased NO_3^- loss in the winter (Hultberg et al., 1994). A lot depends on mineralisation and nitrification in the soil. Both will increase with increasing temperature and milder winters (global warming) and there is an enormous potential nitrate store sitting in the organic layers (Teveldal, 1993).

In the introduction we emphasised we wanted to look at the uncertainties involved and how these affected the two budgets. The catchment base cation budget was analysed for uncertainty in the various fluxes. Table 21 shows that even with fairly generous error estimates on the fluxes, Ca and Mg still show a tendency to be lost from the exchange surfaces, while K is gained and Na stays at equilibrium in the long-term.

Generally the chemical state of the stream does not look good now or in the future. Even with SO_4^{2-} decreasing, the base cations needed to neutralise are also decreasing. NO_3^- levels look set to rise, which will allow more H^+ and Al forms to leach.

The good forest soils on till deposits may not be leaching base cations as much as previously thought. Certainly after timber harvest, when all the branch and needle biomass is returned to the forest soil, the Ca and Mg loss is small (-2.5- 0%) and K is accumulating. However the thin humus soils are losing base cations. Trees situated in draining furrows may do well, but the general quality class of this soil is sinking. This might mean less economically as these sites are harder to manage and harvest.

Summary

The aim was to collect together all previous and new wet deposition, throughfall, soil solution, stream water, soil, mineral and tree data from the Birkenes catchment. Inputs from dry deposition and weathering and outputs in tree uptake were then calculated using a combination of the above data. The data was integrated to produce three catchment budgets: A 10 year output - input budget of the major elements involving wet and dry deposition and stream water data; A base cation (Na, K, Ca + Mg) budget to see the change on the cation exchange complexes, using the base cation capital, input weathering and output from the output-input budget and tree uptake; A nitrogen budget.

An annual long-term base cation budget, including uncertainty estimates for each of the processes, showed on the catchment exchange complexes that (Ca + Mg) have a tendency to decrease (0 to - 2.5 %), K to increase (+ 2.7 to + 4.8 %) and Na to remain at equilibrium (-2.6 to + 1.8 %). The nitrogen budget showed that of the 24 kg N ha^{-1} $year^{-1}$ that remains in the catchment, 45 % is used in net tree uptake, 9 % leaves in the stream as rest-N, 3 % is lost in net gas exchange and 43 % is unaccounted for and is probably used to build up living/dead organic matter.

The registered annual decline in stream water of Ca and Mg of around 2 % is due to both the decreasing SO_4^{2-} concentration and a decrease of Ca and Mg on the catchment exchange complexes. The podzol data suggests no significant reduction in Ca and Mg with time, while bog and thin humus soil time series data are sparse or lacking for comparison. However thin humus soils, which are the dominant soil type in the catchment, have a high base cation content and their drainable water is replaced most frequently, 25 times annually, which suggests considerable cation stripping could occur here. The few bog soil measurements suggested a drop in base saturation, so bog soils could be another site of cation stripping.

The discussion around hydrology and the use of EMMA concluded that it is a useful stream modelling method, when the stream is not experiencing extreme conditions (drought / extreme storms and rapid swings between the 2 conditions). The B horizon pathway suggested by EMMA is more likely to be a mixture of sub-surface flow along the bedrock and B horizon water, although this needs experimental confirmation. Langemya bog could have a large impact on the water during its residence time and is suggested as a possible missing "soil end-member" in the EMMA simulation.

Conclusion

The chemical state of the stream does not look good now or in the future. Even with SO_4^{2-} decreasing, the base cations needed to neutralise are also decreasing. NO_3^- levels look set to rise, which will allow more H^+ and Al forms to leach.

The good forest soils on till deposits may not be losing as many base cations as previously thought. After timber harvest, the podzol budget shows small Ca and Mg losses (0 - 2.5 %) and K accumulation. However the less fertile and harder to manage forest sites covered in thin humus soils are losing base cations. Trees situated in draining furrows may do well, but the general quality class of this soil is sinking.

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PAPER 2

A Comparison of Nutrient Cycling in 30 and 40 year old Norway spruce plots at Nordmoen, south-east Norway.

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Abstract

The solute fluxes in precipitation, throughfall and soil water in 30 (P30) and 40 (P40) year-old Norway spruce plots at Nordmoen, south-east Norway were compared over a four year period. The total deposition of non-marine ions, the net canopy exchange of most ions and the litterfall fluxes of nitrogen (N), potassium (K^+) and magnesium (Mg^{2+}) were greater in P40 than P30. Higher solute fluxes were observed through the forest floor and aluminium (Al^{3+}) became the dominant cation in soil solution in the mineral horizons in P40 as opposed to calcium (Ca^{2+}) in P30. In both plots, litterfall was the dominant input of Ca^{2+} into the soil, while weathering was the dominant input in the Mg cycle. Throughfall represented the dominant input of K^+ into the soil, and tight biological recycling of K^+ was necessary due to low weathering. In the N cycle, organic N was released from the canopy, while half the inorganic N was retained here and little leached into the mineral horizons. Research is urgently needed to quantify the retention and release of N in the canopy. The spatial distribution of chloride (Cl^-) concentration in soil water and how this affects calculation of soil water budgets is discussed.

1. Introduction

The study of nutrient cycling in different-aged forest stands of the same species has naturally tended to concentrate on element contents in tree biomass components, litterfall and forest floor (Turner, 1980; Grier et al., 1981; Sprugel, 1984). More recently the effects of different aged-stands on chemical changes in the upper layers of soil have been shown (Alriksson and Olsson, 1995; Brais et al., 1995). In Wales, stream nitrate concentrations have been positively correlated with the age of surrounding sitka spruce stands (Emmet et al., 1993; Reynolds et al., 1994). The advent of models like MERLIN (Cosby et al., 1997), which simulate processes affecting nitrogen in forest ecosystems demands reliable input data from a chronosequence of forest stands (Emmet et al., 1997). The interpretation of the long-term effects of acid deposition on forests also demands insight into the changes of nutrient cycling with time (Johnson and Lindberg, 1992; De Vries et al.,1995). The sum of nutrient cycling processes at the different ages of a forest stand thus have far-reaching effects on tree nutrient status, forest soil, surface water and groundwater, affecting in turn decisions on forest and water management.

A theory describing nutrient cycling in optimal stand-development of an even-aged stand has been presented by Miller (1995). As foliage and fine roots are the largest sinks of nutrients in the tree, an understanding of the development of the crown and fine root biomass are essential to follow nutrient dynamics. To achieve early dominance, the trees have maximum production of nutrient-rich leaves, twigs and fine roots up until canopy closure. After canopy closure, the trees compete between themselves and growth is concentrated in stem wood and structural roots, which are less nutrient demanding, while the leaf biomass is simply maintained. Deficiency in water and /or nutrients and events like thinning and insect defoliation can prolong or disrupt the above cycle. Superimposed on this cycle is the effect that the older the tree, the larger its surface area and this often increases the amount of total deposition onto a forest (Horntvedt and Joranger, 1974; Stevens et al., 1992; Beier et al., 1993; Kvindesland et al, 1994). Obtaining a suitable chronosequence of stands to test such theories can be difficult, as differences in soil characteristics related to former land-use and parent material can mask effects caused by nutrient cycling (Marques and Ranger, 1997; Marques et al.,1997). In this respect, the adjacent 30 and 40 year-old Norway spruce (*Picea abies* (L.) Karst.) plots at Nordmoen may serve to study the effects of nutrient cycling because the plots have similar former land use and soil chemical properties.

A detailed 4-year dataset of biogeochemical parameters collected during and after the Integrated Forest Study (IFS) (Johnson and Lindberg, 1992) was used to evaluate ion fluxes of precipitation, throughfall and soil water, comparing the main trends and the dominant processes between two different aged plots. In addition the following sub-objectives were investigated:

1) the spatial distribution of Cl^- concentration in soil water was qualified and how this affected the estimated soil water budgets.
2) estimated weathering rates were tested against field observations.

These sub-objectives have great relevance for the quality of data input to critical load models such as MAGIC (Cosby et al., 1985) and PROFILE (Sverdrup and Warfvinge, 1993).

2. Materials and Methods

2.1 Site description

The study site Nordmoen is located on a flat plain of glaciofluvial deposits 45 km north of the Oslo fjord at an elevation of 200 m. The soil is classified as a Cambic Arenosol (FAO) (Stuanes and Sveistrup, 1979). The plant community is classified as EuPiceetum (Kl.62) (Fremstad and Elven, 1987). The dominant tree species is Norway spruce (*Picea abies* (L.) Karst.) with 3% Scots pine (*Pinus sylvestris* (L.)). The dominant species in the field layer are Bilberry, (*Vaccinium myrtillus* (L.)) and Wavy hair-grass (*Deschampsia flexuosa* (L.) Trin.), while different moss species cover the ground layer. More details on vegetation can be found in Røsberg (1991). The Gardermoen meteorological station is situated 10 km south of Nordmoen and the annual standard normals (1961-90) show a mean temperature of 3.8 °C (174 days with temperature below 0 °C) and mean precipitation of 862 mm. The snow pack usually remains from December to April. The average annual total sulfur and nitrogen deposition for 1986-1990 is 9.3 and 10.4 kg ha^{-1} respectively (Statens forurensingstilsyn 1988, 1989, 1990, 1991, 1992).

Two plots, a 30 year old Norway spruce plot (P30) and a 40 year old Norway spruce plot (P40) were monitored for bulk precipitation, throughfall, litterfall and soil water at four different depths over a four year period (Oct. 1986 - Sept. 1990). These plots were established in 1986 as part of the Integrated Forest Study (IFS) programme (Johnson and Lindberg, 1992). They are square plots, 1000 m^2, situated nearly adjacent to each other (see Figure 1). Selected data for trees and soils, measured in 1986 are given in Table I and Table II.

Table I. Stand details of a 40 year-old (P40) and a 30 year-old (P30) Norway spruce stand at Nordmoen, measured in 1986.

Plot	age	basal area (m^2 ha^{-1})	height (m)	density (trees ha^{-1})
P40	40	30.5	13.0	2100
P30	30	19.3	8.1	3240

Table II. Soil chemical properties of plots, P30 and P40 at Nordmoen, mean of two samples collected in 1986.

SITE	HORI -ZON	THICK -NESS	pH [a]	CEC [b]	Base saturation [b]	Exch. Al[b]	$Ca(H_4PO_4)_2$ sol. SO_4	C/N
		cm		$mmol_c kg^{-1}$	%	%	mmol kg^{-1}	
P40	Oi	1.1	4.19	429.5	63.2	2.2	1.73	37
	Oe+a	3.6	3.68	271.2	43.7	16.3	1.45	36
	E	1.1	3.71	100.4	12.3	71.4	0.17	30
	Bs	17.1	4.34	51.6	3.6	93.9	0.35	24
	BC	14.0	4.71	8.5	8.3	91.1	0.86	14
P30	Oi	2.0	4.44	556.4	70.1	1.7	3.07	36
	Oe+a	4.3	3.75	243.9	49.3	10.3	1.79	44
	E	1.8	3.61	89.5	8.4	71.7	0.19	35
	Bs	24.6	4.38	44.9	3.3	93.9	0.53	25
	BC	20.5	4.72	7.7	9.9	88.6	0.60	13

[a] 10ml soil:25 ml water. [b] Extracted with NH_4NO_3.

2.2 Methods

Bulk precipitation was the arithmetic mean of 3 measurement sources (Figure 1); data from the NILU (Norwegian Institute for Air Research) monitoring station at Nordmoen (Statens forurensingstilsyn, 1996), data from the 3 rain collectors on one stand placed in open terrain from the Nordmoen permanent research plot of the Norwegian Monitoring Programme (OPS) (Aamlid, 1992) and data collected for the IFS project in 2 standard NILU collectors (Statens forurensingstilsyn, 1996). In the winter, snow samples were collected in a plexiglas cylinder from a 0.5 x 0.5 m snowboard and volume was registered by collection in a plastic bucket.

Twenty throughfall and litterfall collectors were placed beside each other in fixed positions in both plots according to a randomised design. The throughfall sampling bottles were placed in the ground in holes lined with pieces of black tubing, keeping the solution dark and cool. The collection area of the bottles was 109.36 cm^2. The 20 collectors were pooled according to a fixed random design into four groups (five collectors in a group) per plot and the water chemistry was analysed on a monthly basis during the snow-free period. During the winter, snow throughfall samples were collected on similar snowboards to those used for bulk precipitation. Also two snow lysimeters, constructed of cylindrical polyethylene containers were placed in each plot to collect meltwater. After winter 1986-1987, five plastic buckets were in addition placed randomly in each plot to collect snow throughfall. The litterfall was collected in square plastic trays, which each had a collection area of 1140 cm^2. The walls and bottom of the tray were drilled for drainage and a meshed cloth hanging over the tray caught the litter. Litterfall mass was collected and sorted monthly during the snow-free period. For chemical analysis, the litter was divided into needle, lichen and a rest litter component (twigs, bark, cones, seeds) and bulked to produce one composite summer and one composite winter sample. Stemflow was collected, but the fluxes were negligible (Johnson and Lindberg, 1992) and so are not included here.

Lysimeters (fritted glass, 4-5.5 µm pore size, made from modified Buchner funnels) were installed beneath the forest floor (O), the E horizon, and Bs horizon and within the upper BC horizon. A tension of 10 KPa was supplied continually to each plate bottle collection system by a central vacuum pump. The lysimeters were in use from October 1986. There are two pits in each of the P30 and P40 plot with two lysimeter replicates in each pit, although the samples were bulked to one composite sample per pit horizon for chemical analysis. Soil water chemistry was analysed on a monthly basis during the snow-free period and as often as physically possible during the winter. The water, soil and plant samples were stored, prepared and analysed as described in Ogner et al., (1991). Briefly in water, Cl^-, SO_4^{2-} and NO_3^- are determined by ion chromatography, ammonium by a modified flow injection method and total-N is determined as nitrate after digestion with persulfate. Other elements are determined by ICP simultaneously. In both plants and soil analysis, total-N is determined as ammonium after Kjeldahl digestion and total elements (Al, Ca, K, Mg, Mn, Na) are determined by ICP in acid digested samples. Exchangeable cations of soil are extracted in NH_4NO_3 and determined by ICP.

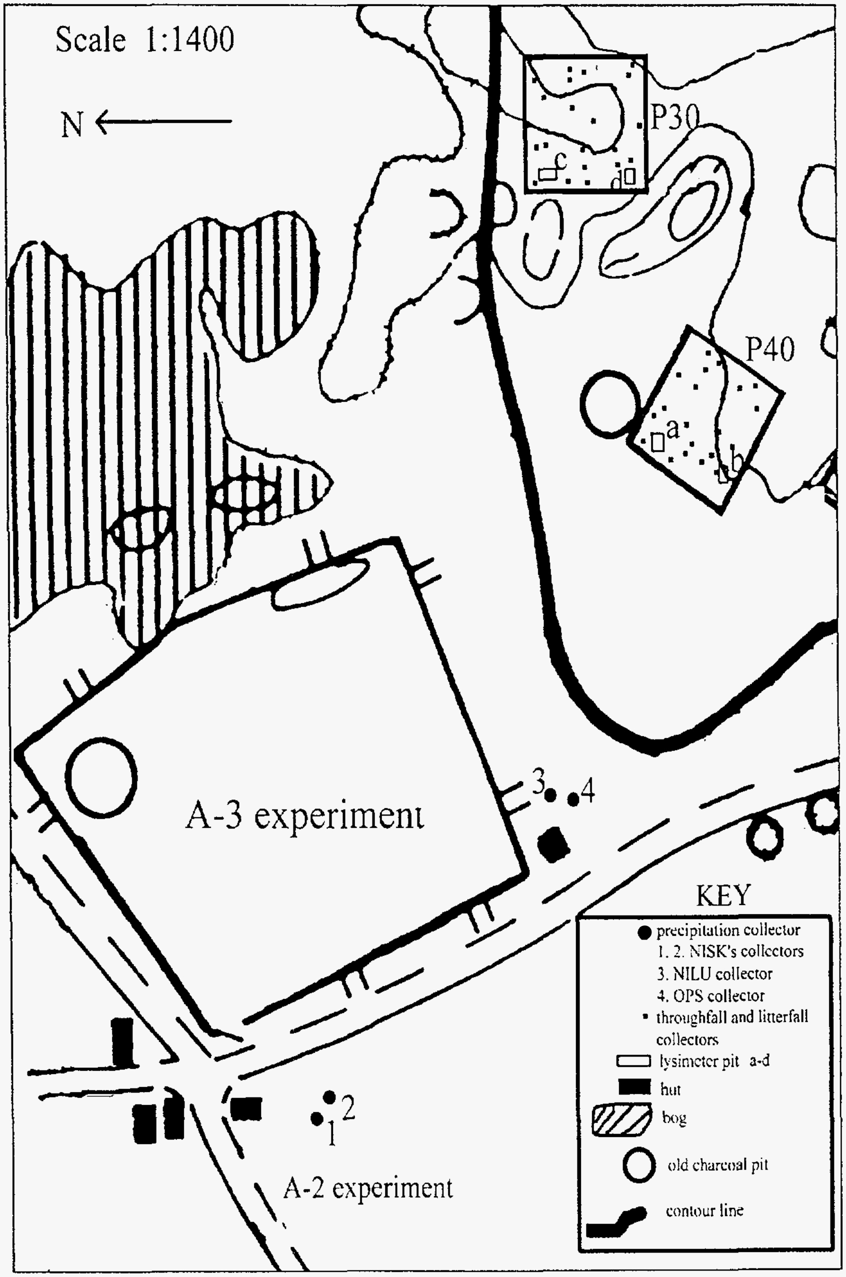


Fig 1. Map of Nordmoen showing P30 and P40 plots and location of bulk precipitation, throughfall and litterfall collectors and lysimeter pits.

2.3 Calculations

The water fluxes were calculated on a four year basis (1986-1990) by assuming that Cl^- is conservative in the soil. In equation form, where thr denotes throughfall and l denotes the soil horizon (l = O, E, Bs and BC):

$$Clflux_{thr} = volume_{thr} \times [Cl]_{thr} = Clflux_{l} = volume_{l} \times [Cl]_{l} \qquad [1]$$

The annual dry deposition was calculated by using the throughfall/wet deposition ratios for sodium (Na^+) and sulfate (SO_4^{2-}) as references. The method is quite similar to that used at Solling (Ulrich, 1983; Matzner, 1989; Bredemeier et al., 1988). Sodium and sulfate are assumed conservative through the canopy (i.e. they are therefore assumed to be neither taken up or exuded by needles and other flora.) All Na^+ is assumed to derive from seawater. The wet deposition of all other elements is first split into a marine and non-marine component. The Na^+ dry deposition flux is used to calculate the marine dry deposition, by using the ratio of Na^+ to other elements in sea water. The non-marine dry deposition flux of SO_4^{2-} is then found by subtraction. The ratio of SO_4^{2-} to other elements in non-marine wet deposition is assumed to be the same as in non-marine dry deposition. Using these ratios and the non-marine dry deposition flux of SO_4^{2-}, the non-marine dry deposition flux of other elements are found. The calculation of dry deposition of K^+ is shown below as an example, where thr denotes throughfall, wd denotes wet deposition, dd denotes dry deposition, m denotes marine, nm denotes non-marine, ssK/Na denotes seasalt ratio of K^+:Na^+:

$$ddNa^+ = ThrNa^+ - wd\ Na^+ \quad \text{(same for } SO_4^{2-}\text{)} \qquad [2]$$

$$mwdK^+ = wdNa^+ \times ssK/Na \qquad [3]$$

$$nmwdK^+ = wdK^+ - mwdK^+ \qquad [4]$$

$$mddK^+ = ddNa^+ \times ssK/Na \qquad [5]$$

$$nmddK^+ = nmdd\ SO_4^{2-} \times nmwdK^+/nmwd\ SO_4^{2-} \qquad [6]$$

$$ddK^+ = mddK^+ + nmddK^+ \qquad [7]$$

Volume weighted average concentrations were used to calculate the four year throughfall and soil water fluxes.

The organic anion flux was not measured but calculated as the missing anion charge from charge balance (Feger, 1995):

$$\text{organic anion} = (H^+ + Al^{3+} + Ca^{2+} + Mg^{2+} + K^+ + Na^+ + NH_4^+ + Mn^{2+}) - (SO_4^{2-} + NO_3^- + Cl^-). \qquad [8]$$

The flux in litterfall was calculated by multiplying the arithmetic mean of the chemical composition of litterfall (needle and rest litter) by the annual mean litterfall mass for the 4-year study period.

To calculate the gross nutrient uptake in above-ground biomass, data on tree growth and tree chemical properties from 1986-1988 in the two plots were used (Johnson and Lindberg, 1992):

$$\text{Gross nutrient uptake} = \text{foliar litterfall} + \text{requirement for wood} + \text{NCE} \qquad [9]$$

where,

$$\text{Net Canopy Exchange (NCE)} = \text{Throughfall} - \text{Total deposition} \qquad [10]$$

(positive values mean release from the canopy and negative values, retention)

3. Results and Discussion

3.1 Water fluxes

The estimated annual water fluxes for P30 and P40 were very similar (Table III). Although canopy interception was slightly larger in P40 than P30, evapotranspiration (E) was slightly lower in P40 than P30. As often interception and evapotranspiration are found to be larger under older and more widely-spaced trees (Kittridge, 1948), this is a surprising result. The general level of canopy interception, 26 to 27% compares favourably with other measurements of interception in Norwegian spruce forest plots (Aamlid, 1992;1993). The low throughfall volumes collected in the winter 1987-1988 caused lower annual SO_4^{2-} throughfall fluxes than SO_4^{2-} wet deposition fluxes, which ruled out total deposition calculations (see 2.3 and Figure 2) on both plots for this year. This suggests that throughfall collection of snow and rain was inadequate in a winter with frequent thawing and refreezing, and that not all the sample water was collected.

Table III. Estimated annual water fluxes for the plots, P30 and P40 at Nordmoen over the 4-year period (Oct. 1986 - Sept. 1990).

Plot	Precipitation (P) (mm)	Canopy Interception, % of P	Throughfall (T) (mm)	Estimated water flux at bottom of BC horizon, (D)[1] (mm)	Evapotranspiration E = T-D (mm)	Total evapotranspiration, (ET) % of P ET = (P-D)/(P x100)
P40	930	27	675	351 (435)[2]	324 (240)	62 (53)
P30	930	26	690	343 (421)	347 (269)	63 (55)

[1] D was calculated according to equation 1.

[2] The values in brackets are calculated under the assumption that 14.1µmol $l^{-1}$of Cl^- is due to mineralisation of OM (Teveldal, 1993), ie

$$Clflux_{thr} = volume_{thr} \times [Cl]_{thr} = Clflux_{BC} = volume_{BC} \times ([Cl]_{BC} - 14.1)$$

The total annual evapotranspiration (ET) for the two plots, 62 to 63 % or 579 to 587 mm can be compared to the value of 50.4% or 400 mm found by water balance methods in the Romerike area (Jørgensen and Østmo, 1990). This area includes both lakes, agricultural land, towns etc., so it is reasonable to assume that ET will be greater than 50 % for a purely forested area. Average pan evaporation (1968 - 1974) measured from May- September in the same district by Otnes (1973, 1975) was 412 mm. Possibly our ET values are overestimated because the flux of drainage water under the BC horizon (D), calculated based on measured Cl^- concentrations (equation 1), was estimated too low.

Manderscheid et al. (1995) suggest that 4 to 5 years of soil water Cl^- concentration is a minimum for calculating water fluxes at Solling. So over a 4 year period, it should have been possible to calculate the water fluxes from layer to layer at Nordmoen based on the Cl^- concentrations. However the average Cl^- concentrations for a 4-year period at Nordmoen (Table IV) do not increase smoothly down the soil profile. Thus, estimating water fluxes at Nordmoen based on measured Cl^- concentrations in soil water and assuming that Cl^- is inert involves a large amount of uncertainty.

Table IV. Average volume weighted Cl^- concentrations ($\mu mol_c l^{-1}$) in different horizons in the P40 and P30 stand at Nordmoen as measured between 1986 -1990.

	P40	P30
throughfall	38	38
O	94	55
E	55	46
Bs	112	47
BC	74	76

Commonly Cl^- has been assumed to be conservative in the soil and only a few studies raise the issue of Cl^- cycling (Peters, 1991). Teveldal (1993) found evidence of a small increase in Cl^- (14.1 μmol l^{-1}) in the forest floor at Nordmoen which he attributed to mineralisation. Chloride increased in stream water after harvesting in a catchment (Haveraaen, 1981). Chloride concentration in stream water also increased strongly in a catchment after forest fire relative to the control catchment (Nygård,1997). This suggests that small amounts of Cl^- are mineralised in the forest floor and washed out. Including a contribution from Cl^- mineralisation gives a more realistic ET of 54 % (Table III).

The observed variability of Cl^- concentration in the soil (Table V) may also be due to spatial variation in water flow. Root channels may disturb the flow of water, quickly transporting relatively dilute water into the soil. Fine roots, on the other hand may take up water and concentrate the solution, so the lysimeter under would receive relatively little and concentrated water. This may be the explanation for the large concentration of Cl^- found under the Bs horizon of pit 2, P40 (Table V).

Table V. *The mean chloride concentration ($\mu mol_c l^{-1}$), [Cl^-], in the throughfall composite samples and in the soilwater from the two pits in plots P30 and P40 over the four year period (Oct.1986 - Oct 1990). The relative distance* of the lysimeter to the nearest tree trunk is also shown.*

	P30				P40			
throughfall composite sample (4 per plot)	1	2	3	4	1	2	3	4
[Cl^-]	29	47	60	39	55	44	45	36
soil water sample	pit 1 [Cl^-]	relative distance*	pit 2 [Cl^-]	relative distance	pit 1 [Cl^-]	relative distance	pit 2 [Cl^-]	relative distance
O	52	1.5	40	1.8	82	1.3	65	3.3
E	52	1.8	35	3.0	45	2.0	59	2.3
Bs	71	1.4	29	1.7	71	0.5	197	1.5
BC	71	1.5	83	2.3	71	0.9	56	1.0

The spatial variability of Cl^- in throughfall may also cause considerable spatial variation in Cl^- concentrations in soil water. At Norway spruce sites near the sea, Cl^- concentration in throughfall was shown to increase as distance to stem decreased (Beier et al., 1993; Gundersen et al., 1995). At an older spruce site, 500km inland, Cl^- concentration in throughfall was not dependent on distance from the stem, but was highly variable in input (Seiler & Matzner, 1995). At Nordmoen the mean of four composite throughfall samples for both plots show a range in Cl^- concentration of 29-60 $\mu mol_c l^{-1}$, but no data exists to test the relationship between Cl^- concentration in individual throughfall samples and distance to the tree trunk. The average relative distance of the two lysimeters in each pit to their nearest tree trunk can be compared to the mean chloride concentration for each lysimeter (Table V). For the O horizon, the greater the distance from the nearest trunk, the lower the Cl^- concentration for each plot. For the mineral layers, this pattern of lower Cl^- concentration with greater distance from the trunk can be found in one plot but the opposite pattern is found in the other plot. This suggests that spatial variation of throughfall flux has the greatest influence in the forest floor and less influence in the mineral layers.

Future field investigations, where the aim is to quantify and model solute fluxes in throughfall and down the soil profile should plan carefully the type and placement of throughfall collectors and lysimeters in relation to distance from trees. We highly recommend measuring water tension or moisture contents for calibration of modelled soil water fluxes and not to rely on the Cl^- concentration in soil water.

3.2 Total Deposition and Net Canopy Exchange

On the basis of the dry deposition data, computed according to equations 2-7, the total deposition and Net Canopy Exchange (NCE) (equation 10) for the P40 and P30 plots (average of 3 years) were calculated (Figure 2).

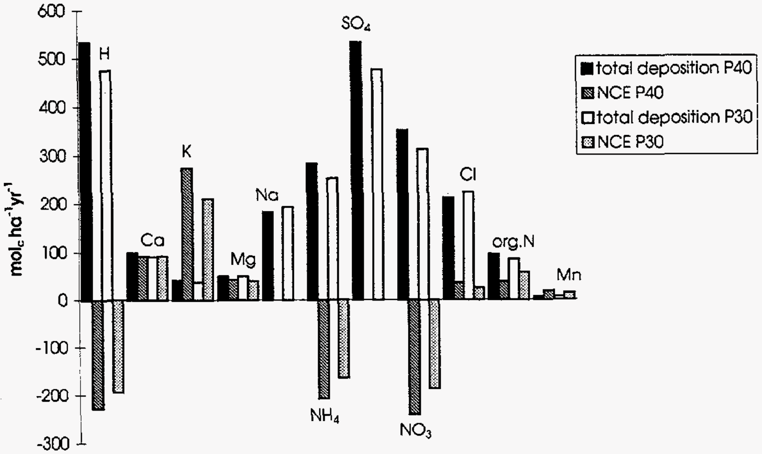


Fig.2. Total deposition and Net Canopy Exchange for the P30 and P40 plots (Average of three years; Oct.1986 - Sept. 1987, Oct.1988 - Sept. 1989, Oct.1989 - Sept. 1990).

Total deposition is dominated by hydrogen (H^+), SO_4^{2-}, ammonium (NH_4^+), nitrate (NO_3^-), Na^+ and Cl^-. The estimated seasalt deposition is similar in both plots. Deposition of H^+, SO_4^{2-}, NH_4^+ and NO_3^- is greatest in P40. The NCE of P40 is greater than P30 for all ions except Ca^{2+} and organic N. Both canopies retain NO_3^-, NH_4^+ and H^+ , while K^+, Mg^{2+}, Ca^{2+}, manganese (Mn^{2+}), Cl^- and organic N are leached. Both Na^+ and SO_4^{2-} are assumed conservative through the canopy, (i.e. NCE = 0). The results also show how foliar exudation of both K^+ and Mn^{2+} is higher than total deposition.

3.3 Solute fluxes at the two plots

Figure 3. shows the annual volume weighted fluxes through the P30 and P40 forest plots.

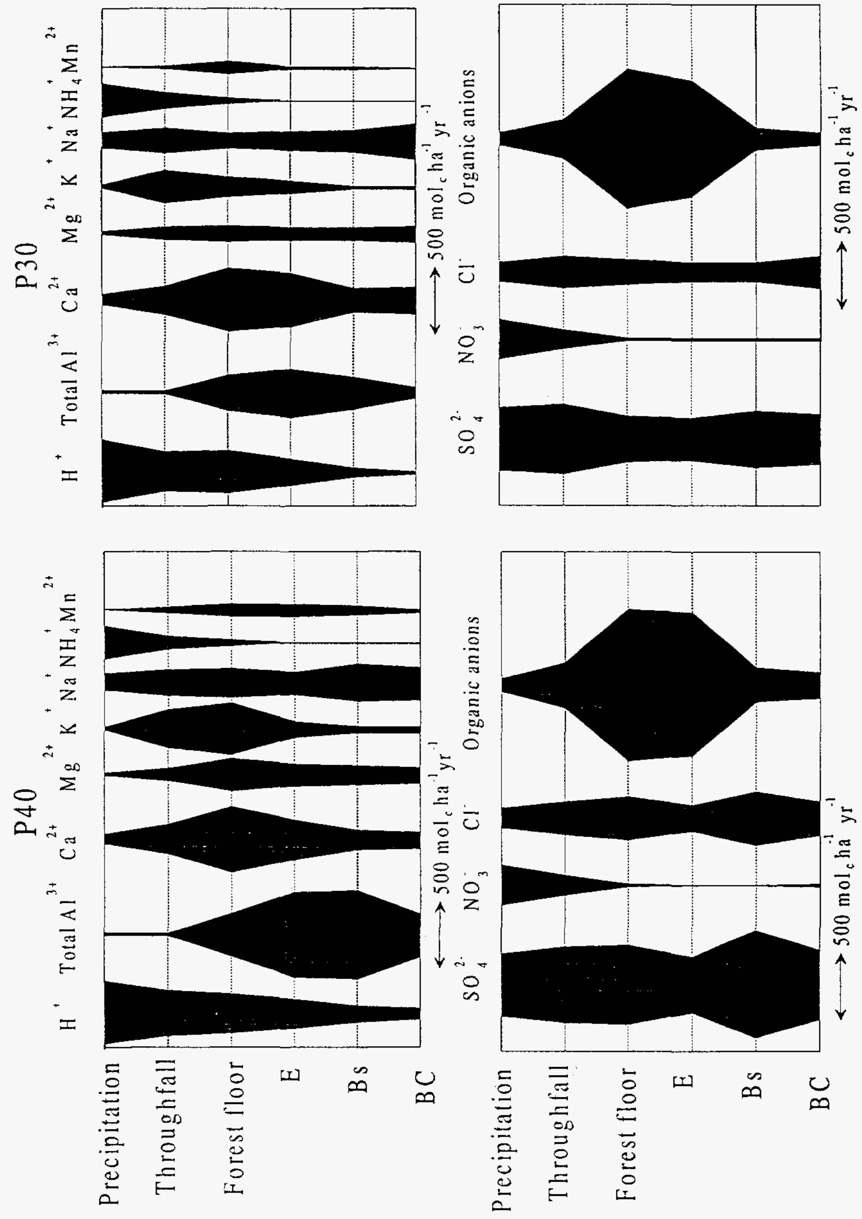


Fig. 3. Average volume weighted fluxes through the P30 and P40 plots, 1986-1990.

Similarities between P30 and P40

Sulphate is the dominant anion from precipitation to BC horizon, except for the forest floor and E horizon where organic anions become the main anion. Both NO_3^- and NH_4^+ show a relatively large input with precipitation, which disappears in the canopy and the forest floor. The dominant cation in precipitation and throughfall is H^+, but its flux diminishes down the soil profile and Ca^{2+} becomes the dominant cation in forest floor. The fluxes of Ca^{2+}, Mg^{2+} and K^+ tend to increase largely from precipitation to forest floor and then decrease down the profile.

Differences between P30 and P40

A comparison of mean soil chemical properties (Table II) shows that the soils have chemical properties within the same range for each horizon in the two plots. Differences in the amount of exchangeable cations would affect the nutrients available for uptake and differences in the C/N ratio might indicate different decomposition activities. Although spatial variation in soil cannot be ruled out, we believe the main differences in nutrient cycling between the plots can be attributed to the difference in tree age and size. The fluxes of most solutes (H^+, K^+, Mg^{2+}, Na^+, Mn^{2+}, SO_4^{2-}, and organic anions) in throughfall and through the forest floor are larger in P40 than P30. In the mineral layers of P40, the fluxes of SO_4^{2-} and organic anions are also larger and total Al^{3+} becomes the dominant cation from the E horizon and downwards in P40. Also Gundersen (1994) found that the higher the spruce tree, the higher the SO_4^{2-}, Cl^- and Al^{3+} concentrations found in soil water under the tree. The trees in P40 are 10 years older, 5 m higher and the total above-ground surface area per tree must be assumed greater than in P30. The total deposition for both H^+ and SO_4^{2-} is greater at P40 probably because of the tree's larger surface area (Figure 2). The net canopy exchange is also generally greater for P40 than P30 (Figure 2). In addition litterfall quantity (Røsberg, 1991) and the amount of Mg, K and N returned in litterfall (Figures 5-7) is greater in P40 than P30. These latter two observations fit in with Miller's hypothesis: The trees in P40 are at a stage in their life cycle, 15 years after canopy closure, when nutrient demand to an increasing degree is met by the biomass cycling fluxes of translocation, throughfall and litterfall. The trees in P30 however are only five years after canopy closure, so demand will to a greater degree be covered by net soil uptake, meaning the biomass cycling fluxes will be relatively smaller (Miller 1986; Miller, 1995). An older and larger tree might be expected to show higher evapotranspiration and hence cause higher concentrations of solutes in soil water compared to a younger tree. However this difference was not seen in the estimated soil water fluxes for the two plots (Table III). In the soil mineral layers at P40, the larger SO_4^{2-},Cl^- and organic anion flux through the profile carries more cations with it, and in contrast to P30, less Ca^{2+} but more Al^{3+}, K^+ and Mn^{2+} leaches out the profile.

3.4 Nutrient Cycling Budgets

The Ca^{2+}, Mg^{2+} , K^+ and N budgets using the water fluxes (Figure 3), total deposition (Figure 2) and rates of weathering (Table VI) are presented in Figures 4 to 7.

Weathering and total deposition are the two 'external' inputs of base cations to a forest ecosystem, while net canopy exchange (NCE) is an internal flux. Total deposition fluxes of Ca^{2+} and Mg^{2+} (Figure 2) are less than estimated weathering fluxes (Table VI), whereas the total deposition flux of K^+ is of the same order of magnitude as the weathering flux of K^+. The deposition flux of Na^+ is much greater than the estimated weathering rates for this cation. Correct weathering fluxes for Ca^{2+}, Mg^{2+} and K^+ are thus very important in nutrient cycling budgets and for modelling critical loads for forest soils.

Weathering fluxes

Different methods produce a wide range of weathering rates as shown for Nordmoen in Table VI.

Table VI. Estimates of weathering rates for Nordmoen as obtained in different studies. All values in $mol_C\ ha^{-1}\ year^{-1}$.

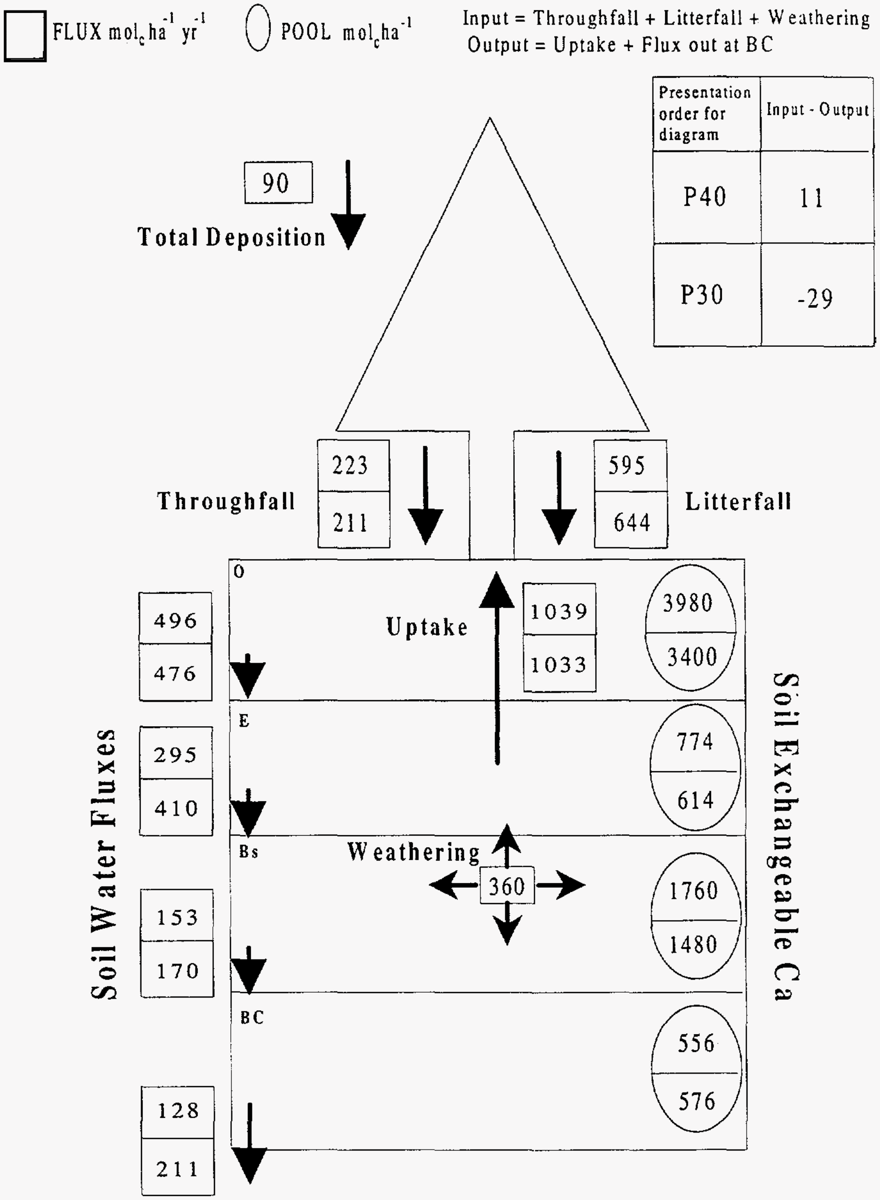
Reference	Method	Ca	Mg	K	Na
Teveldal, 1993; Teveldal and Jørgesen, 1996	6 year lab. soil column study	360*	180*	40*	70
Sogn et al., 1993	6.5 year field soil column study	250	80	0	70
Sogn, 1994	modelling - Magic	80	44	71	54
Wright et al., 1991	modelling - Magic	8	117	40	116
Teveldal et al., 1990	historical rate	18	97	41	34

* used in Figures 4 - 6.

The historical, field and experimental weathering rates were also found to be quite different at Solling (Wesselink et al., 1995). Although experimental rates, extrapolated to field conditions gave similar weathering rates to those calculated in the field by long-term element budgets for K^+, this method failed badly for Mg^{2+}. These authors emphasised the continued importance of long-term field weathering rates found by input-output budgets. We too have most confidence in the long-term weathering rate found in undisturbed soil (Teveldal, 1993; Sogn et al., 1993; Teveldal and Jørgensen, 1996) as even small mechanical disturbances in soil can alter leachate water chemistry quite dramatically (Teveldal, 1993) and grain surface coatings are left complete on minerals (Courchesne et al., 1996). For this reason, we include the most recent estimates of weathering rates from the 6-year soil column study in the base cation cycles (Figures 4 - 6), although other weathering rates are also compared with the field data.

The Calcium cycle (Figure 4)

The flux of Ca^{2+} in litterfall is greater in P30 than P40, despite litter quantity (Røsberg, 1991) and fluxes of other elements (Figures 5-7) being generally greater in P40. In addition, from below the E horizon and downwards, there is a greater Ca^{2+} flux in soil water in P30 than P40. In both plots, the main input of Ca^{2+} to the soil is via litterfall, then from weathering and least comes via throughfall. Ca^{2+} is not translocated in conifers to any degree (Miller, 1995) and accumulates in the needles with time, so the Ca^{2+} content in fresh litterfall needles is much greater than in current year live needles. One possible uptake source for Ca^{2+} , Mg^{2+} and K^{+} not included in these diagrams is the groundwater, which a combination of deep roots and shallow water table at certain places and times of the year allows (Bjor and Huse, 1988). This would naturally reduce the uptake from the soil, but estimating uptake from this source was impossible for this study. It is interesting to compare the weathering flux estimated from the soil column study with the estimated soil water fluxes. As the soil column studies indicate that weathering was found to occur mainly in the upper boundary of the Bs horizon (Teveldal, 1993), one might expect the base cation fluxes in Bs and BC to reflect to some degree this weathering activity, although uptake and cation exchange processes will also affect the Ca^{2+} flux. For both plots the Ca^{2+} flux in the B horizons is on average about half this weathering flux. If one uses the lowest Ca^{2+} weathering rate, 8 $mol_c\,ha^{-1}year^{-1}$, modelled by MAGIC (Table VI), the soil water fluxes in the B horizons are approximately 20 times larger than this weathering flux. Although the weathering rate from the soil column studies seems to fit better with the soil water fluxes observed in Bs and BC, it is the highest weathering rate in Table VI and may have been calculated too large due to mineralisation of litter. Nitrification was very high in the soil columns (Teveldal, 1993) suggesting a lot of decomposition of organic matter. Litterfall is rich in Ca^{2+} and litter studies show Ca^{2+} accumulates in litter before leaching out (Berg and Tamm, 1994; Berg and Cortina, 1995; Johansson and Grälls, 1989). Little is known about the dynamics of Ca^{2+} in humus and soluble organic matter, but organic matter both in the O and Bs horizon could act as a Ca^{2+} source even after 6 years of leaching. The input-output diagram shows whether the soil profile is losing (negative) or gaining (positive) exchangeable calcium and is based on the fluxes shown in the diagram. In both plots the Ca budget is nearly in balance with P40 soil showing a slight gain and P30 a small loss (0.5% per annum) of exchangeable Ca^{2+}. However if the lowest Ca^{2+} weathering rate of 8 $mol_c ha^{-1}year^{-1}$ is used, the input - output budget would suggest an annual loss of 6 % of the exchangeable pool of Ca^{2+}, which would lead to 0 exchangeable Ca^{2+} within 17 years. Exchangeable Ca^{2+} levels in the O horizon of a plot adjacent to P40 obtained in 1996 (i.e. 10 years after levels reported here) have not decreased noticeably (4000 $mol_c ha^{-1}$) (De Wit, pers. comm,). This suggests that the weathering rate for Ca^{2+} is certainly much higher than 8 $mol_c ha^{-1}year^{-1}$.



Input = Throughfall + Litterfall + Weathering
Output = Uptake + Flux out at BC

Presentation order for diagram	Input - Output
P40	11
P30	-29

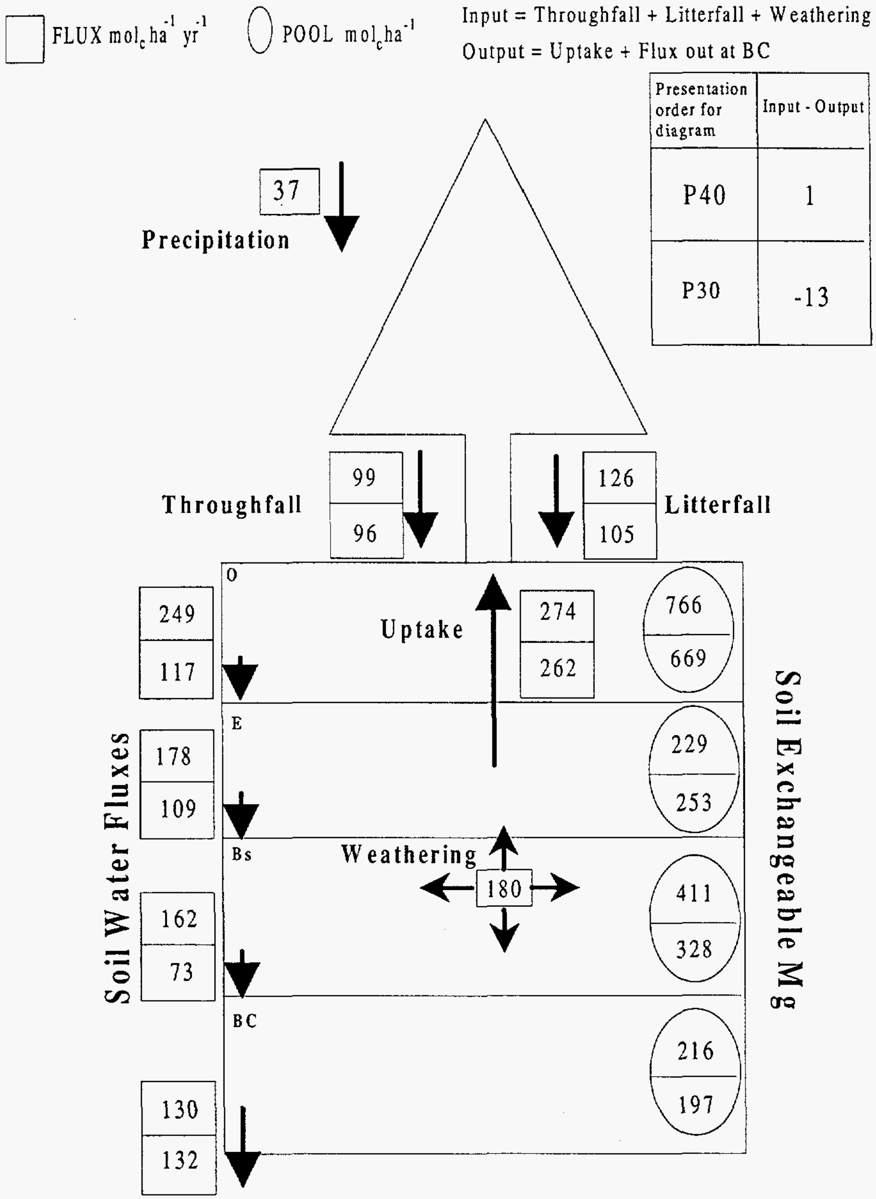
Fig. 4. The Calcium Cycle, with emphasis on input and output fluxes to the soil.

The Magnesium Cycle (Figure 5)

In contrast to Ca, the flux of Mg^{2+} in throughfall, litterfall and all soil water fluxes, except BC is greater in P40 than P30. The weathering flux estimated from soil column studies (180 $mol_c ha^{-1} year^{-1}$) and the rate obtained by MAGIC, (117 $mol_c ha^{-1} year^{-1}$) both compare reasonably with the average flux from Bs and BC in P40 and P30 plots (124 $mol_c ha^{-1} year^{-1}$). Assuming the soil column weathering rate is correct, the main input to the soil is via weathering, followed by approximately equal inputs of litterfall and throughfall. The Mg^{2+} input-output diagram shows much the same pattern as for Ca^{2+}, with P40 in balance and P30 showing a small loss. This slight difference between a gain of exchangeable cations in P40 and a loss of exchangeable cations in P30 in both the Ca and Mg cycle fits Miller's hypothesis that following canopy closure there can be a recharge of soil exchangeable nutrients provided the soil mineralogy is adequate (Miller, 1986).

The Potassium Cycle (Figure 6)

Similarly to Mg, the flux of K^+ in throughfall, litterfall and soil water is consistently larger in P40 compared to P30. Uptake is the main output and little leaching occurs under BC. Different estimates of the weathering rate of K^+ show the least variation and the smallest rate among the base cations (Table VI). The weathering flux for K^+ is similar to the K^+ flux found in the B horizon lysimeters. In both plots, the throughfall flux, mainly due to foliar leaching of K^+, is double the litterfall flux which is approximately double the weathering flux. The input-output diagram shows a large loss of exchangeable K^+ in both plots, 6 % and 3 % per annum in P40 and P30 respectively. Such a large loss in P40 seems unrealistic, and perhaps gross uptake from soil (equation 9) has been overestimated and the requirement for K in new wood is covered to a larger degree by translocation. This result of a larger loss of exchangeable K^+ for the older stand (P40) is in contrast to the pattern found in both the Ca and Mg cycle and does not fit Miller's hypothesis, although clearly soil mineralogy is unable to recharge the deficit in the K^+ exchangeable pool after canopy closure. For both plots, the diagram emphasises the large tree demand for K^+ and the small soil pools of K^+. It shows clearly how important the return of foliage and branches will be to the soil at harvest for rebuilding these exchangeable pools.



Presentation order for diagram	Input - Output
P40	1
P30	-13

Fig. 5. The Magnesium Cycle, with emphasis on input and output fluxes to the soil.

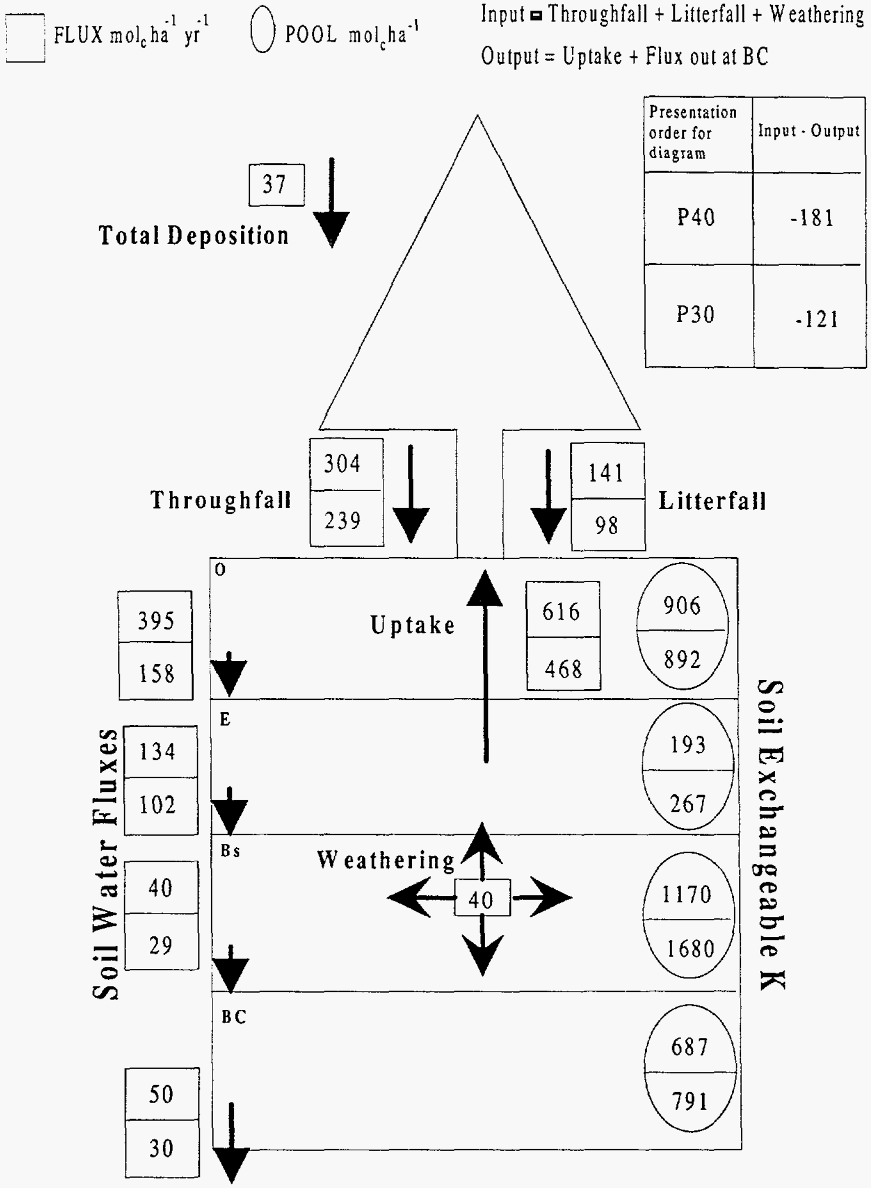


Fig. 6. The Potassium Cycle, with emphasis on input and output fluxes to the soil.

The Nitrogen Cycle (Figure 7)

The total N deposition can be divided into 40% NH_4^+, 50% NO_3^- and 10% organic N. The total deposition of all forms of N was larger to P40 than P30. Over half of both the NH_4^+ and NO_3^- is retained in the canopy, again more in P40 than P30. Uptake by needles (Friedland et al., 1991) and branches (Boyce et al., 1996; Thoene et al., 1996), lichen (Lang et al., 1976) and algae (Thomsen, 1992) could all explain this. Organic N is produced in the canopy, although its production does not cover the loss of inorganic N in this study. Both NH_4^+ and NO_3^- are reduced strongly on passage through the forest floor, slightly reduced in the E horizon and remain at this low level through the rest of the profile. Organic N remains high in the E horizon but is reduced substantially in the B horizons, suggesting adsorption and possibly consumption. The flux of organic N is slightly higher throughout P40. Litterfall is the large input of N into the soil and uptake the large output. The sum of throughfall NH_4^+ and NO_3^- covers 12 % and 18 % of gross soil uptake in P40 and P30 respectively. Mineralisation of annual litterfall could supply the remaining N uptake in P30 leaving only 82 mol $ha^{-1}year^{-1}$ of N to be immobilised in the soil. In P40, mineralisation of all annual litterfall would still leave a deficit of 105 mol $ha^{-1}year^{-1}$ of N, which could be supplied from the large pools of existing soil N. The application of the MERLIN model at an aggrading Sitka spruce forest (Emmett et al., 1997) also showed that nitrogen for uptake had to be mined from the refractory organic matter (ROM) when the trees became older. This same trend of decreasing soil N in deeper soil horizons was found in stands (20-50 years old) at Nordmoen (Sogn et al., in review).

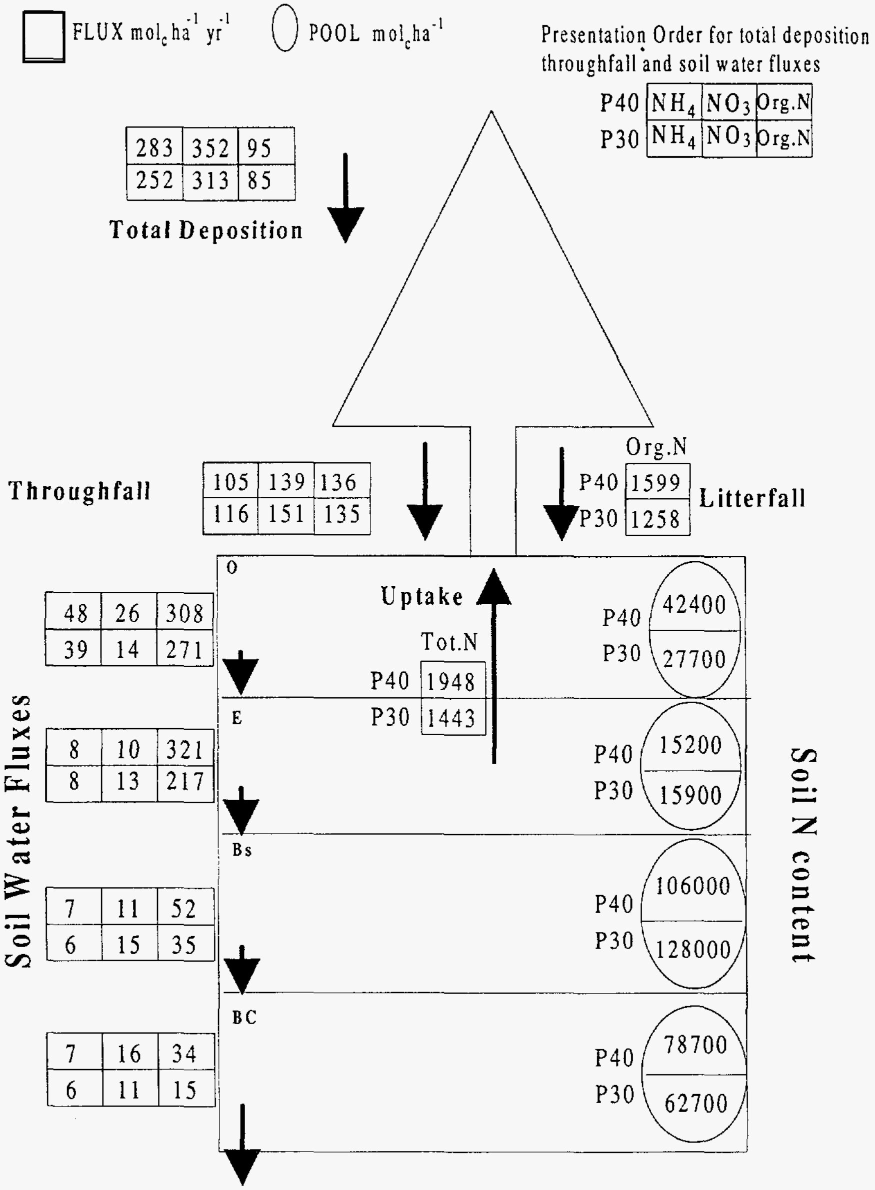


Fig. 7. The Nitrogen Cycle.

4. Conclusion

Data collected at the 30- (P30) and 40-(P40) year-old spruce plots at Nordmoen, south-east Norway showed the following similarities between the plots.

- Total deposition was dominated by seasalt, H^+, SO_4^{2-}, NO_3^- and NH_4^+.
- Both canopies retained approximately half of the annual inorganic N, while organic N was released. Uptake and exudation by trees and epiphytes could explain this but more research is needed.
- For Ca, litterfall was the most important input to the soil, while for Mg, weathering was the largest input to the soil. The whole K cycle indicated tight recycling, where throughfall was the dominant input to the soil. On the broad scale, the Ca and Mg cycle show balance for the input-output budgets for soil, while the K cycle shows a large annual loss of exchangeable K^+.

The following differences in solute fluxes and nutrient cycling between P40 and P30 were found, which we attribute to difference in stand age.

- Total deposition of non-marine components and NCE of all ions except Ca^{2+} and organic N were greater in P40 than P30, due to its 5 m extra height and larger surface area. Litterfall fluxes of Mg^{2+}, K^+ and N were also greater in P40 than P30. This caused higher nutrient fluxes of most elements in throughfall and through the forest floor. In the mineral horizons, Al^{3+} became the dominant cation in soil solution in P40, while Ca^{2+} was the dominant cation in P30.
- The greater NCE and litterfall flux of most elements, except Ca^{2+} on P40 compared to P30 fits with Miller's hypothesis that biomass cycling fluxes like throughfall and litterfall will be greater in an older stand than a stand only five years after canopy closure.

  In addition, the following two conclusions are general for both plots.
- Soil water fluxes were not accurately determined from Cl^- concentrations in throughfall and in the different soil layers, even on a four year scale. The high spatial variability in average Cl^- concentration in soil water under the forest floor was influenced by the spatial variability of Cl^- in throughfall. The spatial variability in water flow caused by roots may cause spatial variability of Cl^- concentration in soil water in the mineral horizons. In future nutrient cycling studies, the type and placement of throughfall collector relative to the tree trunk and lysimeters must be well-planned. In addition tension and moisture content measurements are recommended for more reliable calibration of hydrological models.
- Estimated weathering fluxes from a 6- year soil column study (Teveldal and Jørgensen, 1996) and modelled by MAGIC (Wright et al., 1991) were compared to the soil water fluxes in the B horizons and input-output budgets in the base cation cycles. The K^+ and Mg^{2+} weathering rates of both methods fitted

well with the field data. For Ca^{2+}, the low MAGIC rate (8) leads to an annual loss of 6% exchangeable Ca, while the high soil-column rate (360) leads to balance in the input-output budget. However the high rate from the soil column experiment could have been overestimated due to the problem of separating Ca^{2+} generated by mineralisation of organic matter, which litter studies indicate is released relatively slowly, from Ca^{2+} generated by mineral weathering.

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PAPER 3

Calibration of the Nutrient Cycling Model (NuCM) at Nordmoen, a Norway Spruce site in south-east Norway

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Abstract

The Nutrient Cycling Model (NuCM) is a process oriented research model simulating nutrient cycling in forest ecosystems. Here, NuCM is calibrated using one year of data from a Norway spruce stand at Nordmoen, south-east Norway under ambient conditions. A comprehensive list of the parameters used is included. Concentrations of base cations could be calibrated satisfactorily in canopy throughfall. By contrast, nitrogen compounds could not be calibrated well, as no description of processes removing nitrogen from water in the tree canopy is included in NuCM. The soil solution concentrations of Na^+, Mg^{2+}, K^+, SO_4^{2-}, Cl^- and H^+ were simulated adequately. However inorganic N was simulated poorly due to the lack of a description of immobilisation of inorganic N and inflexibility on user control of nitrification parameters. Total monomeric Al (Ala) was calibrated reasonably in all horizons, but E, where it was underestimated. Organic monomeric Al (Alo) was poorly simulated due to limiting the definition of organic Al in NuCM to only the complex, $Al^{3+}R^{3-}$. DOC was adequately simulated in the O and E horizons, but poorly simulated in the B horizons. The description of biogeochemical processes in NuCM are evaluated as surplus, adequate, poor or missing after the outcome of the calibration. In a companion paper the calibrated model is tested using a three year data set from the same site.

Keywords: Nutrient Cycling Model (NuCM), calibration, throughfall, soil solution

1. Introduction

Several models have been developed to quantify the effects of atmospheric deposition on nutrient cycling in forest ecosystems (Reuss, 1976; Ulrich, 1980; Binkley and Richter, 1987; Tiktak and van Grinsven, 1995). One of these, the NuCM model (**Nu**trient **C**ycling **M**odel ; Munson et al., 1992) was developed as part of the Integrated Forest Study (IFS) (Johnson and Lindberg, 1992). NuCM has been used to investigate S deposition scenarios (Liu et al., 1991; Johnson et al., 1993), to simulate base cation leaching (Johnson, 1995) and to simulate effects of harvesting and tree species change on nutrient cycling (Johnson et al., 1995). NuCM was also tested on Norwegian lysimeter data with increasing N deposition levels (Sogn et al, 1995 ; Sogn and Abrahamsen, 1997). However so far NuCM has not been calibrated and tested at the forest stand level under natural conditions.

NuCM is a process-orientated research model, which evaluates the movement and distribution of nutrients and quantifies the effect of deposition quality on nutrient cycling in forests. NuCM, demanding an extensive parameter set (103 parameters) and having a daily, weekly or monthly timestep, aims at reproducing seasonal and annual variations far more accurately than less complex models. In Norway, the models MAGIC (Cosby et al., 1985) and PROFILE (Sverdrup and Warfvinge, 1993) have been most commonly used to simulate the effects of S and N deposition on forest ecosystems (Wright et al., 1991; Frogner et al., 1992). These models simulate results on an annual basis and do not include dynamic modelling of nutrient cycling processes like exudation, litterfall and mineralisation.

In this paper the NuCM model is calibrated using one year of data for a Norway spruce stand at Nordmoen (south-east Norway). Calibration, in this context, is the estimation and adjustment of model parameters and constants to improve the agreement between the model output (simulation) and the data set (Rykiel. 1996). The main objectives are (1) to give a comprehensive description of how parameters were obtained and (2) pinpoint weaknesses and strengths of process descriptions in light of the calibration result. In a companion paper, the calibrated model is tested on three years data from the same site.

2. Model description

NuCM is a dynamic process oriented model describing the atmospheric, hydrologic, vegetation and soil processes which alter the chemical composition of water as it moves through the tree canopy and down the soil profile. NuCM simulates the concentration of 14 solutes; Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, Ala^{3+} (total monomeric Al), Alo (organic monomeric Al), NH_4^{+}, NO_3^{-}, SO_4^{2-}, Cl^{-}, DOC, PO_4^{-}, SiO_2 and DIC and the two variables pH and ANC. These components are tracked through several pools, including canopy, bole and roots, leaf litter, snowpack, soil solution, soil minerals, soil exchange surfaces and soil organic matter. Many processes are included to simulate the fluxes in and out of these pools (Fig. 1). Full details of these processes can be found in Munson et al., (1992) and Kvindesland (1997). Tree growth is modelled more empirically as a function of a user-defined development stage and the availability of nutrients and water. The model separates the hydrology and biogeochemistry simulations. The hydrology simulation must be run first and always uses a daily time step. However the time step specified for the biogeochemical simulation can be daily, weekly or monthly.

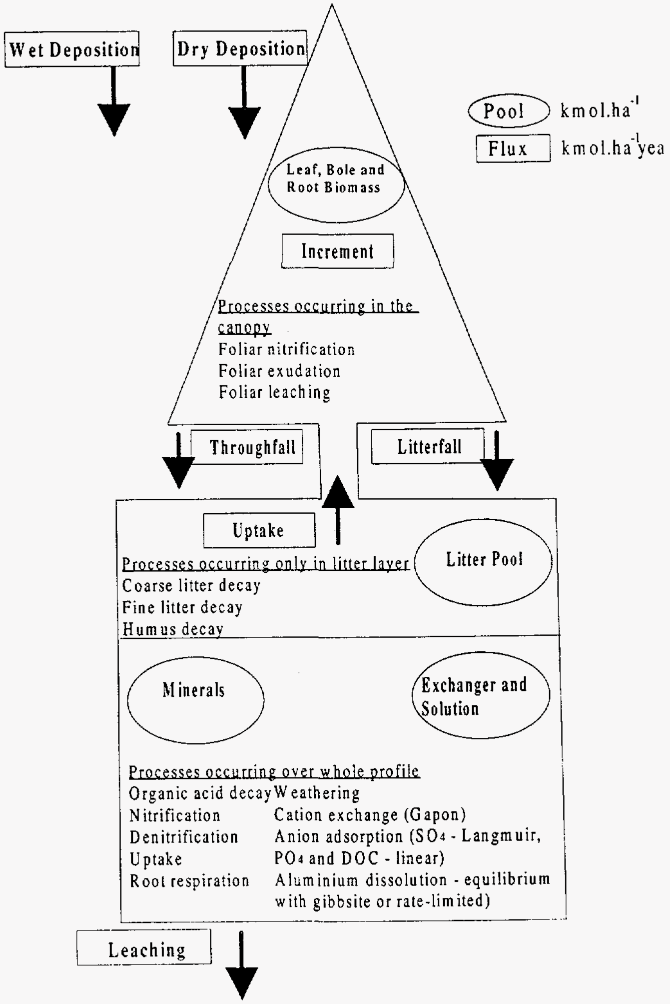

Fig. 1. An overview of the NuCM model. Main processes, pools and fluxes in the biogeochemistry simulation.

3. Site description

The study site Nordmoen is located on a flat glaciofluvial plain 45 km north of Oslo at an elevation of 200 m. The soil is classified as a Cambic Arenosol (FAO) (Stuanes and Sveistrup, 1979). For the calibration we used the P30 plot, which is a natural 30 year old Norway spruce (*Picea abies* (L.) Karst) plot, 1000 m^2 square in size. The second year (1 Oct. 1987 - 30 Sept. 1988) of the measurement period (Oct. 1986 - Dec.1990) was chosen for the calibration to avoid lysimeter installation effects. The lysimeters were made from modified Buchner funnels and a tension of 10 Kpa was supplied continually to each plate. Average total S and N deposition was each about 10 kg $ha^{-1}year^{-1}$. Full details of the site, measurements and methods used can be found in Kvindesland and Røsberg, (in review).

4. Parameters used

4.1 Meteorological data

The meteorological data required to run NuCM includes daily values of precipitation and maximum and minimum air temperature (Fig. 2). Monthly values of soil temperature, solute concentrations in precipitation ($mmol_c l^{-1}$), and ambient air concentrations ($\mu mol_c m^{-3}$) are also required. The latter two are input in the AIR file.

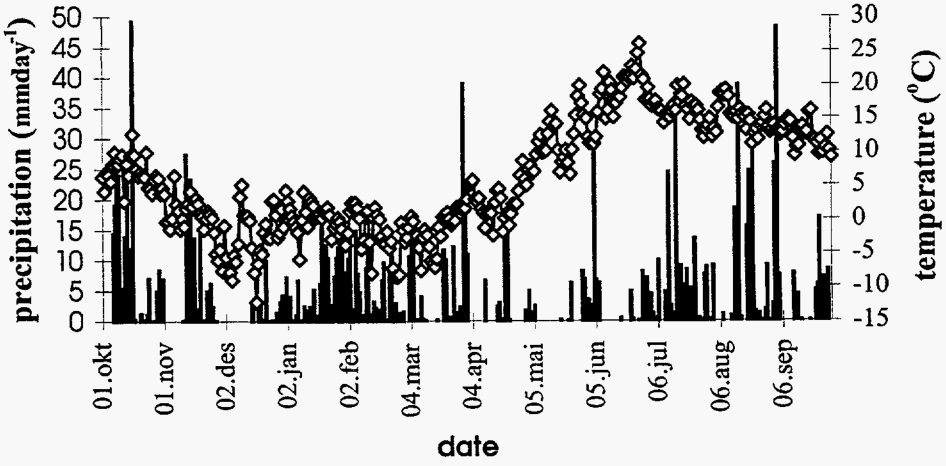

Fig. 2. Daily precipitation and daily mean air temperature.

The canopy processes in NuCM, which include foliar exudation (natural leaching) and foliar leaching (leaching caused by low pH), both add components to throughfall but no processes remove components, except H^+/OH^- from the incident precipitation. However about half the incident inorganic N disappears in the canopy in P30 (Kvindesland and Røsberg, in review). To circumvent problems associated with modelling N immobilisation in the canopy, measured concentrations of NH_4^+ and NO_3^- in throughfall and simulated throughfall volumes were used as the infiltration solution to the soil during the growing season. In contrast, measured concentrations of Ca^{2+}, Mg^{2+} and K^+ in precipitation are used in the AIR file throughout the year. Foliar exudation rates and amplification factors (see Tables 6 and 7) were calibrated to measured throughfall concentrations for these three base cations. The final composition of the H2O file (the monthly result file from the hydrology simulation) and AIR file, which are used as input to the biogeochemical simulation are shown in Fig. 3. The hydrology routine in NuCM is replaced with hydrology results of the SOIL model (Jansson, 1991) for reasons discussed in "Results and Discussion".

June - November: (growing season)

AIR FILE: Precipitation quality for all components (except Ca, Mg and K) is replaced with measured throughfall concentrations. For Ca, Mg and K, measured precipitation concentrations used. No ambient air quality.

H2O FILE: NuCM precipitation, throughfall and infiltration results replaced with SOIL's infiltration results. NuCM's percolation results replaced with SOIL's percolation results.

December - May: (dormant season)

AIR FILE: Measured precipitation and ambient air quality used.

H2O FILE: NuCM precipitation and throughfall results replaced with SOIL's. precipitation. NuCM's infiltration replaced with SOIL's infiltration. NuCM's percolation results replaced with SOIL's percolation results.

Fig. 3. The final composition the AIR and H2O input files used in the biogeochemnical simulation.

4.2 Soil parameters

Tables 1-5 list the soil physical and chemical parameters, the organic matter parameters, mineralogy and weathering parameters and equilibrium constants. Details of how the parameters were obtained are specified under each table.

Table 1. Soil physical and chemical parameters

Parameter Name	Descripition, Units	O	E	Bs	BC	C
thick	soil layer thickness, cm	7	1	25	20	600
etwght[1]	fraction of roots in layer, decimal fraction	0.78	0.02	0.15	0.05	0
CEC[2]	$mmol_c kg^{-1}$	251	89	44	8	4
dnsity	bulk density of soil, $g\ cm^{-3}$	0.09	0.87	1.11	1.25	1.44
PCO_2[3]	soil CO_2 gas concentration factor	8	10	20c	30c	55

[1] average of all living roots, June 1986 (Røsberg, pers. comm.).

[2] extracted in 1M NH_4NO_3 (Ogner et al., 1991).

[3] O and E layer (Fernandez and Kosian, 1987), Bs and BC layer by calibration to soil pH, C layer based on Nordmoen data (Jørgensen and Østmo, 1990).

c = calibrated.

Table 2. Soil Chemistry parameters

Parameter Name	Description, Units	Mineral / Layer	NH_4	Ala	Ca	Mg	K	Na	SO_4	NO_3	Cl	PO4	SiO2	Organic C	H
mnwth[1]	dissolution	CHLO				6000							4000		
	stoichiometry	ALBI						1000					2000		
	of minerals,	BIOT				4000	1000						2000		
	$mmol_c$ mol^{-1} weathered	ANOR			2000								2000		
disun	distribution	O	90c	0c	20c	20c	70c	50c	35c	90c	70c	40c	30c		
	of uptake of	E	5c	0c	10c	10c	10c	5c	30c	5c	5c	5c	30c		
	nutrients in	Bs	5c	0c	40c	40c	20c	25c	35c	5c	20c	25c	30c		
	profile, %	BC	0c	100c	30c	30c	0c	20c	0c	0c	5c	30c	10c		
solion[2]	dissolved ions	O	2.7	32.2	158.9	31.7	14.2	24.9	77.8	2.4	36.1	4.2	1.3	61.7	
	in	E	2.0	80.1	110.8	27.3	20.9	36.9	75.5	2.2	37.6	5.1	2.2	45.5	
	soil solution, $\mu mol_c l^{-1}$	Bs	1.5	51.1	51.7	19.2	9.8	41.0	126.7	1.7	37.2	1.0	1.5	7.5c	
	DOC & SiO2 (*mg* l^{-1})	BC	1.8	14.6	52.8	36.4	7.2	71.8	110.4	1.5	54.6	1.0	1.6	4c	
adscat[3,4]	adsorbed anions or	O	5.0c	7.95	39.4	7.65	10.6	0.60	0.38			2.72		250c	28.8
	cations held on	E	0	60	4.4	1.82	1.91	0.35	0.38			1.15		330c	31.5
	CEC, cations %	Bs	0	84	1.24	0.28	1.4	0.35	1.06			0. 98		1400c	12.7
	anions, $mmol_c kg^{-1}$ DOC $mg.kg^{-1}$	BC	0	74.5	2.87	1.05	4.05	1.83	1.20			3.32		400c	15.7

[1] Chlorite (CHLO) and biotite (BIOT) incongruent reaction, albite (ALBI) and anorthite (ANOR), congruent reaction (Stumm and Morgan, 1981).

[2] Average volume weighted concentrations from 4 lysimeters in 2 pits in P30, Oct 1987- Septt 1988, except Ala, average concentration (Jan. 1987-June 1988).

[3] Average of 2 soil pits dug in 1986. Ca, Mg, K, Na and exch. acidity extracted at soil pH in 1M NH_4NO_3. Exchangeable NH_4 was not measured, but calibrated to damp NO_3 leaching in the winter. Adsorbed SO_4 is extracted by $CaPO_4$. PO_4 extracted by sulphuric acid (Ogner et al., 1984). Adsorbed organic C is calibrated to one tenth of the measured organic C in Bs and BC layers as determined by a Carlo Erba Elemental analyser 1106 in profile 3 Stuanes and Sveistrup, 1979.

Table 3 Organic matter decay and Nitrification Parameters (for further details, see appendix)

Parameter Name	Description, Units	Value
xlit[1]	Weight fraction of coarse litter in top soil layer	0.03
fines[1]	Weight fraction of fine litter in top soil layer	0.07
humus[1]	Weight fraction of humus in top soil layer	0.9
frlch (1)[2]	Fraction of immediately leachable ions in litterfall	0.01
frlch (2)[3]	Fraction of S & N that mineralises when fine litter decomposes into humus, i.e. fraction not incorporated into humus	0.37
frlch (3)	Fraction of S and N that is incorporated into organic acid during humus decay	0.7c
oaln[5]	Amount of N in organic acid, mole N mol^{-1} acid	0.9
oals[5]	Amount of S in organic acid, mole S mol^{-1} acid	0.17
oalc[5]	Amount of C in organic acid, mole C mol^{-1} acid	30
lbrate[6]	coarse litter to fine litter breakdown rate, $year^{-1}$	1.3c
fbrate[6]	fine litter to humus breakdown rate, $year^{-1}$	0.7c
hbrate[7]	humus to fulvic acid breakdown rate, $year^{-1}$	0.075c
oarate[8]	organic acid breakdown rate, $year^{-1}$	0.0001c
mcton	Microbial C:N ratio	24c
ckvnh4[9]	Nitrification rate for Soil day^{-1}	0c

[1] The coarse litter was assumed to be the needle litterfall for one year at this site, 1992 kg $ha^{-1}year^{-1}$, fine litter was defined as the Oi layer (2 cm, 0.02 g cm^{-3}) and humus as the Oe+a layer (4.3 cm, 0.125 g cm^{-3}). The weight of the components per $1m^2$ was then calculated.

[2] It is known that K and Mn can have high initial leaching from litter, while Ca accumulates (Berg & Tamm, 1994) but as frlch(1) covers all ions (except C, N, H and S), a low value was chosen.

[3] 1- [Total N (Oi)+Total S(Oi)]/[Total N (Oe+a)+Total S(Oe+a)]

[4] NuCM lumps the N and S fractions of fine litter and humus, purely to simplify the mathematical expression as I am unaware of any research showing that N and S react similarly during litter decay.

[5] Mean contents in soil fulvic acid (Schnitzer, 1978).

[6] lbrate and fbrate were calibrated to keep the N, S and Ca contents of the fine litter pools accumulating slightly during a year.

[7] hbrate was calibrated as a compromise between fitting the high DOC soil solution concentration and avoiding too high SO_4 and NH_4 soil solution concentrations.

[8] Calibrated as low as possible to fit the high DOC soil solution concentration.

[9] Calibrated to zero - see 5.2.2.

c = calibrated

Table 4.Weathering parameters

Parameter Name	Description, Units	Layer	Mineral 1	Mineral 2	Mineral 3	Mineral 4	Mineral 5
nmnrls[1]	name of minerals (max. 5, minerals 1-4 are choice-free)	all	chlorite	albite	biotite	anorthite	gibbsite *
mnmwt	mineral molecular weight, g mol^{-1}		554.2	262.2	417.3	278.2	78
wexp[2]	rate dependence on H^+ ion for weathering rate		0.5	0.5	0.5	0.5	0.4
slcomp[3]	% by weight	O	0	0	0	0	5
	of each	E	0.49	3.88	1.65	3.88	5
	mineral type	Bs	2.59	3.96	1.95	3.96	5
		BC	4.9	3.92	3.75	3.92	5
ckmnr[4]	specific	O	0	0	0	0	25
	reaction rate	E	0.005	0.06	0.24	0.3	25
	weathering	Bs	0.005	0.037	0.04	0.037	0
	constant	BC	0.001	0.03	0.02	0.03	0

Mineralogy[1] in each layer[3] (Teveldal et al., 1990; Teveldal, 1993). The weathering rate[4] and rate dependence on pH[2] have been calibrated to fit the weathering fluxes found in column experiments (Teveldal and Jørgensen, 1996). * Gibbsite dissolution in all layers was set as rate-limited.

Table 5.Equilibrium constants

Parameter Name	Description, Units		Value
KC1	H_2CO_3*, pka, M		6.38d
KC2	HCO_3, pka, M		10.3767d
KH	Henry's Law constant, CO_2 (g), logkH, M atm^{-1}		-1.4054d
KSO	Gibbsite solubility constant, logkso		8.38d
KAL1	$Al^{3+} + H_2O = Al(OH)^{2+} + H^+$, logk1		-4.987d
KAL2	$Al^{3+} + 2H_2O = Al(OH)_2^+ + 2H^+$, logk2		-10.133d
KAL4	$Al^{3+} + 4H_2O = Al(OH)_4^- + 4H^+$, logk4		-22.159d
KAR1	$AlR = Al^{3+} + R^{3-}$, logkR		8.029d
KR1[1]	$H_3R = H_2R^- + H^+$		2c
KR2[1]	$H_2R^- = HR^{2-} + H^+$		4.5c
KR3[1]	$HR^{2-} = R^{3-} + H^+$		8c
KW	H_2O, pkw, M		14.1652d
B2[2]	SO_4 adsorption, half saturation	O	3600
	constant, $\mu mol_c l^{-1}$	E	3500
		Bs	2030
		BC	1540
EMAX	max. adsorption capacity, $\mu mol_c kg^{-1}$ (all layers)		18000

[1] KR1 - KR3, calibrated to fit measured soil pH. [2] B2 = $\{(EMAX[SO_4(l)])/adscatSO_4(l))\}-[SO_4(l)]$ where l denotes value for soil layer and EMAX came from Langmuir adsorption isotherm of Nordmoen soil (Singh, 1984). c = calibrated d = default model value

Table 6. Nutrient cycling monthly parameters

Parameter Name	J	F	M	A	M	J	J	A	S	O	N	D
udistc [1]	0	0	0	0.025	0.25	0.3	0.15	0.15	0.1	0.025	0	0
laic [2]	10.1	10	10	9.9	10	10.3	10.4	10.5	10.2	10.2	10.2	10.2
lfc [3]	0.009	0.009	0.009	0.009	0.009	0.014	0.022	0.012	0.03	0.022	0.007	0.002
betac [4]	0.00	0.002	0.002	0.002	0	0.004	0.02	0.015	0.02	0.055	0.02	0.002

[1] monthly fraction of annual nutrient uptake by trees, calibrated

[2] leaf area index,L.A.I., m^2m^{-2}, based on litterfall data and foliage mass in August 1987, which is assumed as max. L.A.I. Litterfall is subtracted to find new L.A.I.

[3] Litterfall, kg m^{-2} $month^{-1}$, measured all summer months and as one lump sum from December - May, which has been divided equally among the 6 months of the dormant season (Røsberg, pers. comm.).

[4] foliar exudation rate, day^{-1}, calibrated to fit Ca^{2+}, Mg^{2+} and K^+ throughfall concentrations

Table 7. Biomass Chemical Composition

Parameter Name	Description	Units	N	Al	Ca	Mg	K	Na	S	Cl	P	C	Si
lfcmpc[1]	leaf	$mg.g^{-1}$	12.6	0.2	4.4	1.0	4.5	0.1	1.0	0.01	1.5	400.0	5.0
trcmpc[2]	trunk/bole	$mg.g^{-1}$	0.9	0.001	0.7	0.1	0.7	0.01	0.01	0.003	0.1	400.0	
rtcmpc[3]	root	$mg.g^{-1}$	6.4	4.14	2.3	0.6	2.1	0.01	0.5	0.003	1.0	400.0	
excamp[4]					0.016c	0.014c	0.026c						

[1] Needle data from plot P30 1986-1988, (Johnson and Lindberg, 1992, appendix, Table NS-1.) and Røsberg, (pers. comm.).

[2] Bole data from A2, 1983 (Røsberg, pers. comm.).

[3] Fine roots (< 0.5mm), from plot P30, 1986. (Røsberg, pers. comm.).

[4] Foliar exudation amplification factor, calibrated to fit throughfall concentrations. c = calibrated.

4.3 Tree parameters

Tables 6-8 list monthly nutrient cycling parameters, the chemical composition of the biomass and tree stand parameters. Again details of how the parameters were obtained are specified under each table.

Table 8. Tree stand parameters

Parameter Name	Description, Units	Value
standc[1]	standing biomass (only bole), kg m^{-2}	5
rtbioc[1]	root biomass, kg m^{-2}	2.9
constc[1]	conversion factor from leaf surface area to leaf dry weight, g cm^{-2}	0.0154
respcg[2]	active respiration, mg C $cm^{-2}s^{-1}$	4.73E-10
respcm[2]	maintenance respiration, mg C $cm^{-2}s^{-1}$	4.73E-10
uplimit, lowlimit	Uptake limits for nutrients i.e the plant can cope with 10% over or under the normal nutrient demand, before growth and uptake of other nutrients is affected.	0.1 d 0.1 d
growc[3]	Annual vegetation growth rate, kg m^{-2} $year^{-1}$	0.385
closec	Number of years to stand closure	20
agec	Current stand age	30

[1] (Johnson and Lindberg, 1992, appendix, Table NS-1)

[2] Respiration (Dahl and Mork, 1959)

[3] Annual growth (Johnson and Lindberg, 1992, appendix, Table NS-3)

d =default model value

4.4 Calibration procedure

A monthly timestep was chosen for the biogeochemical simulation as analysed lysimeter data was available at a monthly timestep during the frost-free season.

The original NuCM manual (Munson et al., 1992) suggests an order for the biogeochemical calibration, which I followed during the first rounds of calibration. Throughfall was calibrated first. The organic matter pools of fine litter and humus were investigated for Ca, NH_4 and SO_4 and annual changes in these pools were calibrated as low as possible with very slight accumulation. Anion concentrations were then calibrated followed by cation concentrations. Finally the calibrated model was run for 10 years to check for non-realistic changes in the exchangeable soil pools and throughfall and soil solution phases.

5. Results and discussion

5.1 Hydrology

The NuCM hydrological model produced a reasonable annual hydrology budget, where throughfall volume was simulated satisfactorily. This year was very wet compared to the long-term average precipitation (880 mm) at Nordmoen (Vose and Swank, 1992) so the high percolation to groundwater can be partly explained by high precipitation, although no reliable independent measurement exists to check the simulated percolation to groundwater (Table 9). The calibration of the monthly water fluxes has proved difficult because NuCM bases the percolation calibration on Cl^- concentrations in soil water, which were found to vary greatly spatially in all soil layers (Kvindesland and Røsberg, in review).

Table 9. Annual hydrological budget (mm) 1.Oct.1987- 30 Sept. 1988

	Precipitation	Canopy Evaporation	Throughfall	Evapotranspiration	Percolation to groundwater
Simulated by NuCM	1300	250	1050	295	755
Measured	1300	360	940		

Previously the SOIL model (Jansson, 1991) which describes the transfer of water and energy in soils had been applied to the same dataset. SOIL uses water potentials in addition to soil water Cl^- concentrations as a second constraint on the monthly water fluxes (Eckersten et al., 1995). Measured snow depths, simulated snow depths by SOIL and monthly simulated snow water equivalents by SOIL and NuCM are compared (Fig. 4). The SOIL snow depth correlates visually well with the measured snow depth, while the NuCM snow water equivalent shows poor correlation to both the SOIL snow water equivalent and measured snow depth, particularly from late March onwards. The SOIL and NUCM simulated water fluxes through the BC horizon are compared to water volumes collected by tension lysimeters (Fig. 5) The simulated water fluxes correlated visually well for the snow-free months (May - December). However the percolation pattern differed in the winter with NuCM simulating fairly high percolation in January-March and low in April-May, while SOIL had lower percolation the first three months and high percolation in April/May. NuCM appears to simulate too little snow and too much snow melt from January - March.

As the SOIL results compare best with measured snow depths and the water volumes found in the lysimeters in April/May, I chose to replace NuCM's hydrology results with SOIL's.

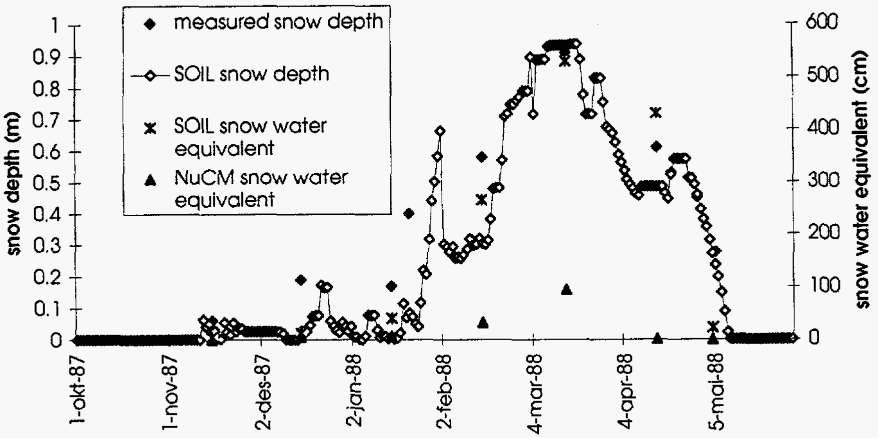

Fig. 4. Simulated snow water equivalents and simulated and measured snow depths at Nordmoen, winter 1987-1988.

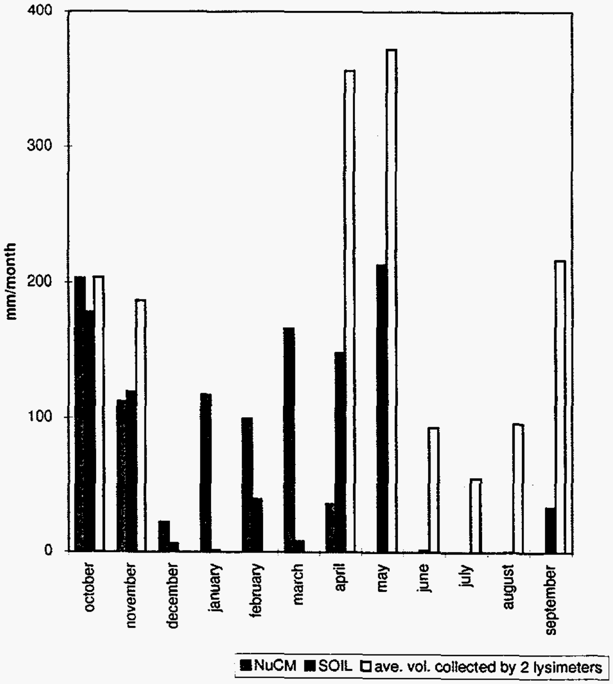

Fig. 5.
Waterflow draining from the BC horizon simulated by NuCM and SOIL and average water volume collected by two tension lysimeters under BC horizon. The conversion from ml/month (y) to mm/month (z) was based on on the lysimeter plate area surface (z=y/5.55).

5.2 Chemistry

To evaluate the quality of the simulations, the normalised mean absolute error (NMAE) is used, which is defined according to:

$$MAE = \frac{\sum_{i=1}^{N} |S_i - O_i|}{N}$$

$$NMAE = \frac{MAE}{\overline{O}}$$

Where S_i and O_i denote the simulated value and observed value, $\overline{O}$, the observed sample mean and N, the number of simulations/observations.

The NMAE accounts for the spread in the deviations, $S_i - O_i$. The closer to zero, the better the simulation.

5.2.1 Throughfall

Only Ca^{2+}, Mg^{2+} and K^+ results are shown for throughfall (Table 10 and Fig. 6) and the results appear reasonable, although a lumped exudation rate for all elements cannot explain the finer dynamics of exudation in the growing season. Base cations in throughfall were simulated from wet and dry deposition and exudation from needles whereas the other ions were input as throughfall and are therefore no test of the simulation (see 4.1, Fig. 3).

In respect to the variables, bulk deposition and throughfall normally measured at Norwegian forest sites, the detailed canopy interactions of NuCM, including wet and dry deposition, foliar nitrification, exudation and leaching are processes, which are impossible to parameterise at the present data level. It is tempting to use measured throughfall concentration and volume as an input to the soil system, which considerably reduces the number of measurements and parameters needed. However this excludes nutrient cycling aspects such as the quantification of the exudation of base cations, which is part of the uptake equation. On the other hand, NuCM's detailed canopy interactions do not include the sorption/uptake of inorganic N, nor the production of organic N which was observed at Nordmoen (Kvindesland and Røsberg, in review). Present research on Norway spruce suggests that foliar uptake of NH_3, NO_2 and HNO_3 in gaseous and vapour form (Perez-Soba and Van der Eerden, 1993; Thoene et al., 1996; Norby et al., 1989), foliar uptake and twig uptake of NO_3^- and NH_4^+ from wet deposition (Bowden et al., 1989; Boyce et al., 1996) and epiphytic uptake of inorganic N (Lang et al., 1976; Friedland et al., 1991) are all possible sinks. Tree uptake will obviously affect biomass contents and the calculated uptake and requirement for N. Lichen and algae uptake will affect litterfall fluxes.

Table 10. Simulated and Observed Mean and the Normalised Mean Absolute Error (NMAE) of the throughfall and soil water horizons for the period October 1987-October 1988. (Units $\mu mol_c\, l^{-1}$*, except DOC* $mg l^{-1}$*)*

	NO_3-N	NH_4-N	Ca	K	Mg	SO_4-S	Cl	Na	Ala	DOC	pH	Alo
thrfall Sim			38	27	13							
thrfall Obs			34	29	14							
thrfall nmae			0.1	0.1	0.0							
O Sim	9	3	137	13	30	77	46	32	39	63	4.13	3
O Obs	2	3	152	13	31	77	35	25	34a	62	4.16	32a
O nmae	2.8	0.3	0.1	0.0	0.0	0.0	0.3	0.3	n.a.	0.0	0.0	n.a.
E Sim	9	3	108	15	24	64	44	29	61	38	4.38	5
E Obs	2	2	114	18	28	70	39	34	90a	45	4.23	70a
E nmae	3.7	0.4	0.1	0.2	0.1	0.1	0.1	0.2	n.a.	0.2	0.0	n.a.
Bs Sim	7	1	68	10	26	122	56	41	52	8	4.46	3
Bs Obs	2	2	47	8	18	120	39	40	58a	13	4.70	8a
Bs nmae	3.1	0.4	0.5	0.3	0.5	0.0	0.4	0.0	n.a.	0.4	0.1	n.a.
BC Sim		0.5	57	8	31	104	48	60	16	4	4.79	1
BC Obs	n.a.	n.a.	46	7	32	108	47	70	14a	9	5.19	2a
BC nmae	n.a.	n.a.	0.2	0.1	0.0	0.0	0.0	0.1	n.a.	0.6	0.1	n.a.

(N = 7 (Oct. 87, Nov. 87, mean Dec. 87-May 88, June 88, July 88, Aug. 88, Sept. 88) as lysimeter water was collected once during the winter period.). n.a.=not available.

[a] Ala and Alo observed are the mean of the period of Al speciation (Jan. 87- June 88) as these means were used as initial parameter values.

Research is needed to quantify the N consumption and release by these different sinks, before a process-oriented description can be incorporated into NuCM. However an empirical solution to change existing model structure would be to use throughfall concentration and volume on a monthly basis as input to the soil. Then on an annual basis use wet deposition and throughfall to calculate the dry deposition and net canopy exchange (NCE) (for method see Kvindesland and Røsberg, in review). If the NCE is positive, it is assumed due to exudation and if negative, it is assumed due to consumption by needles or epiphytes. These quantities could be taken into account in the annual uptake calculation.

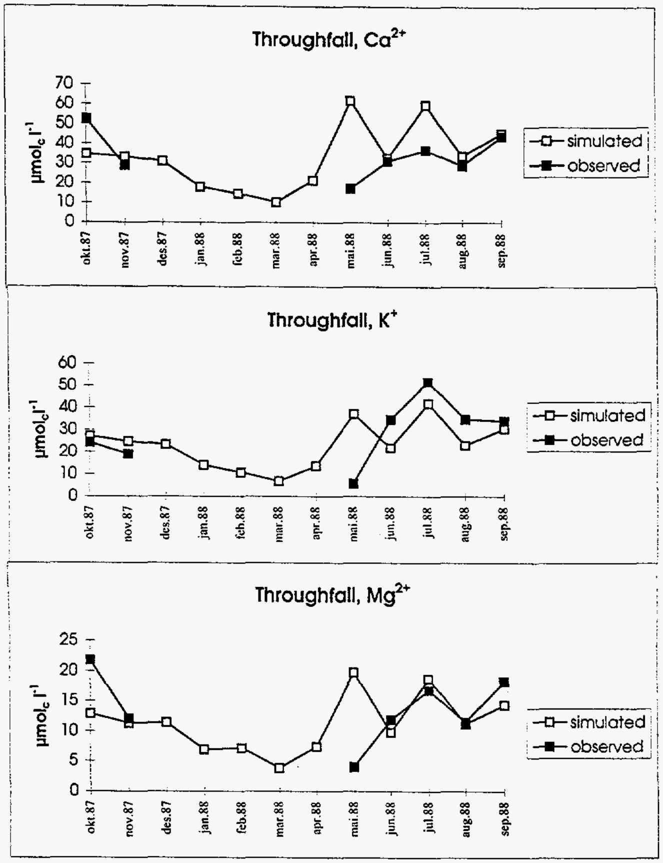

Fig. 6.

Simulated and observed throughfall water for P30 plot, Nordmoen, October 1987 - September 1988. A. Ca $^{2+}$. b.K$^+$ and c. Mg^{2+}.

5.2.2 Soil solution

Nitrate in all horizons of the soil had a high NMAE. This was mainly caused by the high NO_3^- concentrations in soil water simulated throughout the winter. However the lysimeter sample collected at the end of May and representing soil water possible to extract between 1 Dec. 1987 and 31 May 1988 was very low (Fig. 7). The NO_3^- concentration in throughfall was also simulated very high during the winter (Fig. 7) and there are no throughfall observations during this period to elucidate if this a correct simulation or not. If the concentration of NO_3^- in winter throughfall is being simulated too high, this may be one of the main reasons for the simulated high concentrations of NO_3^- in winter in the soil horizons. Alternatively the observations may be biased as some NO_3^- may have been leached during the winter without being registered by the lysimeter, especially if the lysimeter tubing was frozen rather than the soil. The low observed concentration of NO_3^- in the soil water may also be due to immobilisation of NO_3^- by microbes during the winter (Stark and Hart, 1997), a process not included in NuCM. The nitrification rate was put on zero to minimise this problem. However the combination of no nitrification and plant uptake resulted in very low simulated NO_3^- concentrations during the growing season. NuCM also overestimated leaching of NO_3^- at Coweeta (Johnson et al., 1993) and also in lysimeters containing Scots pine, where the same pattern of low NO_3^- in the summer and high NO_3^- in the winter was found (Sogn et al., 1995; Sogn and Abrahamsen, 1997). Nitrification is modelled using a temperature dependent Michaelis-Menten rate expression. Unfortunately Ks, the half saturation constant is fixed to 0.1 and cannot be altered via the normal user screens; a compiler is needed to alter it. Such a low Ks causes a steep inclination up to Vmax, i.e. a large amount of variation in nitrification even on a small difference in NH_4^+. A higher Ks would cause a gentler inclination i.e. less variation in nitrification with a change in NH_4^+.

Both the simulations and observations suggest low concentrations of NH_4^+, yet the NMAEs for all horizons are greater than or equal to 0.4. Despite its low concentration, some fluxes of NH_4^+ are large e.g. NH_4^+ fluxes associated with mineralisation, ammonification and plant uptake (Gundersen et al, in press).

Concentrations of Ca^{2+}, K^+ and Mg^{2+} in soil water show reasonable NMAEs in all horizons but Bs, where weathering overpredicts the concentration. The time plot of Ca^{2+} (Fig. 8) shows the dynamics of Ca^{2+} are well simulated in the top two layers, but the poor simulation in the Bs horizon naturally affects the simulated concentrations in the BC horizon. Weathering has been estimated in many ways at Nordmoen (Kvindesland and Røsberg, in review), but most confidence was placed in the rates derived from column experiments (Table 4; Teveldal and Jørgensen, 1996). Because NuCM limits the number of weatherable minerals to four, it was impossible to match the results reported by Teveldal and Jørgensen (1996) exactly.

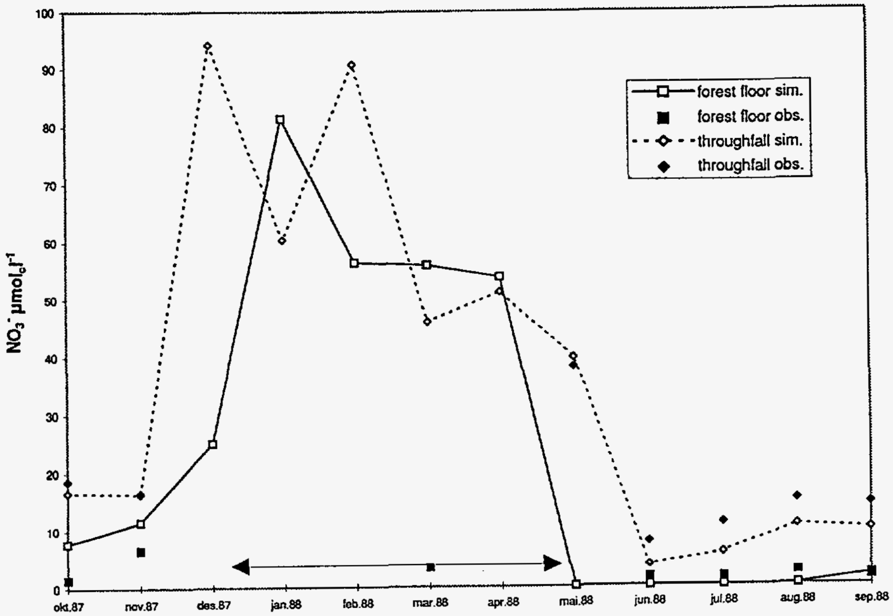

Fig. 7. Simulated and observed NO_3-N concentration in throughfall and forest floor soil water at the P30 plot, Nordmoen, October 1987-September 1988. The frost period (December 1987-May 1988), is indicated by a horizontal arrow, with the lysimeter sample collected at the end of May (centre of arrow) representing the whole period (continual tension applied to lysimeter).

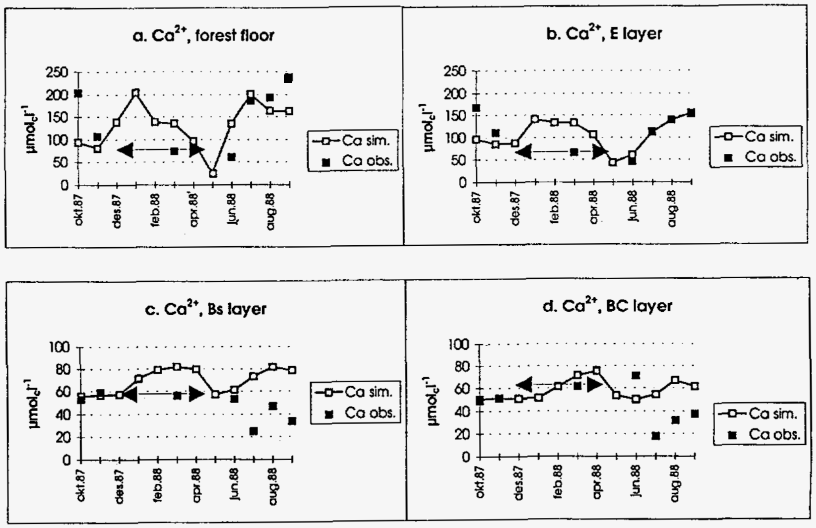

Fig. 8. Simulated and observed Ca^{2+} concentration in soil water at the P30 plot, Nordmoen, October 1987- September 1988. The frost period (December 1987 - May 1988), is indicated by a horizontal arrow, with the lysimeter sample collected at the end of May (centre of arrow) representing the whole period (continual tension applied to lysimeter).

Concentrations of Na^+ and Cl^- show intermediate NMAEs, which was not unexpected for Cl^-, considering the noise in the measured Cl^- data (Kvindesland and Røsberg, in review). The NMAE of SO_4^{2-} was low, which perhaps reflects its lesser involvement in biological processes and a successful description of adsorption (Sogn and Abrahamsen, 1997). The time plot of SO_4^{2-} (Fig. 9) shows close agreement between simulated and observed except in the starting period in the top two layers.

The concentration of Ala is simulated successfully in the O and BC horizons and is slightly underpredicted in the E and Bs horizon. The Ala time plots (Fig.10) show that simulated Ala follows the dynamics of measured Alt (monomeric and polymeric Al) fairly well. All horizons were simulated using a 5% gibbsite content and a rate limited approach to gibbsite equilibrium. Gibbsite equilibrium or rate limited equilibrium and cation exchange are still the most common controls used to model Al solubility in forest-soil- atmosphere models (Tiktak and van Grinsven, 1995). However recent studies suggest that Al concentrations in the soil solution are mainly controlled by sorption with soil organic matter (Walker et al., 1990; Dahlgren and Walker, 1993; Mulder and Stein, 1994). The model WHAM (Tipping et al., 1995) includes the control of dissolved Al by complexation reactions with soil organic matter.

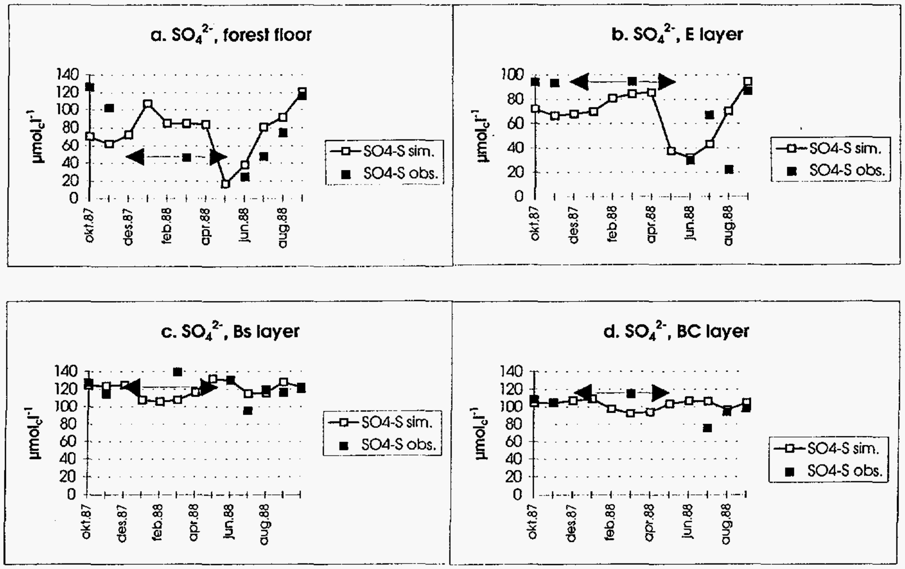

Fig. 9. Simulated and observed SO_4-S concentration in soil water at the P30 plot, Nordmoen, October 1987- September 1988. The frost period (December 1987 - May 1988), is indicated by a horizontal arrow, with the lysimeter sample collected at the end of May (centre of arrow) representing the whole period (continual tension applied to lysimeter).

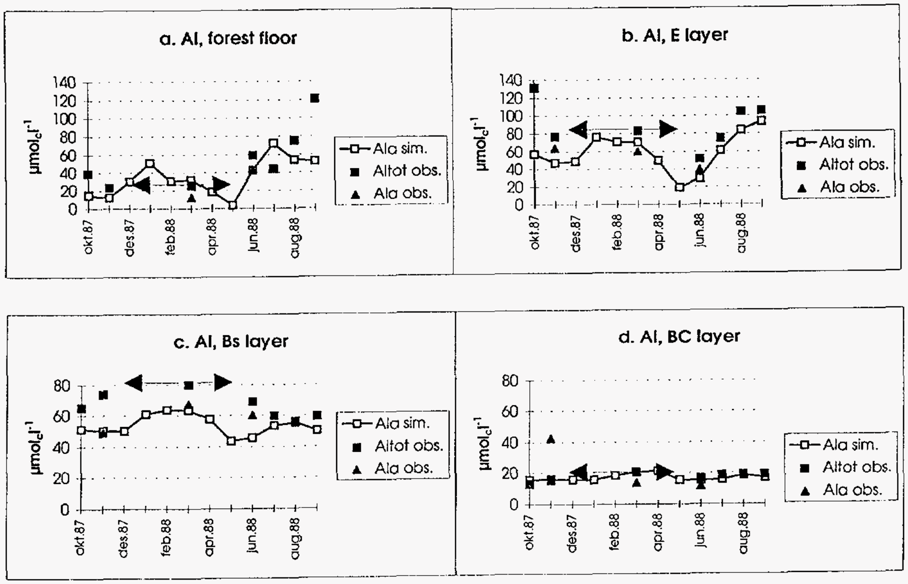

Fig. 10. Simulated and observed total Al and monomeric Al (Ala) concentration in soil water at the P30 plot, Nordmoen, October 1987- September 1988. a. forest floor. b. E horizon c. Bs horizon and d. BC horizon. The frost period (December 1987 - May 1988), is indicated by a horizontal arrow, with the lysimeter sample collected at the end of May (centre of arrow) representing the whole period (continual tension applied to lysimeter).

Organic Al (Alo) in soil solution was simulated considerably lower than measured in the O, E and Bs horizons, which in turn means more monomeric inorganic Al (Ali; Ali =Ala - Alo) is simulated than actually present in the O and E horizons. Probably Alo is simulated unrealistically low because the definition of organic Al in NuCM is limited to the complex, $Al^{3+}R^{3-}$, when in reality Al^{3+} is likely to complex with all ranges of partly dissociated organic acids (Lundstrøm, 1993; Taugbøl, 1993).

Measured DOC was only determined by ICP (Ogner et al., 1991), which has a low accuracy. Originally mean DOC values (Dec 87-Sept 88) were used as initial parameter values for all layers. However calibration of initial pH in NuCM is largely determined by the DOC concentration, the pKa values and CO_2 pressure in the soil, in that order of ranking for the pH range found at Nordmoen. The higher the DOC concentration, the lower the pH. Although the net result of buffering reactions such as weathering, gibbsite dissolution and cation exchange reactions also affect pH, calibration of pH by these reactions is ruled out by the involvement of too many parameters and interactions. By calibrating the pKa values to buffer over a wider pH range, it was possible to use the observed mean DOC concentrations as initial parameters in the O and E horizons. In the B horizons, the initial DOC parameters had to be calibrated to half of the observed mean DOC concentrations in order to keep the pH simulated for these layers at a higher level. Thus this compromise calibration in the B horizons causes both pH and DOC to be simulated too low. Despite the pH being simulated 0.3 units lower than observed pH in the B horizons, the pH NMAE is low in all horizons, because of pH being a logarithmic scale. The DOC NMAE is satisfactory in the top two layers and high in the B horizons, due to the pH/DOC compromise described above. In the DOC vs time plot (Fig. 11), the measured and simulated data are compared. The dynamics in the top two layers were well captured by the model, but this was only possible by calibrating adsorbed organic C (Table 2) and hence DOC adsorption on soil to a minimum. Despite the unreliability of measured data, other studies also show more variable DOC concentration in the higher layers (Easthouse et al., 1992; Zech et al., 1994; Raastad and Mulder, in press). The lower B horizons were poorly simulated at one flat level, showing none of the dynamics of the measured data. This is despite the reduction in the adsorbed organic C parameter to one tenth of the value of C measured there.

Dissolved organic C in soil water either comes from canopy leachate, is leached direct from litter, produced during litter decomposition or is actively exuded from roots. So the results of the DOC calibration, should reflect the description of the above processes in NuCM. Neither root litter production/decay nor root exudation is included in NuCM. Fine root turnover is a very important contribution to carbon and other nutrient cycles (Vogt et al., 1986), yet as root data is so scarce both to formulate decay process and use as input data, many models exclude root litter production and decay. The present description of litter decomposition has not only a large effect on the DOC concentration but also concentrations of other nutrients in the forest floor. However several aspects of the present description are unsatisfactory. The definition of the coarse litter fraction of the forest floor, which was defined as one year of needle litterfall here (Table 3; Fig.12) can be difficult. Needles can take several years migrating through the field layer or fall direct to the soil dependent on ground vegetation (Edsberg, pers. comm.). The initial nutrient contents of the different litter fractions in the forest floor could be more accurately initiated by using measured total contents of the different layers, rather than the current method (see Fig. 12) (Johnson et al., 1993).

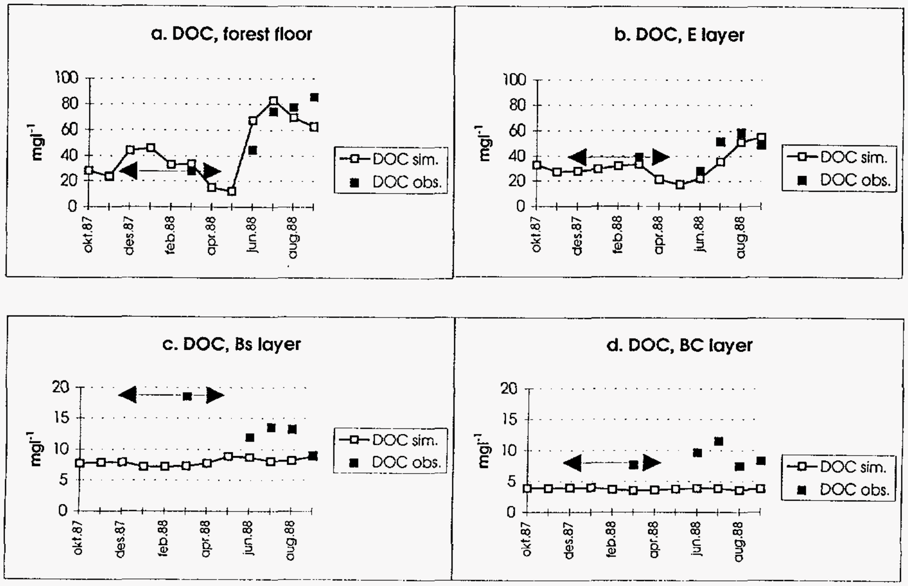

Fig. 11. Simulated and observed DOC concentration in soil water at the P30 plot Nordmoen, October 1987- September 1988. A. Forest floor. B. E horizon c. Bs horizon and d. BC horizon. The frost period (December 1987 - May 1988), is indicated by a horizontal arrow, with the lysimeter sample collected at the end of May (centre of arrow) representing the whole period (continual tension applied to lysimeter).

The processes involved in NuCM are summarised into 4 classes after this calibration (Table 11). Although there would be many advantages in simplifying canopy process description in NuCM, foliar exudation proved to be described adequately. NuCM successfully describes the interactions of SO_4^{2-} in the soil, while base cation and Cl^- concentrations in soil water are simulated adequately. The description of processes involving nitrogen needs improvement, both in the canopy, where N retention must be included and in the soil, where more flexibility is needed in the nitrification parameters. Also immobilisation of N in the soil by microbes should be included. Although adding to complexity, a realistic nutrient cycling model should include root litter production, decomposition and exudation.

Table 11. Classification of NuCM chemical processes, after this calibration

Process	Missing	Poor	Adequate	Surplus	Where commented*
dry deposition				x	5.2.1, p.2 & 3
foliar nitrification				x	5.2.1, p.2 & 3
foliar exudation			x	x	5.2.1, p.1, 2 & 3
foliar leaching				x	5.2.1, p.2 & 3
immobilisation of inorganic N in the canopy	x				5.2.2., p.2
nitrification		x			5.2.2., p.2
root litterfall and decay	x				5.2.2., p.9 & 10
root exudation	x				5.2.2., p.9 & 10
organic Al complexes		x			5.2.2., p.7
(de/ad)sorption of Al to organic matter	x				5.2.2., p.6
definition and calculation of physical & chemical components of litter layer		x			5.2.2., p. 9
weathering		x			5.2.2., p.4
soil processes modelling SO_4 & Cl			x		5.2.2., p.5
soil processes modelling base cations (except weathering)			x		5.2.2., p.4 & 5

* Section and paragraph(p) number refer to Results and Discussion.

6. Conclusion

The NuCM hydrology calibration gave adequate monthly values of soil water percolation during the snow-free months, but inadequate values during the winter months, due to poor snow simulation and problems simulating a frost layer. For this reason the results of a hydrology simulation with SOIL were used.

Throughfall chemistry was adequately calibrated for the base cations (foliar exudation). However measured inorganic N was roughly halved in the throughfall compared to total deposition yet despite the large number of processes describing throughfall in NuCM, no processes remove ions (except H^+/OH^- in charge balance).

The soil solution concentrations of Na^+, Mg^{2+}, K^+, SO_4^{2-}, Cl^- and H^+ could be calibrated adequately in NuCM . Inorganic N was not simulated well in NuCM due to the lack of a description of microbial immobilisation of inorganic N and inflexibility on user control of nitrification parameters. The high and uncertain NO_3^- concentration simulated by NuCM during the winter could also have caused the poor simulation of inorganic N. Ala could be calibrated reasonably in all horizons but E, where it was underestimated. Recent research has shown that Al solubility in the organic layers of soil is largely controlled by adsorption or desorption from organic matter and incorporation of this pool and process into the Al formulations in NuCM could improve future Ala simulations. Organic and inorganic monomeric Al (Alo and Ali respectively) were simulated poorly due to limiting the definition of organic Al in NuCM to the complex, $Al^{3+}R^{3-}$, when in reality Al^{3+} may also complex with partly dissociated organic acids. The DOC level and dynamics were simulated satisfactorily in the O and E horizons, once initial adsorbed DOC and organic acid breakdown rate were calibrated to a minimum and the humus to organic acid breakdown rate was calibrated to a maximum. Neither the DOC level nor dynamics were simulated well in the lower two layers.

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Appendix

Litter decay and nitrification

The forest floor is defined into three layers or components, coarse litter, fine litter and humus as shown in Fig. 12.

	Definition	**Chemistry**	**Reactions**
Coarse litter	needle litterfall in one year	needle chemistry	coarse litter $\rightarrow$ fine litter + cations
Fine litter	Oi horizon (2 cm, 0.02 g cm^{-3})	needle chemistry - cations	fine litter $\rightarrow$ humus + NH_4 + SO_4
Humus	Oe+a horizon (4.3 cm, 0.125 g cm^{-3}).	adsorbed cations, anions, CEC, CO_2 pressure	humus $\rightarrow$ organic acid + NH_4 +SO_4 +H + CO_2

Fig. 12. An overview of the definitions, chemistry and reactions occurring in the forest floor. (The definitions are for this calibration of P30 forest floor, Nordmoen).

These 3 components of litter are decayed following rate expressions with first order dependency on the mass of the reactant.

$$-d[\text{coarse litter}]/dt = \text{lbrate} \times 1.05^{(\text{soltmp}-12)} \times [\text{coarse litter}] \qquad 1$$

$$-d[\text{fine litter}]/dt = \text{fbrate} \times 1.05^{(\text{soltmp}-12)} \times [\text{fine litter}] \qquad 2$$

$$-d[\text{humus}]/dt = \text{hbrate} \times 1.05^{(\text{soltmp}-12)} \times [\text{humus}] \qquad 3$$

where lbrate, fbrate and hbrate are the decay rates of coarse litter, fine litter and humus respectively and soltmp is the monthly average soil temperature in the forest floor.

A fraction (frlch(1)) of immediately leachable ions is subtracted from the coarse litter mass and added to the solutes.

In NuCM, the fine litter decay rate to humus (fbrate*) is dependent on the C:N ratio in the fine litter (FCtoN). If the ratio of the input microbial C:N ratio (mcton) to the C:N ratio in fine litter(FCtoN) is less than 1, then the rate is reduced.

$$\text{fbrate}^* = \text{fbrate} \times [(\text{mcton})/(\text{FCtoN})] \qquad 4$$

Because S and N are partly mineralised when fine litter decomposes to humus, this fraction of NH_4 and SO_4 (frlch(2)), is subtracted from going to humus and added to the solutes. The humus C, N and S content is decomposed to organic acid (3). One can follow the calculation sequence for N. The ratio (RN) of N being oxidised from humus in a timestep (OXDN) divided by the amount of N in organic acid (oa1n) is used to govern the production of organic acid (5). This is the minimum of the ratios of N or C or S multiplied by the fraction of S and N that is incorporated into organic acid during humus decay (6).

$$RN = OXDN/oa1n \qquad 5$$

$$FA = DMIN1(RN,RC,RS) \times frlch(3) \qquad 6$$

The decay of organic acids occurs in all layers of the soil. This is programmed as a zero-order reaction, even though the manual implies it is first-order.

$$-d[organic\ acid]/dt = oarate \times 1.05^{(soltmp(L)\ -23)} \qquad 7$$

Nitrification, V, the concentration of NO_3^- produced / NH_4^+ consumed, is described by a Michealis -Menten type process, dependent on temperature and only occurring at pH greater than or equal to 2.

$$V = VMAX \times [NH_4] / (RKSNH4+ [NH_4]) \qquad 8$$

where

$VMAX = ckvnh4 \times (1.1^{(soltmp-20)}) \times timstp$
ckvnh4 = nitrification rate for soil (Table 3)
$[NH_4]$ = NH_4^+ concentration
RKSNH4 = half saturation constant, $[NH_4]$ where the rate of conversion is half the maximum = 0.1 (fixed in file BLOCKDAT.FOR)

Exchange processes and Aluminium dissolution

Cation exchange is modelled using the Gapon equation in relation to Ca^{2+} for 6 cations, Mg^{2+}, K^+, Na^+, Al^{3+}, H^+, NH_4^+ (Munson et al., 1992). Anion exchange for SO_4^{2-} is modelled as a pH dependent Langmuir adsorption isotherm (Johnson et al., 1993), while organic acid and phosphate adsorption is modelled as a pH dependent linear adsorption (Munson et al., 1992). These are calculated internally within the model from the initial parameter values for the adsorbed and solution concentrations of the ions. The equilibrium between different forms of Al,

$Al(OH)^{2+}$, $Al(OH)_2^+$, and $Al(OH)^{4-}$, AlR and organic acid, H_3R, H_2R^-, HR^{2-}, R^{3-} is governed by the input equilibrium constants (Table 5), pH, [Al] and DOC concentration (Munson et al., 1992). Aluminium can either be in equilibrium with gibbsite, $K_{SO} = [Al^{3+}]/[H^+]^3$, or the dissolution of Al can be rate limited.

$$-d[Al(OH)_{3(s)}]/dt = k\,([Al_e] - [Al]) \qquad 9$$

where

$[Al_e]$ = the concentration of Al^{3+} for a solution in equilibrium with $Al(OH)_3$ at the pH of the soil solution

k = the reaction rate coefficient

PAPER 4

Testing the Nutrient Cycling Model (NuCM) at Nordmoen, a Norway Spruce site in south-east Norway

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Abstract

The Nutrient Cycling Model (NuCM) is a process oriented research model simulating nutrient cycling in forest ecosystems. NuCM, after calibration to one year of data (companion paper), was tested on three years data from the same site. The concentrations of Mg^{2+} and K^+ in throughfall were simulated successfully on a monthly basis, while Ca^{2+} and pH were not. Despite NuCM's detailed description of canopy interactions, a mechanism to describe consumption of inorganic N in the canopy is lacking. Research quantifying dynamically the consumption and release of nitrogen in the tree canopy and the sinks involved is needed, before this process can be incorporated in NuCM.

In soil solution, NuCM simulated successfully the concentrations of SO_4^{2-} and Ala (total monomeric Al) in all horizons and H^+, Ca^{2+}, Mg^{2+} and Na^+ in the O and E horizons. Nitrate concentration in soil water was simulated too high in the dormant season, due to the simulated high nitrate concentrations in throughfall in winter, the lack of a description of N immobilisation by soil microbes and an inflexible nitrification description. The nutrient cycling fluxes of NH_4^+ and Mg^{2+} were simulated successfully, but cycling of Ca^{2+} was not. The incorporation of a simple description of translocation/accumulation to and from needles in NuCM would improve the simulation of Ca (and other nutrients) in fluxes of litterfall and uptake.

Given a scenario of 40% and 25% reduction in S deposition and nutrient uptake respectively, NuCM predicted soil exchangeable Ca^{2+} and K^+ of 590 and 280 $mol_c\ m^{-2}$ respectively. On the other hand, MAGIC predicted zero in both pools. When assuming no change in atmospheric deposition and nutrient uptake, NuCM predicted a much higher critical load than MAGIC. The reason for these different predictions, are large differences in the simulated rates of weathering and nutrient uptake.

Keywords: Nutrient Cycling Model (NuCM), testing, nitrate, calcium cycling

1. Introduction

Acid deposition constitutes a major environmental problem over large areas of Europe and N. America. Anthropogenic emissions of sulfur and nitrogen compounds lead to accelerated acidification of soil, surface water and groundwater (Zak et al., 1997). Model predictions provide a fundamental basis for emission abatement policy decisions in Europe. In Norway, the models MAGIC (Cosby et al., 1985) and PROFILE (Sverdrup and Warfvinge, 1993) have been most commonly used to predict the future effects of S and N deposition on forest ecosystems (Wright et al., 1991; Frogner et al., 1992). These models include important soil processes such as cation exchange, weathering and sulfate adsorption to predict on an annual basis solute concentrations in soil water and the pool of exchangeable cations in soil. However they do not include nutrient cycling processes like exudation, litterfall and mineralisation.

The Nutrient Cycling Model (NuCM) is a process oriented research model, which simulates the effects of atmospheric deposition on forest nutrient cycling by evaluating the movement and distribution of nutrients in the forest ecosystem. NuCM (Munson et al., 1992) was developed as part of the Integrated Forest Study (IFS)

(Johnson and Lindberg, 1992) and has been used to investigate the effects of S deposition scenarios (Liu et al., 1991; Johnson et al., 1993), to simulate base cation leaching (Johnson, 1995) and to simulate effects of harvesting and tree species change on nutrient cycling (Johnson et al., 1995). NuCM was also tested on a Norwegian lysimeter experiment where solutions with different N levels and pH were applied to bucket lysimeters containing small Scots pine trees under cover (Sogn et al, 1995b ; Sogn and Abrahamsen, 1997). However so far NuCM has not been thoroughly tested at the forest stand level under ambient conditions. NuCM, compared to MAGIC and PROFILE demands an extensive parameter set (94 parameters) and uses daily, weekly or monthly timesteps. Therefore NuCM is expected to perform better on an annual basis than these less complex models. In a companion paper, NuCM has been calibrated for a Norway spruce stand at Nordmoen, south-east Norway, using one year of data (Kvindesland, in review). Here the calibrated model will be tested using data for three other years from the same site.

The objectives are (1) to compare three years of simulated and observed throughfall and soil solution data, (2) discuss the results in light of simulated and estimated nutrient cycling fluxes and (3) predict the effect of a 40 and 90% reduction in S deposition for a 48-year period. Results will be compared with existing MAGIC predictions.

2. Method

The study site Nordmoen is located on a flat plain of glaciofluvial deposits 45 km north of Oslo at an elevation of 200 m. The soil is classified as a Cambic Arenosol (FAO) (Stuanes and Sveistrup, 1979). The P30 plot, which is a natural 30 year old Norway spruce (*Picea abies* (L.) Karst) plot, 1000 m^2 square in size, was chosen for the calibration and testing of NuCM. Four years of data were available from October 1986 to December 1990. The second year (1 Oct. 1987 - 30 Sept. 1988) of this period was used for calibration (Kvindesland, in review) to avoid lysimeter installation effects. Meteorological data was monitored daily at Nordmoen (April 1986- October 1989) and continually at the meteorological station, Gardermoen, 10 km south of Nordmoen. Wet and dry deposition was monitored at Nordmoen by the Norwegian Institute for Air Research (Statens forurensingstilsyn, 1987, 1988, 1989, 1990, 1991). Precipitation, throughfall and lysimeter chemistry were monitored monthly at plot P30 during the frost free period. Full details of the site, measurements and methods used can be found in Kvindesland and Røsberg, (in review). Presentation of the parameters used, calibration procedure and results of the calibration can be found in Kvindesland, (in review).

Three new input files were needed for the model test: a meteorology file containing daily precipitation, temperature, relative humidity, wind speed and cloudiness, a deposition file (AIR file) containing precipitation and ambient air concentrations and a hydrology result file (H2O file) needed to run the chemistry simulation. As during the calibration (Fig 3; Kvindesland, in review), a similar construction of the AIR and H2O files was used to model throughfall concentrations to circumvent problems associated with modelling N immobilisation in the canopy. This means that only the base cations and pH are properly simulated in throughfall (i.e. using wet and dry deposition and canopy processes). The concentrations of NH_4^+ and NO_3^- in throughfall and precipitation used in the AIR file are shown in Fig. 1. The shaded symbols for NH_4^+ and NO_3^- displayed at the end of each winter represent the average throughfall concentration collected in snow lysimeters or buckets for the whole frost period and were not used as inputs.

The hydrology was simulated by the SOIL model (Jansson, 1991) using the model parameters obtained by calibration (Kvindesland, in review).

The MAGIC model was applied at Nordmoen (Wright et al., 1991), using the 40 year-old adjacent stand, plot P40 (Kvindesland and Røsberg, in review) and first two years of the dataset (1986-1988). After calibration to the existing data, MAGIC was run to make predictions for the period (1988 - 2038) under four different scenarios. The NuCM model was run for 48 years (1986 - 1934), where the existing four years of hydrology and deposition data were repeated twelve times. Three of the original MAGIC scenarios were chosen so MAGIC and NuCM results could be compared:

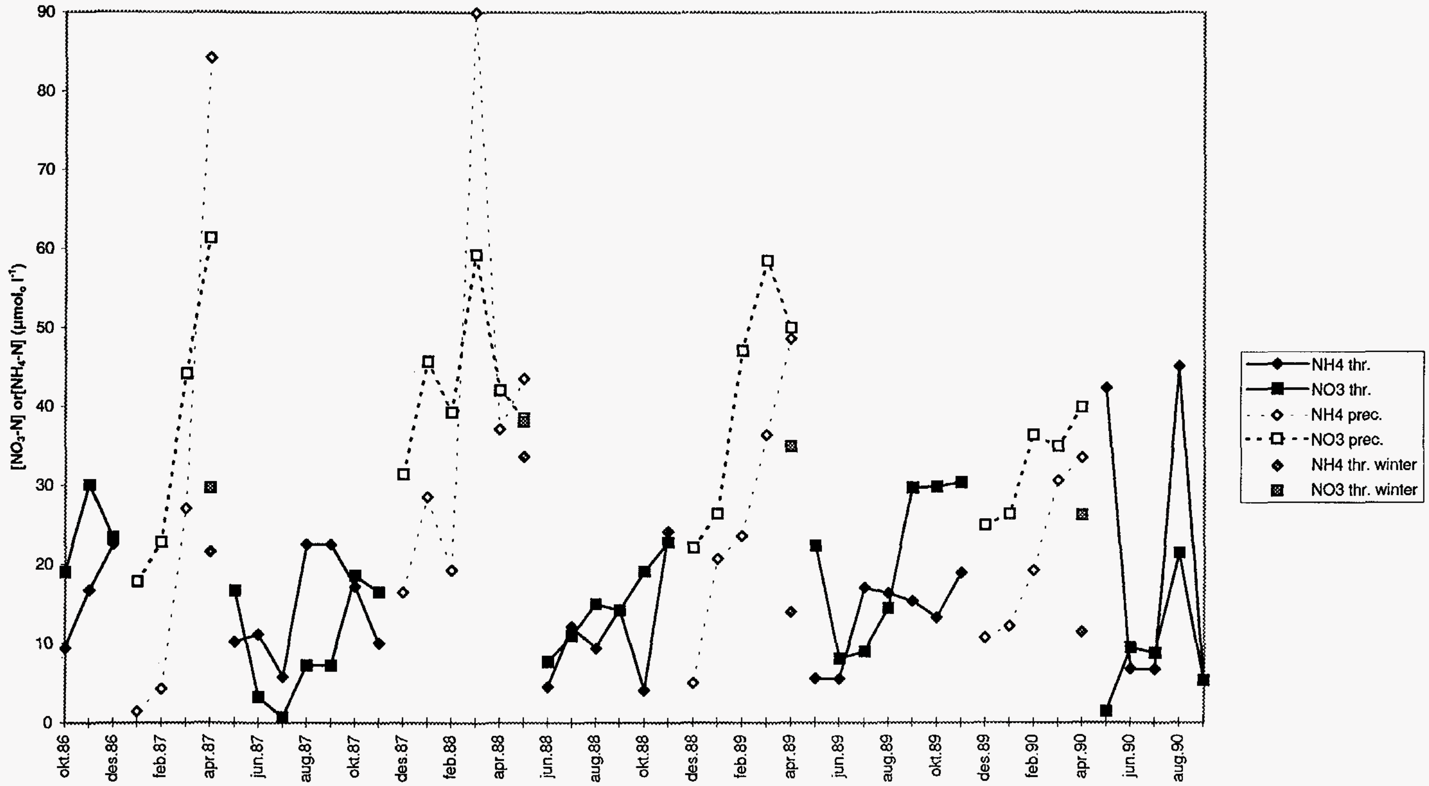

Fig.1. Nitrate and ammonium concentrations in throughfall and precipitation used in AIR file (throughfall concentrations in winter not used in AIR file, but shown for comparison).

1. a linear 40 % reduction in total S deposition between 1989 and 2005 and then constant at this new level to 2038 and linear 25% reduction in uptake over whole period (1989-2038) (40/25 scenario)
2. 90 % reduction in total S deposition between 1989 and 2005 and then constant at this new level to 2038 and linear 25% reduction in uptake over whole period (1989-2038) (90/25 scenario)
3. 90% reduction in total S deposition (as scenario 2) and abrupt 100% reduction in uptake (e.g. trees harvested in 1989, grass planted) (90/100 scenario).

Due to the differences in model structure, changes were made when simulating NuCM for the same scenarios. For scenario 1, total S deposition was reduced by 40 % in 1986 and was constant over the whole period (1986-2034) and likewise in scenarios 2 and 3, S deposition was reduced by 90% in 1986 and was then constant over the whole period (1986-2034). This causes the total S deposition in MAGIC to be 15 % higher than NuCM in scenario 1 and 130 % higher than NuCM in scenario 2 and 3. In scenario 3, tree growth was reduced by 90% in 1987 and then kept at this low rate until 2034.

2.1. General definitions

To evaluate the quality of the simulations, the normalised mean absolute error (NMAE) is used, which is defined according to:

$$MAE = \frac{\sum_{i=1}^{N} |S_i - O_i|}{N}$$

$$NMAE = \frac{MAE}{\overline{O}}$$

Where S_i and O_i denote the simulated value and observed value, $\overline{O}$, the observed sample mean and N, the number of simulations/observations. The NMAE accounts for the spread in the deviations, S_i - O_i. The closer to zero, the better the simulation.

In the simulations, four years were considered from October 1986 until September 1990. The second year (Oct. 1987 - Sept. 1988) was the calibration year. To simplify description of the throughfall and soil solution results, the word level refers to the average concentration of a solute, while dynamics refers to the monthly changes in concentration over the four year period.

3. Results and Discussion

3.1. Recalibration

After running the calibrated model (Kvindesland, in review) for the four years with observations, it became clear that the simulations were unsatisfactory for a number of elements. Therefore it was decided to recalibrate the model. The calibrations needed to improve the simulations include:

a. Before recalibration, the simulated uptake flux of inorganic N and Al was 2177 and 212 mol_c ha^{-1} $year^{-1,}$ respectively. Both fluxes are simulated too large, the N flux being 1.5 times the estimated uptake. After rechecking parameters involved in uptake, it became clear that the chemical contents of roots refers to coarse structural roots and not fine roots as I had used during calibration. This is because NuCM includes no root litter production and so any element taken up for root growth stays there until the tree dies. The word 'roots' is used throughout the NuCM menu inputs and manual (Munson et al., 1991) and should be changed to 'coarse structural roots' to avoid misunderstandings. The chemical composition of roots (Table 7. Kvindesland, in review) were changed as shown in Table 1 and the root biomass from 2.9 to 0.65 kg m^{-2} (coarse root biomass (O-BC horizon) >1 cm in diameter (Røsberg, pers. comm.)) .

Table 1. The chemical composition of coarse roots at Nordmoen (> 1cm) (mg g^{-1}) (Røsberg, pers. comm.)

N	Al	Ca	Mg	K	Na	S	P
2.72	0.31	2.82	0.38	2.57	0.04	0.26	0.63

b. The annual mineralisation of all elements in the first year was simulated a little lower than the annual litterfall flux, which was also observed under calibration (Kvindesland, in review). However in the next three years, the mineralisation flux was simulated higher than the average litterfall flux for all components. This is clearly unrealistic as the following observations underline. Both the thickness of the humus (Oe+a) horizon and its content of total C and N have been observed to increase over a 100 year period at Nordmoen (Sogn et al., in review). Generally N accumulation in Norwegian forest soils is estimated in the interval 0.2 - 1.7 g N m^{-2} year $^{-1}$ (Sogn et al., 1995a). During the original calibration, the organic matter decay rates were calibrated to match the high concentrations of DOC concentration in the O and E horizons. In this recalibration, both the coarse litter decay rate (lbrate) and humus decay rate (hbrate) (Table 3; Kvindesland, in review) were decreased to 1.00 and 0.04 respectively.

c. The concentrations of all elements in needles and hence litterfall were simulated to decrease during the four year period however observations of element concentrations in needles from P30 showed no decreasing trend between 1986-1988 (Røsberg, pers.comm.). The simulated decrease was due to a limitation in N uptake, which was in turn limiting uptake of the other elements into the needle biomass. The uptake distribution for NH_4 between horizons (Table 2; Kvindesland, in review) was therefore recalibrated in order to reduce the number of months when roots experienced N deficiency in the mineral horizons of soil and hence limitation of N uptake occurred. Before 90% of NH_4 was taken up in the O horizon, and 5% in both the E and Bs horizons. Now 97.5% is taken up in the O horizon, 0.25 % in the E horizon and 2.25% in the Bs horizon.

Recalibration of the calibrated model was necessary after using one year of data for calibration (Kvindesland, in review). Ideally many years should be used for calibration (which normally are not available). However if only using one year for calibration, it is very important to simulate 5-10 years ahead and check in particular the cycle fluxes. Using this method, points a and b could have been picked out as strange results, without needing to calibrate against measured data. Although solute concentrations in throughfall and soil solution and soil exchangeable pools were checked in a 10 year run during the calibration (Kvindesland, in review), cycle fluxes were not due to the time such a check would take. Presentation of cycling results in NuCM are as individual diagrams per element per year and hence time trends are not obvious, without the considerable work involved in exporting and reorganising data in a worksheet.

3.2. Hydrology

The simulated annual hydrological budget is presented in Table 2. It is important to note that the first two years are wet (precipitation > 130 cm), the third year is drier (105 cm) and the last year dry (90 cm).

Table 2. Annual hydrological budget simulated by SOIL (all values in cm). Hydrological years from October 1st - September 30th.

period	precipitation	evaporated interception	throughfall	actual evapotranspiration	water flux O & E	water flux Bs	water flux BC	water flux out C
86-87	132	27	105	44	84	72	69	61
87-88	134	21	113	32	93	82	80	81
88-89	106	20	86	39	64	52	50	47
89-90	91	19	72	36	52	43	42	36

The dry summers of 1989 and 1990 cause SOIL to simulate low moisture contents and water fluxes in the lower B horizons. This causes high simulated concentrations of NO_3^- and Cl^-, which lead to high simulated concentrations of Ca^{2+}, Mg^{2+}, and Ala (Figs. 2, 3 and 4) occurring in these horizons during this period. Observed concentrations of these ions are not so high. Van Grinsven, (1988) noticed the same phenomenon with high peaks of Cl^- in late summer simulated by ILWAS in the lower mineral horizons of a soil in the Netherlands and yet no general observation of such high peaks in the field. He suggested that capillary rise of dilute groundwater could cause the lower observed concentrations, a process not included in SOIL. Lysimeters could also be pulling water from wetter areas lower down, suggesting that lysimeter samples are not always representative during dry periods In addition there are fewer observations in these dry periods because the lysimeters do not always yield enough water for analysis of anions.

3.3. Throughfall and canopy cycling fluxes

The only elements in throughfall truly simulated by NuCM are the base cations, Ca^{2+}, K^+ and Mg^{2+} and H^+. As in the calibration, the other elements were input as observed throughfall concentrations. The statistical test gives Ca^{2+} , Mg^{2+} and K^+ NMAEs of 0.6, 0.2 and 0.1 respectively (Table 3). Visually (Fig. 5), the simulated K^+ concentration compares well dynamically with the observed values, except in the spring of the third and fourth years and this is why the simulated mean is lower than the observed. The simulated Ca^{2+} and Mg^{2+} concentrations only sometimes fit the dynamics of the observed values, although the Mg^{2+} level of observed and simulated agree well.

Foliar exudation modifies concentrations of base cations and H^+ in throughfall. Exudation provides 75%, 90% and 65% of the Ca^{2+}, K^+ and Mg^{2+} flux in throughfall. Dry deposition (<0.5 %) and wet deposition provide the rest (Table 4). Foliar exudation is controlled by a common monthly exudation rate for all ions and foliar amplification factors for each ion. It appears difficult to simulate Ca^{2+} at a monthly timestep as the exudation pattern varies widely between years, while K^+ and Mg^{2+} can be simulated reasonably at this timestep. NuCM balances the exudation of positive base cations by exuding OH^- ions, which causes substantial increases in simulated pH (Fig. 5). As discussed previously (Kvindesland, in review), NuCM does not include any canopy processes which consume inorganic N or produce organic N as was observed at Nordmoen (Kvindesland and Røsberg, in review). The production of organic anions and uptake of NO_3^- would help balance the exudation of positive ions. The average estimated throughfall flux for the four year period (Table 4) compares well for the simulated throughfall fluxes of K^+ and Mg^{2+}, but is overestimated for Ca^{2+}. The simulated exudation flux is over double the estimated exudation flux for Ca^{2+} and about double for Mg^{2+}, while the K^+ exudation flux compares well.

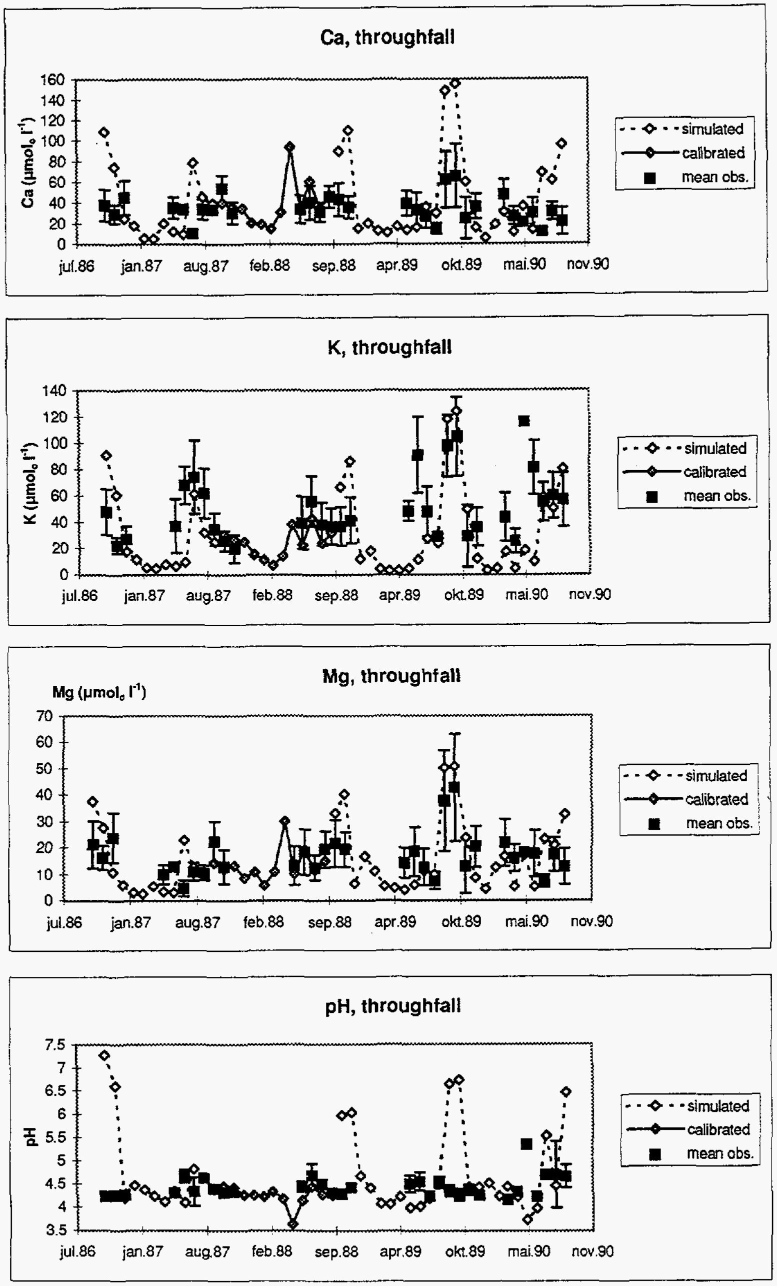

Fig. 5. Simulated, calibrated and observed concentrations of calcium, potassium and magnesium and pH in throughfall for the period (Oct. 1986-Sept. 1990).

Table 3: Simulated and Observed Mean and the Normalised Mean Absolute Error (NMAE) of throughfall and soil water for the period October 1986-October 1990 (excluding October 1987-October 1988). (Units μmol$_c$ l^{-1}, except DOC mg l^{-1})

	NO_3-N	NH_4-N	Ca	K	Mg	SO_4-S	Cl	Na[1]	pH	sim. Ala[2]cf. obsAlt	sim.Ala cf.obs Ala	Alo[2]	DOC
thrfall sim.			42	32	16								
thrfall obs.			32	47	16								
NMAE			0.6	0.1	0.2								
O sim.	55	2.7	174	16	44	127	128	45	4.06	79	79	5	37
O obs.	4	15.4	152	65	41	114	65	53	4.10	87	42	41	84
NMAE	6.1	0.8	0.1	0.8	0.0	0.1	0.6	0.0	0.0	0.1	0.1	0.9	0.5
E sim.	56	2.5	133	13	35	89	106	39	4.30	98	98	9	25
E obs.	5	3.1	134	37	36	103	55	69	4.24	115	121	101	41
NMAE	7.5	0.3	0.1	0.6	0.1	0.1	0.4	0.2	0.0	0.2	0.5	0.9	0.4
Bs sim.	42	1.9	92	10	38	107	99	51	4.44	66	66	3	8
Bs obs.	4	1.8	47	10	22	120	54	63	4.75	62	58	9	14
NMAE	11.4	0.0	0.9	0.0	0.7	0.1	1.0	0.1	0.1	0.0	0.1	0.7	0.4
BC sim.	60	2.0	71	11	46	79	96	73	4.66	41	41	1	3
BC obs.	5	2.2	52	9	32	101	80	89	5.32	18	9	2	9
NMAE	10.4	0.1	0.4	0.3	0.4	0.2	0.0	0.1	0.1	1.4	1.0	0.5	0.7

[1] NMAE period begins in May 1987 rather than Oct 1986 to avoid installation effects.

[2] Ala = total monomeric Al, Alt = total monomeric and polymeric Al, Alo = organic monomeric Al

Table 4. Average annual fluxes (all mol_c *ha*$^{-1}$ *year*$^{-1}$ *except Ala and DOC mole ha*$^{-1}$ *year*$^{-1}$*) simulated by NuCM compared to estimated fluxes in the same period, October 1986 - Sepetember 1990.*

element	wet deposition	dry deposition	exudation	throughfall	snowmelt	total uptake	litterfall	mineralisation	leaching	CEC/AEC change	weathering
NH_4 sim.[a]	232	4		235	174	1574	1375[d]	1196	9	-154	
NH_4 calc.[b]				116		1408[c]	1258		6		
NO_3 sim.	277	2		278	225	109			165		
NO_3 calc.				151		1408[c]			11		
Ca sim.	84	0	243	322	117	768	332	321	346	-139	323
Ca calc.			92	211		1033	664		211		360
Mg sim.	42	0	78	118	51	267	122	116	196	34	260
Mg calc.			39	96		262	105		132		180
K sim.	27	0	207	230	75	470	173	168	53	-66	58
K calc.			211	239		468	98		30		40
Na sim.	230	1		229	132	7	4	4	349	37	150
Na calc.				193							70
SO_4 sim.	681	4		681	374	101	92	52	572	92	
SO_4 calc.				477							
Ala sim.	8			7	2	16	11	11	50	136	184[c]
DOC sim.	208			206	42			253	59	407	

[a] sim. denotes simulated by NuCM

[b] calc. denotes calculated fluxes, where both data and methods are described in Kvindesland and Røsberg, (in review).

[c] Sum of inorganic N uptake

[d] NuCM tracks N in organic matter as NH_4.

[c] Al from gibbsite dissolution.

Despite all the efforts to minimise inorganic N in throughfall, the simulated fluxes for both NH_4^+ and NO_3^- are more or less double that calculated (Table 4). About 20% of the nitrate flux and 30 % of the ammonium flux entering the soil in the four years stem from throughfall during the growing season, while the rest enters during the dormant season. Throughfall flux is used as model input during the growing season (Fig 3; Kvindesland, in review), but during the dormant season measured ambient air concentrations and precipitation concentrations are used to simulate total deposition and throughfall (Fig. 1). As dry deposition was calibrated to a minimum, the measured elevated concentrations of N in precipitation in winter are the main cause of the overestimated fluxes (Fig. 1). In NuCM all snow falling on the canopy is routed directly to the soil and no interaction occurs with the canopy at all. However in reality snow and frost can rest on trees for several days. As inorganic nitrogen is consumed in the canopy during the growing season, it is possible the same can happen during the dormant season. It is also possible that the calculated throughfall flux has been estimated too low, i.e. too low measurements of winter chemistry and water volume. This is due to the practical problems of collecting throughfall as snow. Throughfall collectors were reduced from 20 summer collectors to two snow lysimeters, two snow boards and five plastic buckets (beginning winter 1987-1988). In an unstable winter, snow lysimeters do not function so well, snow can blow off full buckets and the first snow melt, which generally has the highest ionic strength can be lost.

3.4. Soil Solution and cycling fluxes

The observed NO_3^- concentration in all horizons is low, 4 - 5 $\mu mol_c\ l^{-1}$ and shows only the slightest tendency to increase during winter. The simulated concentration in the O (Fig. 2) and E horizon peak highly in the winter and decrease to observed concentrations in the growing season. The simulated concentration in the B horizons also peak in winter and sink during the growing season but in the last two drier summers they are at a generally raised level (see 3.2). The NMAEs for all four layers are very high, 6-7 in the upper 2 layers and 10-11 in the lower B layers. The observed NH_4^+ concentration is generally low in all horizons. However the first half year the mean observed concentration is quite high in the O horizon (Fig. 2), probably due to an installation effect and this causes a high NMAE (0.8) and poor agreement between simulated and observed means in this horizon. The simulated concentration tends to rise gently in the winter and sink gently in the summer. The dynamics are simulated best in the O horizon and more poorly in the lower three horizons, although the average concentrations of observed and simulated compare well in the mineral horizons.

The nitrate concentration in the dormant season is simulated poorly in all horizons. One of the main reasons could be the NO_3^- flux in throughfall during the winter, which is probably overestimated by NuCM (see 3.3). Soil water samples were generally not collected in the winter, but the few that were and those collected at snow melt do not suggest any large increase in NO_3^- concentration. However, the possibility that water with high NO_3^- concentrations has bypassed cannot be dismissed. Frozen lysimeter tubing disabling water collection yet water transport of early snow melt in macropores in thawing soil could be a partial explanation for the difference. It also seems likely in a soil with a high C:N ratio that immobilisation of N by both microbes (a process not described in NuCM) and possibly roots (tree uptake put to zero in winter) could occur even at low temperatures. In the summer, simulated NO_3^-concentrations reduce to a low level in the O and E horizons every year and in the mineral horizons the first two years. In fact the level is so low that NuCM opts to take 95% of its N uptake as NH_4^+ and only 5 % as NO_3^- (Table 4). I chose to calibrate with zero nitrification rate to avoid very high winter concentrations. The nitrification rate in NuCM is at present very inflexible (Kvindesland, in review) as the half saturation constant is fixed at a low 0.1, causing a large increase in nitrification on slight increase in the NH_4^+ concentration. Even though the simulated throughfall flux of NH_4^+ is twice as high

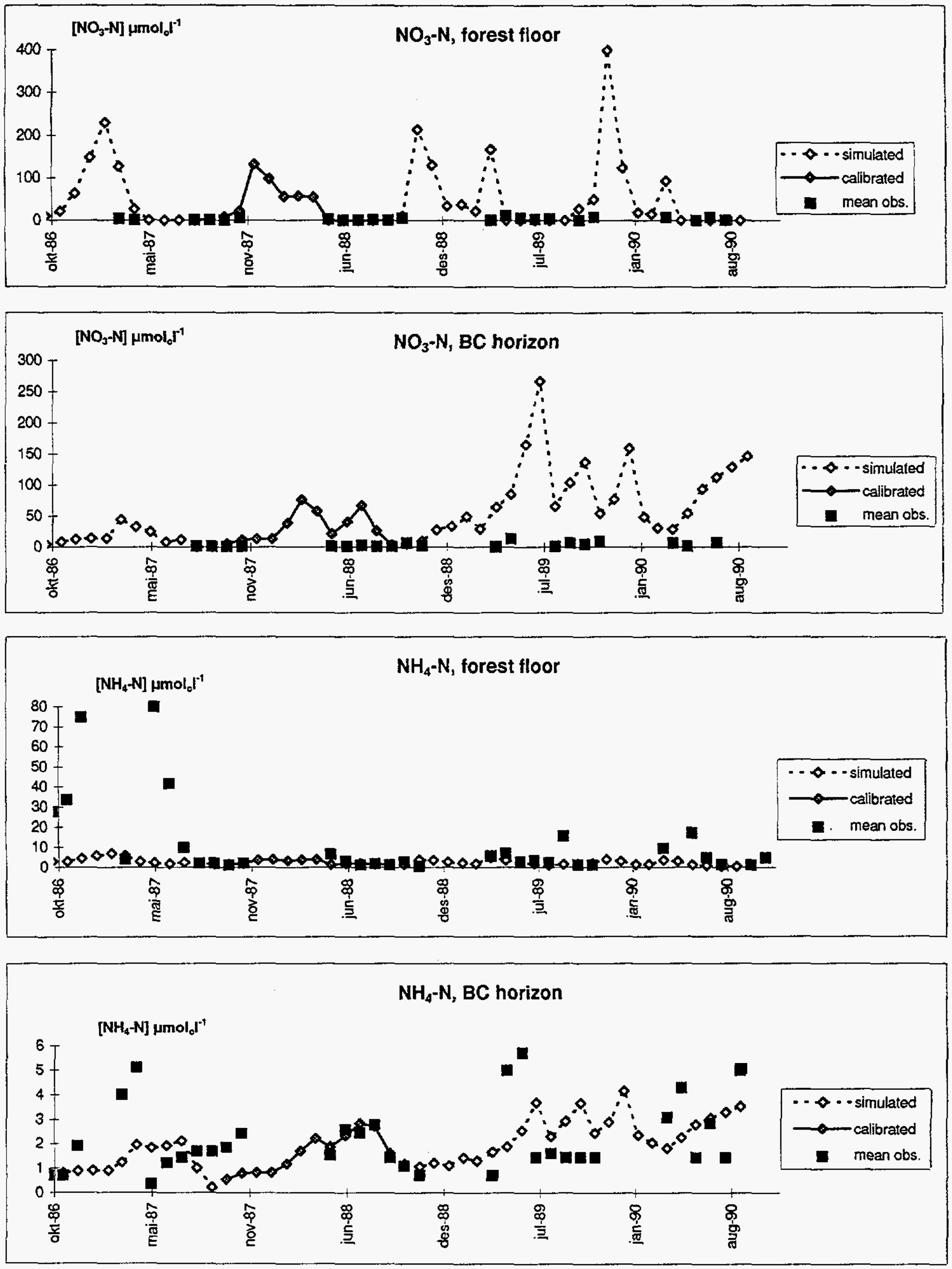

Fig. 2. Simulated, calibrated and observed concentrations of nitrate and ammonium in soil solution under the forest floor and BC horizon for the period (Oct.1986-Sept. 1990).

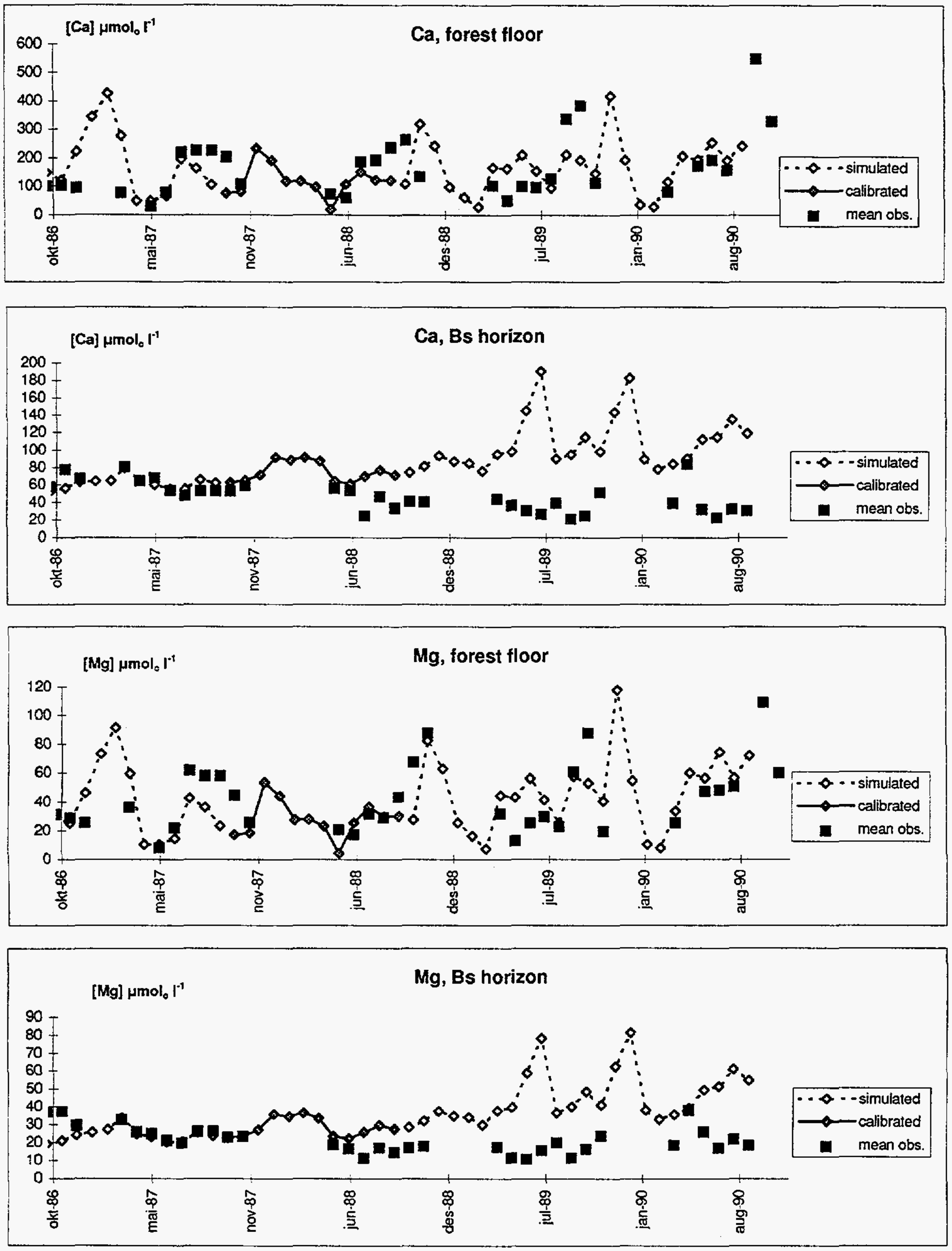

Fig. 3. Simulated, calibrated and observed concentrations of calcium and magnesium in soil solution under the forest floor , and Bs horizon for the period (Oct. 1986-Sept. 1990).

as measured, the NH_4^+ concentration is simulated more successfully than NO_3^- because cation exchange (winter) and uptake (summer) keep the level in check. The total simulated N uptake compared well to the estimated uptake, and both simulated litterfall and leaching fluxes compared well to the observed fluxes (Table 4).

The observed Ca^{2+} concentration is highest and most dynamic in the O horizon (Fig. 3) and decreases in concentration and dynamics down the soil profile. The simulated Ca^{2+} in the O and E horizons appears reasonable from 1986-1988, summer 1989 and not so good in either the spring or autumn 1989. Simulated Ca^{2+} in the Bs horizon compares well to observed Ca^{2+} concentrations the first year but then increases each year to a level far above the observed concentration. This is reflected in the high NMAE for Bs (0.9). This naturally affects the simulated Ca^{2+}concentration in the BC horizon, which is too high during the last two years. Simulated Mg^{2+} concentrations in the O (Fig. 3) and E horizons compare visually well with observed, both on levels and dynamics and the NMAEs are low. Simulated and observed Mg^{2+} concentrations in the B horizons behave very similarly to Ca and the NMAEs are also similar. The Ca^{2+} and Mg^{2+} concentrations in soil solution were simulated reasonably in the upper two horizons. However in the B horizons, the last two dry summers (see 3.2) and the overestimation of Mg^{2+} weathering compared to the calculated flux caused poor simulation of both Ca^{2+} and Mg^{2+} concentrations.

The observed concentration of K^+ in soil solution appears far more dynamic than that simulated in all horizons. This lack of dynamics in simulated concentrations suggests that adsorption of K^+ onto the CEC is probably simulated too strongly by NuCM. The simulated K^+ concentration in the O horizon (Fig. 6) is reasonable for the first year and the second (calibration) year but poor the two last drier years. Both the NMAEs in the upper two horizons are high (>0.5). The B horizons show a reasonable simulated level, reflected in the low NMAEs, but the simulated dynamics does not follow the observed dynamics.

The fluxes of Ca^{2+}, Mg^{2+} and K^+ simulated by NuCM in litterfall, uptake and mineralisation must be discussed in relation to process formulations. NuCM uses current needle content and litterfall mass per month to calculate the flux of nutrients in litterfall. But elements like K, N, P and Mg are translocated from older to newer needles (Miller, 1995), while Ca is not translocated and therefore tends to accumulate as the needles age (Kvindesland and Røsberg, in review).

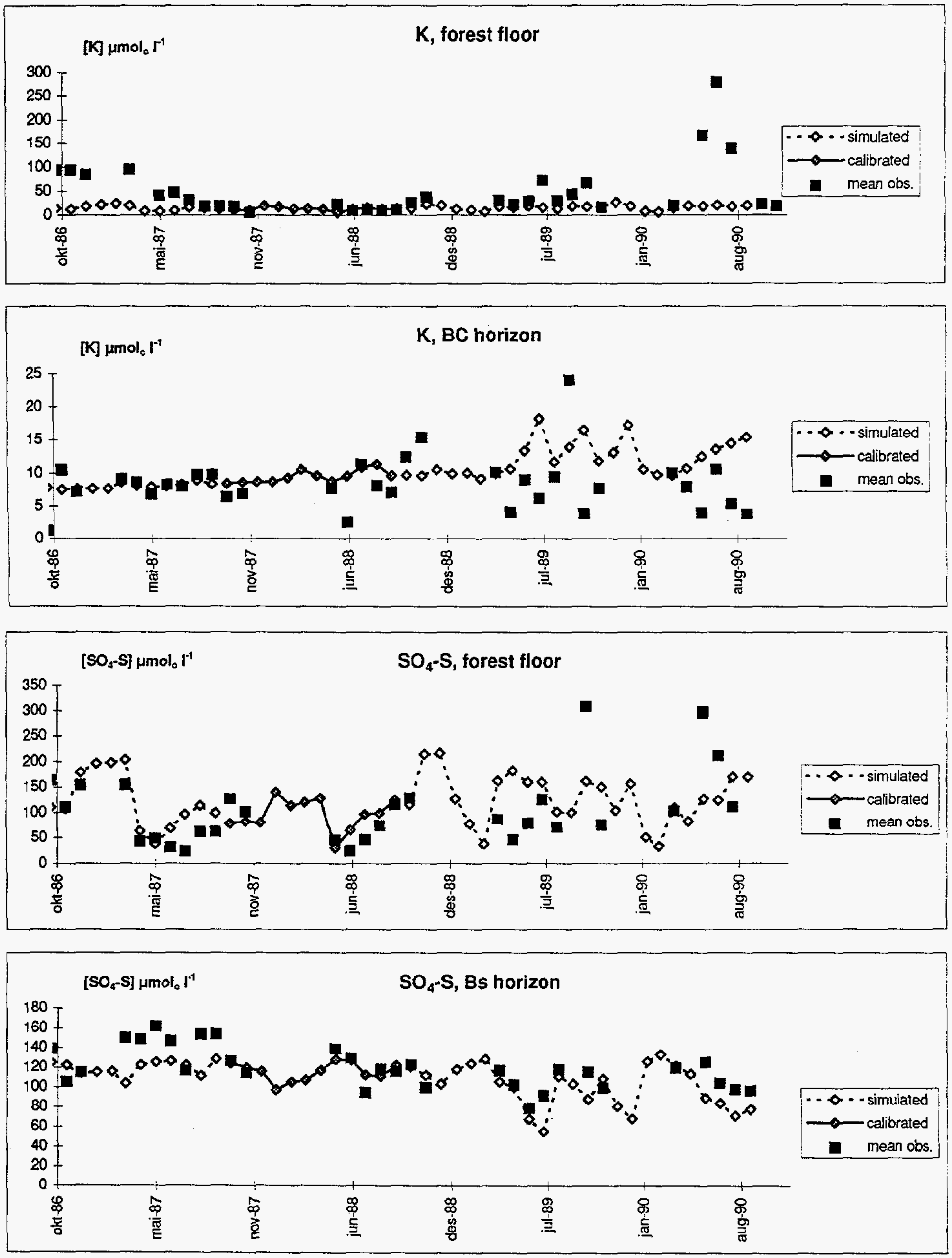

Fig. 6. Simulated, calibrated and observed concentrations of potassium and sulfate in soil solution under the forest floor, Bs and BC horizons for the period (Oct. 1986-Sept. 1990).

NuCM includes translocation for deciduous trees but not coniferous trees. The lack of translocation/accumulation for coniferous trees in NuCM is a serious shortcoming for a model trying to simulate nutrient cycling, as the litter input to soil will be in error for several elements. The translocation occurring between the 7 needle classes of spruce requires a plant physiological model (Mohren and Van der Veen, 1995) and is too ambitious for NuCM. However it is important to at least include the net difference between element content in current and newly fallen needles, so both uptake and litterfall are simulated correctly. As a result of the absence of translocation processes, the simulated Ca flux in litterfall is too low, the K flux is too high and the Mg flux is slightly overestimated (Table 4). NuCM bases uptake on annual needle, bole and root contents and exudation. The exudation of Ca^{2+} was simulated too high, which in turn increases the simulated uptake of Ca^{2+}. Despite this, the simulated uptake of Ca^{2+} is still too low compared to the calculated uptake. Although the low simulated Ca^{2+} fluxes of litterfall and uptake balance each other, this in turn causes a lower Ca^{2+} flux from mineralisation of litter. Therefore to account for the high observed Ca^{2+} concentrations in the O horizon, NuCM strips more exchangeable Ca^{2+} from the soil, whereas probably this additional Ca^{2+} is mineralised. Mineralisation of K^+ is overestimated due to the overestimated litterfall flux. However K^+ is strongly adsorbed on the cation exchange sites and this soaks up some of the extra flux, while the rest is leached.

Timeplots of simulated versus observed SO_4^{2-} concentrations in all horizons (Fig. 6) show good agreement both on level and dynamics. This is also reflected in the low NMAEs (< 0.2). The concentration of SO_4^{2-} in soil solution is simulated well in all horizons. The important SO_4^{2-} fluxes are throughfall (overestimated by NuCM), leaching and adsorption. Successful SO_4^{2-} simulation by NuCM was also achieved on similar Norwegian soil with low S deposition (Sogn and Abrahamsen, 1997).

The observed Alt (total monomeric and polymeric Al) concentration in forest floor (Fig. 4) tends to peak in late summer/ autumn. Simulated Ala (total monomeric Al) versus observed Alt shows good agreement in the O, E and Bs horizons, where the NMAEs are less than or equal to 0.2. The BC level is simulated too high in the last two years, which causes a high NMAE (1.4). Only limited observed data for Ala and Alo (organic monomeric Al) exists. Simulated Ala fits well with the data observed in the O horizon for level and dynamics and in the Bs horizons for level. In both the E and BC horizons, Ala is simulated poorly compared to observed Ala and the NMAEs are high. Organic Al is not simulated well in any layer and especially poorly in the two top layers. All NMAEs are greater than or equal to 0.5. The simulated concentration of Ala, when compared with Alt and the short period of observations of Ala (Fig. 4), was surprisingly well simulated in all horizons but BC. As already described (see 3.2), the last two dry

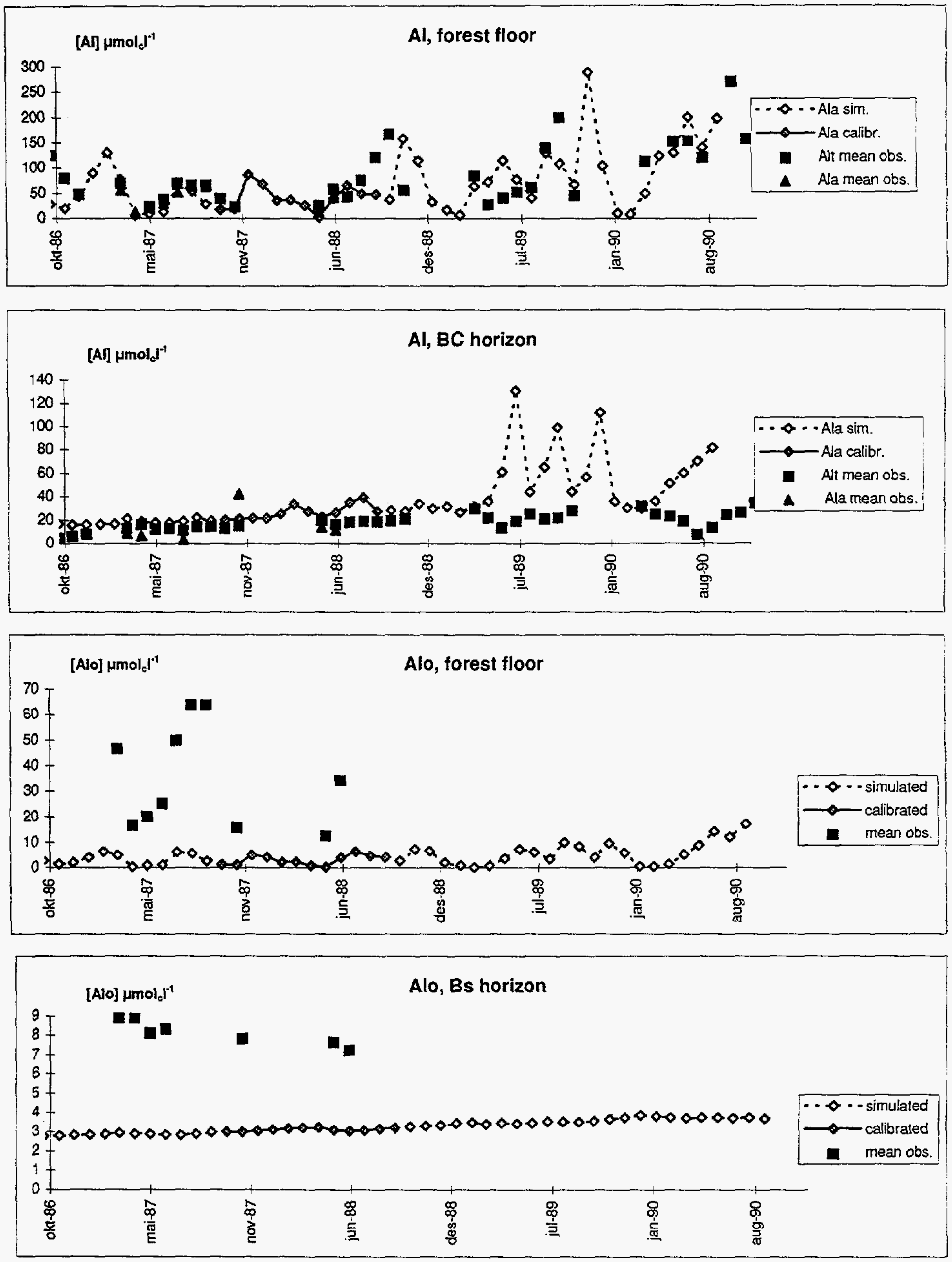

Fig. 4. Simulated, calibrated and observed concentrations of different forms of aluminium in soil solution under the forest floor , and Bs and BC horizons for the period (Oct. 1986-Sept. 1990).

summers led to too high concentrations here. The very poor simulation of Alo concentration is a consequence of limiting the definition of organic Al in NuCM to the complex, $Al^{3+}R^{3-}$ (Kvindesland, in review).

The time plots of simulated and observed pH for the O (Fig. 7) and E horizons show good agreement for both level and dynamics. The pH is tending to increase at snow melt and then decrease during the summer and winter. In both the Bs and BC horizons pH is simulated too low.

The average concentration of DOC was simulated too low in the O and E horizons (Fig.7), although the dynamics were simulated well. In the B horizons, due to calibration problems trying to balance high observed DOC concentrations with high observed pH (Kvindesland, in review), a compromise calibration was chosen, causing lower simulated DOC and pH than observed. The observed DOC concentrations in the B horizons were far more dynamic than the steady simulated levels. The main DOC production in NuCM is by mineralisation of needle litter. Before recalibration, the DOC level in the O and E horizons was simulated successfully (Kvindesland, in review), because coarse litter and humus decay were calibrated to high rates precisely to produce enough DOC. This was not realistic for any of the other elements as the annual mineralisation flux was larger than the litterfall flux, hence the recalibration. There are several explanations for the low simulated level of DOC compared to DOC measured in soil water at Nordmoen. (1). The DOC has another source and this pool and its production of DOC is not described in NuCM, e.g. root exudation and decay of other organic matter. (2). The process description of DOC production from litter plus its solubility are not satisfactory. The main consumption of DOC in NuCM is by adsorption. Sorbed DOC was simulated to build up in all horizons. During the calibration, adsorbed concentrations of DOC had to be minimised in all horizons, despite measured large amounts of adsorbed organic C in the mineral horizons and an assumed large concentration in the forest floor. Otherwise NuCM calculates a very high initial linear adsorption constant, which damps simulated dynamics in all horizons. This suggests the model structure should be changed to allow large initial amounts of adsorbed C and low adsorption rates.

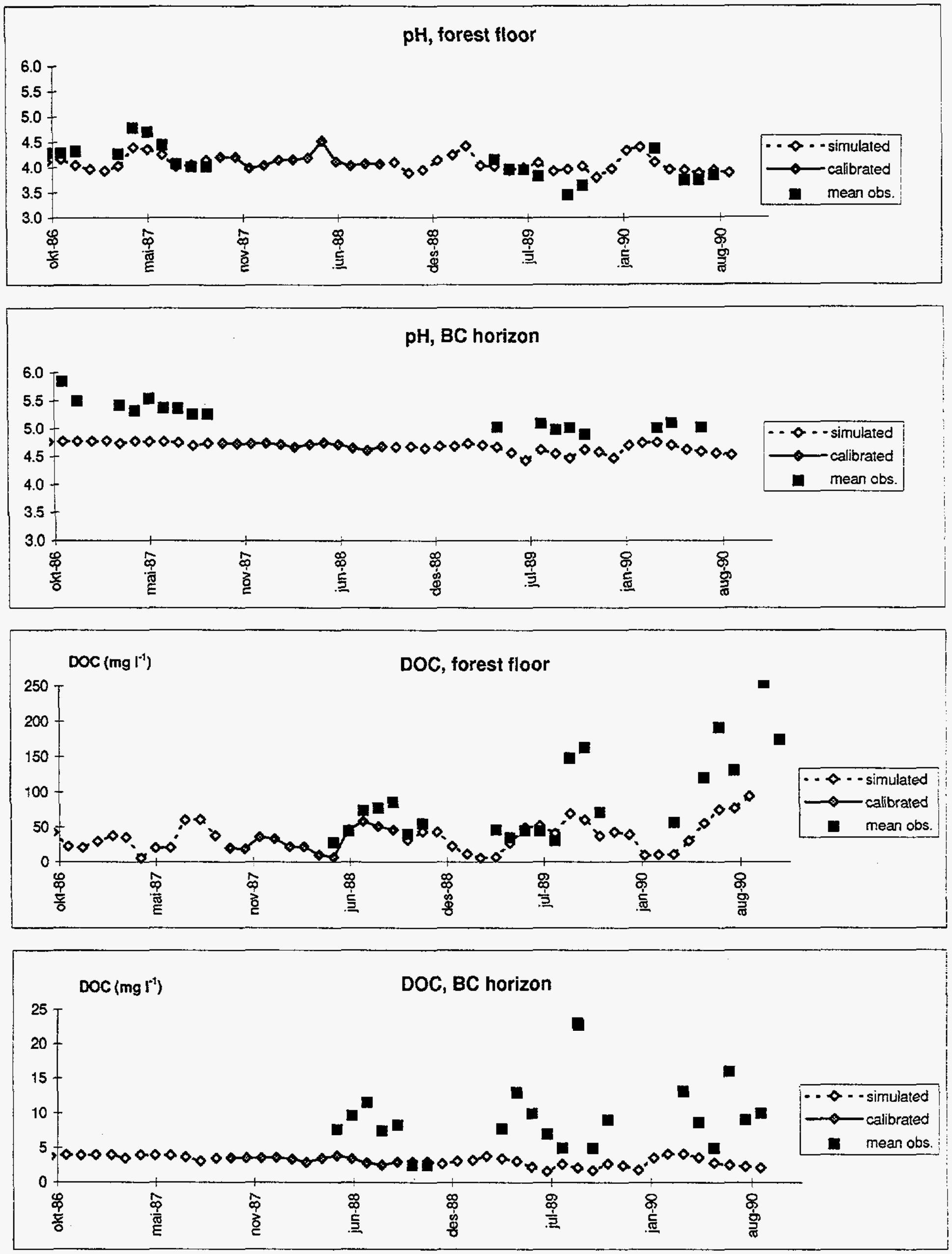

Fig. 7. Simulated, calibrated and observed pH and concentrations of DOC in soil solution for the period (Oct. 1986-Sept. 1990).

3.5 Comparison of NuCM and MAGIC

The MAGIC versus NuCM simulations of the three different scenarios (Table 5) show large differences in the resulting exchangeable Ca^{2+} and K^+ pools at the 40 % and 90 % reductions in S deposition with 25 % reduction in tree uptake. MAGIC simulates 0 $Ca^{2+}$and K^+ for both these scenarios, while NuCM simulates an increase in both these pools for each S reduction compared to the no change scenario (0/0sc.). In terms of critical loads, MAGIC predicts a load of < 0 $mmol_c$ $m^{-2}year^{-1}$ SO_4 for 2038 to ensure the soil profile does not get soil solution with a molar Ca/Al ratio < 1, assuming no reduction in tree uptake (Wright et al., 1991). NuCM on the other hand predicts a Ca/Al ratio of 1.2 and 0.9 in the O and Bs layer respectively in 2034 for the no change scenario (0/0sc.) In other words, with no reduction in uptake and a SO_4 load of 69 $mmol_c$ $m^{-2}year^{-1}$, NuCM predicts a Ca/Al ratio only just below 1.0 in the Bs layer in 2034.

Table 5. The pool of exchangeable base cations ($mmol_c$ m^{-2}) in the upper 60 cm soil as measured and simulated by MAGIC and NuCM for different scenarios . In addition the Ca/Al molar ratio is shown for 1986 and the no change scenario.

	1986 measu-red.	[a]2034 NuCM 0/0sc.	2038 MAGIC 40/25sc.	2034 NuCM 40/25sc.	2038 MAGIC 90/25sc.	2034 NuCM 90/25sc	2038 MAGIC 90/100sc.	2034 NuCM 90/90sc
Ca	976	506	0	589	0	679	957	823
Mg	236	161	96	185	123	209	343	250
K	452	266	0	284	0	315	633	498
Na	137	97	124	107	142	125	164	139
Ca/Al O	3.7	1.2						
Ca/Al Bs	1.3	0.9						

[a] 2034 = ending year of simulation; 0/0 = % reduction of total S deposition in 1988 / % reduction in tree uptake or growth relative to 1988; sc. denotes scenario i.e. NuCM is simulated for a scenario with 100 % of total S deposition in 1988 and 100 % of tree growth in 1988. The other scenarios can be interpreted likewise and are described in more detail in methods.

[b] all MAGIC results from Wright et al., (1991)

Clearly, much of the difference between MAGIC and NuCM is due to differences in assumptions with respect to important fluxes. The net uptake and weathering fluxes shown in Table 6 provide much of the explanation. The Ca^{2+} and K^+ uptake used in MAGIC are twice and thrice respectively the uptake fluxes in NuCM. If P30 uptake data (values in parentheses, Table 6) had been used in MAGIC, although the Ca^{2+} uptake flux becomes more similar to the uptake flux simulated by NuCM, the K^+ flux is still double that in NuCM. The litterfall flux of K^+ was overestimated by NuCM (Table 4) due to no description of translocation in NuCM, which means net uptake of K^+ is underestimated in NuCM (Table 6). The Ca^{2+} and Mg^{2+} weathering flux used in NuCM are 32 times and twice respectively the weathering calibrated by MAGIC. In NuCM, weathering was calibrated as well as possible towards the weathering fluxes based on a 6 year Nordmoen soil column experiment (Teveldal, 1993; Teveldal and Jørgensen, 1996). Weathering rates have been estimated by different methods at Nordmoen (Kvindesland and Røsberg, in review) and the estimates for Ca^{2+} and Mg^{2+} show the largest ranges. Although the real weathering rate for Ca^{2+} cannot yet be picked from this range, use of the column experiment weathering rate led to approximate balance in the Ca^{2+} cycle, while use of the Ca^{2+} weathering rate applied in MAGIC led to 0 exchangeable Ca^{2+} pool in the top 60 cm of soil within 17 years. Soil samples taken from an adjacent Norway spruce stand at Nordmoen in 1996 (i.e 10 years later) indicate exchangeable Ca^{2+} levels of 400 $mmol_c$ m^{-2} in the forest floor alone (De Wit, pers. comm.), which is larger but in the same range as that measured in the forest floor for P30 in 1986 (Kvindesland and Røsberg, in review). This suggests the Ca^{2+} weathering rate applied in MAGIC is too low. However one must conclude that neither MAGIC nor NuCM could produce good weathering estimates on their own and independent field estimates of weathering are vital for realistic future predictions. The MAGIC 90/100 sc. and the NuCM 90/90 sc. compare more favourably, as all exchangeable pools are simulated slightly larger with the 100% reduction in uptake compared to the 90 %.

The MAGIC model simplifies all involvement of the tree's nutrient cycling to a tree uptake versus time graph. NuCM uses a stipulated tree biomass versus time curve as input but the simulated growth may be over or under this line due to luxury uptake or deficiency. In all three scenarios, NuCM simulated growth slightly under the stipulated growth, due to N deficiency. A growth rate that is simulated due to feedback mechanisms between soil solution and the tree is more realistic than a simple uptake curve dictated at the start of the simulation. A fairer future comparison of MAGIC and NuCM should calibrate uptake and weathering fluxes similar in both models, so the effects of including nutrient cycling can become clearer in the results.

Table 6. Comparison of important fluxes ($mmol_c$ m^{-2} $year^{-1}$) simulated by MAGIC for 1988 and NuCM for the period, 1986-1990.

	deposition	net uptake[a, b]	weathering[c, d]
Ca MAGIC	11	30 (26)	1
Ca NuCM	8	19	32 (36)
Mg MAGIC	4	11 (12)	12
Mg NuCM	4	7	26 (18)
K MAGIC	4	27 (21)	4
K NuCM	3	9	6 (4)
Na MAGIC	22	0	12
Na NuCM	23	1	15 (7)

[a] In MAGIC net uptake is calculated as the requirement for wood for 40 year old trees in plot, P40 (Kvindesland and Røsberg, in review) (Johnson and Lindberg, 1992 appendix ns3, R1). In NuCM, uptake is calculated for 30 year old trees in plot, P30:

net uptake = total uptake - (exudation + litterfall) (fluxes from Table 4).

[b] The values in parentheses are the requirement for wood in plot P30 (Johnson and Lindberg, 1992 appendix ns3, R2).

[c] In MAGIC, weathering is calibrated to fit all other measured input data. In NuCM, weathering is calibrated based on column weathering estimates (Kvindesland, in review).

d The values in parentheses are the field weathering fluxes estimated from soil column experiments (Teveldal, 1993; Teveldal and Jørgensen, 1996).

4. Conclusions

The concentrations of Mg^{2+} and K^+, but not Ca^{2+} or H^+ were simulated successfully in throughfall on a monthly basis. There is no process description of the consumption of inorganic N in the canopy, which causes problems in simulating inorganic N concentrations in throughfall. Research is urgently needed to quantify the sinks of this consumed inorganic N and when and where it later returns to the nutrient cycle, so a process-oriented description can be added to NuCM.

In soil solution, the concentrations of SO_4^{2-} and Ala (total monomeric Al) were simulated successfully in all horizons, while pH and the concentrations of Ca^{2+}, Mg^{2+} and Na^+ were simulated reasonably in the O and E horizons. The concentration of NO_3^- in soil solution was simulated poorly in the dormant season due to probably overestimated concentrations of NO_3^- in throughfall and no description of immobilisation of N by microbes. In addition the nitrification description is too inflexible. Prolonged dry spells in the summer led to simulation of too low water fluxes by SOIL, which in turn caused too high concentrations for the anions, NO_3^- and Cl^- and hence the cations, Ca^{2+}, Mg^{2+} and Ala during these months.

The nutrient cycling of the cations, NH_4^+ and Mg^{2+} were simulated successfully. The nutrient cycling of Ca^{2+}, especially exudation, uptake and litterfall was simulated poorly due to no process description of the accumulation of Ca^{2+} in needles as they age. It is vital to describe at least simple accumulation and translocation in needles, where the net difference between element content in current and newly fallen needles is included, so both uptake and litterfall can be simulated better in NuCM.

Future predictions by NuCM were compared to the predictions by MAGIC under different scenarios (sc.). For the 40/25sc. (40 % reduction in S and 25 % reduction in uptake) MAGIC and NuCM predicted very different soil exchangeable pools of Ca^{2+} and K^+ in 2034. While NuCM predicted 590 and 280 $mol_c\ m^{-2}$ of Ca^{2+} and K^+ respectively, MAGIC predicted 0 in both pools. While MAGIC, with continued tree uptake, predicted a critical load of $< 0\ mmol_c m^{-2} year^{-1}$ SO_4 to ensure a molar Ca/Al ratio > 1 in the top 60 cm of soil, NuCM in the 0/0sc. (no change scenario) predicted a Ca/Al ratio of 1.2 and 0.9 in the O and Bs layer respectively in 2034. The reasons for the large difference in predictions lies mainly in simulation of the important fluxes, uptake and weathering. The K^+ net uptake in underestimated in NuCM because K^+ in litterfall is overestimated due to no description of translocation. While MAGIC simulates 1, NuCM simulates 32 for the Ca^{2+} weathering rate (in $mmol_c\ m^{-2}\ year^{-1}$). Although the real Ca^{2+} weathering rate is still unknown , the MAGIC rate appears far too low, so predictions for critical loads and exchangeable pools in 2034 are likely to lie nearer NuCM's predictions than MAGIC's predictions. To compare how the inclusion of nutrient cycling processes affects predictions, NuCM and MAGIC should be compared in the future with similar uptake and weathering fluxes.

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