Modelling of seasonal effects of soil processes on N leaching in northern latitudes*

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Abstract Concentrations of inorganic nitrogen (N) in non-polluted and undisturbed northern rivers are often lower during summer than during the dormant season. The great difference between summer and winter N concentrations probably reflects higher soil water N contents in the dormant season compared with the growing season, when inorganic N is usually retained effectively. Microbial activity in soil is observed even in sub-zero temperatures and it is generally assumed that in the northern latitudes some N mineralization occurs during winter. The dynamic, semi-distributed INCA (Integrated Nitrogen in Catchments) model was applied to the Simojoki river basin in the boreal zone in northern Finland. With this model process rates and loads of N can be simulated in different land use modes. The INCA model was not able to simulate the high inorganic N concentrations in the river water in winter unless N processes in sub-zero temperatures were included. The aim of this study was to compare the simulated N mineralization in two different land use modes: boreal forests on mineral soil and agricultural fields. Net N mineralization occurring during the season when soil is mainly frozen (November–April) accounted for 43% of the annual N mineralization. This work indicates the importance of over-winter N processes in northern areas, which should be taken into account when modelling nutrient leaching.

Keywords Catchment scale modelling; nitrogen leaching; nitrogen mineralization; northern river basin; seasonality

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Introduction

The transfer of nitrogen (N) through rivers to estuaries has greatly increased since human activities, primarily agriculture and combustion of fossil fuels, have significantly altered the global N cycle (Vitousek *et al.* 1997). In many terrestrial, freshwater and marine ecosystems N plays an important role in controlling species composition, diversity and functioning.

The annual hydrological pattern in the boreal zone in northern Finland is dominated by a snowmelt-induced spring flood in late April–May. Most nutrient leaching occurs during this high flow period. Typical hydrological features are long (5–7 months) winters with continuous snow cover and soil frost. Smaller flow peaks occur in autumn due to rainfall. When studying N leaching to watercourses, it is of great importance to estimate both hydrology and N processes correctly during these flow peaks.

Concentrations of inorganic N are often lower during summer than during the dormant season in non-polluted and undisturbed northern rivers (Arheimer *et al.* 1996; Williams *et al.* 1996, 2001; Kaste and Skjelkvåle 2002). Observed high N concentrations in late winter or early spring are often explained by release of N from the snow pack and/or from soil (Arheimer *et al.* 1996; Stottlemyer *et al.* 1997; Williams *et al.* 1996, 2001).

Nitrogen is considered to be the growth-limiting factor in most boreal forest ecosystems and natural ecosystems are characterized by low inputs and outputs of inorganic N. Leaching losses and gaseous losses are generally less than a few $kg\,N\,ha^{-1}\,a^{-1}$ (Gundersen and Bashkin 1994). According to DeLuca *et al.* (2002), N fixation potential in forests of northern Scandinavia and Finland varies between 1.5 and $2.0\,kg\,N\,ha^{-1}\,a^{-1}$. Mineralization of soil

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organic matter is still the most important source of N in boreal ecosystems, and typically annual net mineralization is one order of magnitude higher than inputs from the atmosphere (Stottlemyer and Toczydlowski 1999a, b).

Traditionally, N mineralization in soil is assumed to take place within a temperature range of 5–35°C, and it is generally assumed that the rate of transformation approximately doubles when the temperature increases by 10 degrees (Stanford *et al.* 1973). On the other hand, there is also evidence for microbial activity at lower temperatures, even at sub-zero temperatures. In a field incubation study carried out in a sub-arctic region in northern Sweden Schmidt *et al.* (1999) observed both mineralization and immobilization of N and P in soil during winter. Stottlemyer and Toczydlowski (1999b) found that 40% of net N mineralization (average $15\,\mathrm{kg\,N\,ha^{-1}\,a^{-1}}$) in boreal forest soil occurred during the winter. In laboratory incubations of arctic soils Elberling and Brandt (2003) reported microbial soil respiration to continue at temperatures down to 18°C. Winter soil respiration accounted for about 40% of the annual soil respiration.

Clein and Schimel (1995) observed both C and N mineralization at sub-zero temperatures (−2°C and −5°C) in tundra and taiga soils. Schimel *et al.* (2004) measured high net N mineralization rates during the fall and winter in tussock tundra soils with a heavy snow cover, in which soil temperature never decreased below −7°C, followed by immobilization in thaw. Under lighter snow cover, when the soil temperature decreased to −25°C soil N mineralization was limited. Kähkönen *et al.* (2001) observed organic matter degrading microbial activities to cease at −7.0°C in boreal forest soils. The actual soil temperature below the snow cover never decreased below −3°C at their study site in central Finland. Snow is an effective insulator and thus, in the case of snow-covered soil, the soil temperature is typically only a few degrees below zero although the air temperature may be lower.

If N accumulates in the soil under the snow cover, it would be available for plants in early spring. In boreal coniferous systems carbon uptake increases intensively in spring (Falge *et al.* 2002). In addition, N uptake during snowmelt has been demonstrated in both alpine and arctic ecosystems (Bilbrough *et al.* 2000).

The dynamic, semi-distributed nitrogen model INCA (Integrated Nitrogen in Catchments) was applied to the Simojoki river basin. The study area was located in the boreal vegetation zone in northern Finland. Calibration of the model to observed discharge, inorganic N concentration in river water and annual N balances is described in Rankinen *et al.* (2002, 2004c). The main aims of this study were to simulate seasonal N mineralization rates and to compare the simulations to observed values and rates found in literature references. Two representative land use classes were studied, i.e. forest on mineral soil and agricultural fields. Forest on mineral soil is a common land use mode in river basins, but agriculture is assumed to have a stronger influence on river water quality.

Material and methods

Site description

The river Simojoki discharges to the Bothnian Bay in the Baltic Sea. The river basin ($3160\,\mathrm{km^2}$) can be subdivided into nine sub-basins (Figure 1) (Ekholm 1993). Over the period 1961–1975, mean annual precipitation was 650–750 mm and mean annual runoff 350–450 mm. Mean daily flow at the outlet was $37.2\,\mathrm{m^3\,s^{-1}}$ during 1965–1990. There are about 170–180 winter days and the mean annual temperature is +0.5 to +1.5°C (Perkkiö *et al.* 1995). The duration of the snow cover is from the middle of November to early May and the depth of the snow cover is 40–80 cm. The River Simojoki freezes up at the end of October or the beginning of November and the ice cover usually breaks up in the middle of May. According to the Finnish Meteorological Institute the growing season started on average on

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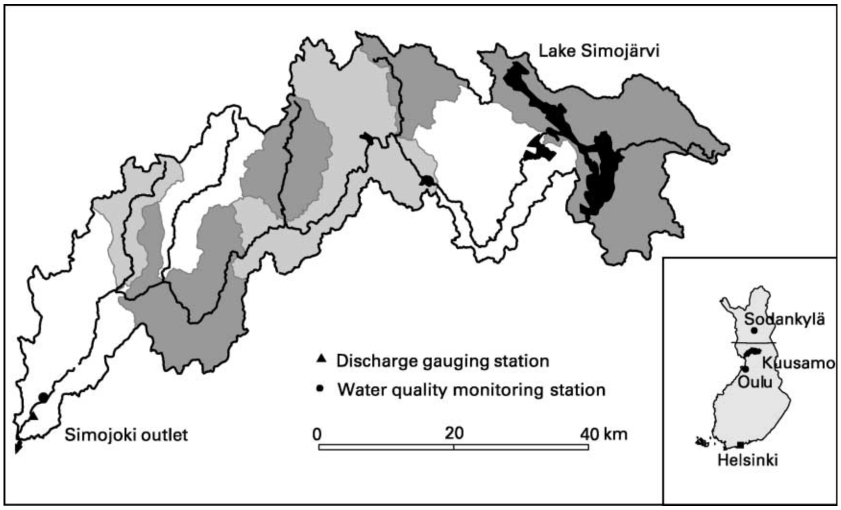

Figure 1 Location of the River Simojoki basin and the observation stations of the Finnish Meteorological Institute in northern Finland

10 May in the period 1961–1990. The length of the growing season was 140 days on average.

The River Simojoki is a salmon river without any major point pollution sources and the dominant human impacts are forestry, agriculture and atmospheric deposition. It is located partly in middle boreal and partly in northern boreal vegetation zone. Peatlands and peatland forests are common in the region and an average of 0.5% of the total catchment area is felled annually. In 1995 peat mining areas covered 0.4% of the catchment area. Urban areas cover only 0.06% and agricultural fields 2.7% of the catchment area (Perkkiö *et al.* 1995). Grass cultivation for animal husbandry is the most common form of agricultural production (Rankinen *et al.* 2004b).

Atmospheric N deposition in the Simojoki river basin is $2.1–2.3\,\mathrm{kg\,N\,ha^{-1}\,a^{-1}}$. According to previous modelling studies (Rankinen *et al.* 2004c) the main N input in undisturbed forested areas is from N mineralization. Simulated N fixation in forested areas is low ($\leq 1\,\mathrm{kg\,N\,ha^{-1}}$). Leaching losses are also low ($\leq 1\,\mathrm{kg\,N\,ha^{-1}\,a^{-1}}$) and denitrification is assumed to occur mainly in forest cut areas on organic soil. Agricultural areas are assumed to be fertilized about $160\,\mathrm{kg\,N\,ha^{-1}}$ (Rankinen *et al.* 2004b). Simulated annual leaching from agricultural areas varies between 10 and $15\,\mathrm{kg\,N\,ha^{-1}\,a^{-1}}$.

The INCA model

On the basis of earlier work by Whitehead *et al.* (1998), a new version of the dynamic, process-based and semi-distributed INCA model was developed and described in detail by Wade *et al.* (2002). In this work the model version in which soil temperature is calculated based on simplification of partial differential equations for combined water and heat flow is used (Rankinen *et al.* 2004a).

The INCA model integrates hydrology, catchment and river N processes and simulates daily NO_3-N and NH_4-N concentrations as time series at key sites, as profiles down the river system or as statistical distributions. The term "semi-distributed" is used, as it is not intended to model catchment land surface in a detailed manner but to use land-use class in a sub-basin as a basic modelling unit. River, soil water and groundwater NO_3-N and NH_4-N concentrations and fluxes are produced as daily time series.

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Three components are included: the hydrological model, the catchment N process model and the river N process model. Sources of N include atmospheric deposition, the terrestrial environment and direct discharges.

Hydrological processes in soil are simulated in the hydrological sub-model. Hydrology within a catchment is modelled using a simple two-box approach, with reservoirs of water in the reactive soil zone and in the deeper groundwater zone. Nitrogen can enter the river system either by lateral flow through the soil surface layers or by vertical movement and transport through the groundwater zone. The land use hydrological model is a daily mass balance model with which daily water flows are computed for soil, groundwater and leaching to the river for up to six land use classes. The river flow model is based on mass balance of flow and uses a multi-reach description of the river system. Within each reach, flow variation is determined by a non-linear reservoir model. In the river the key processes are denitrification and nitrification.

The mass balance equations for both NO_3-N and NH_4-N in the soil and groundwater zones are solved simultaneously with the flow equations. The key N processes that are solved in the soil water zone are nitrification, denitrification, mineralization, immobilization, N fixation and plant uptake of inorganic N in six land use classes. In the groundwater zone it is assumed that no biochemical reactions occur.

Mineralization of N in soil is assumed to occur from an unlimited organic N pool at a constant rate:

$$\frac{\mathrm{d}x}{\mathrm{d}t} = C_{\min} f(SMD) f(T) 100 \tag{1}$$

where $C_{\min}$ is the mineralization rate ($\mathrm{kg\,N\,ha^{-1}\,d^{-1}}$) and x is the ammonium load ($\mathrm{kg\,km^{-2}}$). The factor of 100 converts from hectares to km^2.

Rate coefficients of N processes are temperature- and moisture-dependent. The moisture response is calculated as

$$f(SMD) = \frac{SMD_{\max} - SMD}{SMD_{\max}} \tag{2}$$

where SMD is the daily soil moisture deficit (mm). Temperature response (Bunnel *et al.* 1977) for all temperatures is calculated as

$$f(T) = t_{Q10}^{(T - t_{Q10bas})/10} \tag{3}$$

where t_{Q10} $(-)$ and t_{Q10bas} (°C) are parameters and T (°C) is soil temperature. The parameter t_{Q10} is the factor change in rate with a 10 degree change in temperature and the parameter t_{Q10bas} is the base temperature for N processes at which the response is 1 (Figure 2). The reference value for t_{Q10} is 2 and for t_{Q10bas} is 20 according to Jansson and Karlberg (2001).

Input data and model calibration

As input data the INCA model needs daily data of Hydrologically Effective Rainfall (HER), Air Temperature (T), Soil Moisture Deficit (SMD) and Precipitation (P). The model also needs land use information in each sub-catchment as well as information concerning point sources and atmospheric N deposition.

Hydrological input data were taken from the output of the WSFS (Watershed Simulating and Forecasting System) (Vehviläinen 1994). The basic component of a watershed model is a conceptual hydrological model which simulates runoff using precipitation, potential evaporation and temperature as inputs. The principles of the WSFS are based on the HBV model (Bergström 1976). The system is commonly used for flood forecasts, water resources management and supervision purposes.

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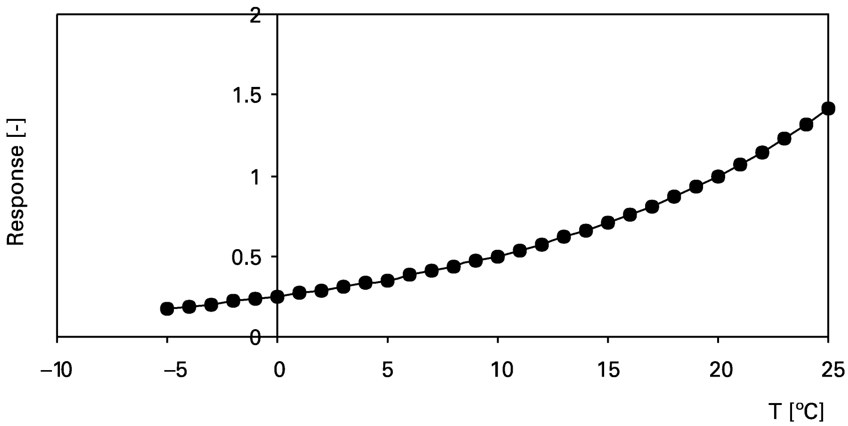

Figure 2 Temperature response for N mineralization ($t_{Q10} = 2$ and $t_{Q10bas} = 20$)

Land use classes were derived from the satellite image-based land use and forest classification of Finland (Vuorela 1997). In this application the six land use classes were: forest on mineral soil, cut forest on mineral soil, forest on organic soil, cut forest on organic soil, agricultural land and open surface water. Forest on mineral soil covers 35%, cut forest on mineral soil 4%, forest on organic soil 52%, cut forest on organic soil 1%, agriculture 2% and open surface water 6% of the whole river basin area.

Inorganic N deposition was calculated using the regional nitrogen transport and deposition model DAIQUIRI (Syri *et al.* 1998; Kangas and Syri 2002). Information of effluents from point sources was available at the databases of the Finnish Environment Institute.

Time series of discharge and inorganic N concentrations are needed along the river for comparison between the simulated and measured values. The River Simojoki has two discharge gauging stations; one is located at the river outlet, while the other is at the Hosionkoski rapids (Hyvärinen 1996). Measured inorganic N concentrations at five observation stations along the river were used in calibration. Water sampling frequency at the outlet of the Simojoki river was 12–17 samples per year during the study period.

In this study net mineralization as the difference between gross mineralization and microbial immobilization was calibrated. The parameter values after calibration are presented in Table 1. Mineralization was assumed to cease at −5°C. Otherwise the calibration was similar to that described in Rankinen *et al.* (2004c).

Results and discussion

Inorganic N concentrations in river water

Both observed and simulated discharges and inorganic nitrogen concentrations (calibration years 1994–1996) are presented in Figure 3. Simulated inorganic N concentrations peak in

Table 1 Parameter values for N mineralization

Process	Land use class	Value	Unit
C_{min}	Forest	0.35	kg ha^{-1} d^{-1}
	Arable	1.3	kg ha^{-1} d^{-1}
t_{Q10}	Forest	2	–
	Arable	2	–
t_{Q10bas}	Forest	20	°C
	Arable	22	°C

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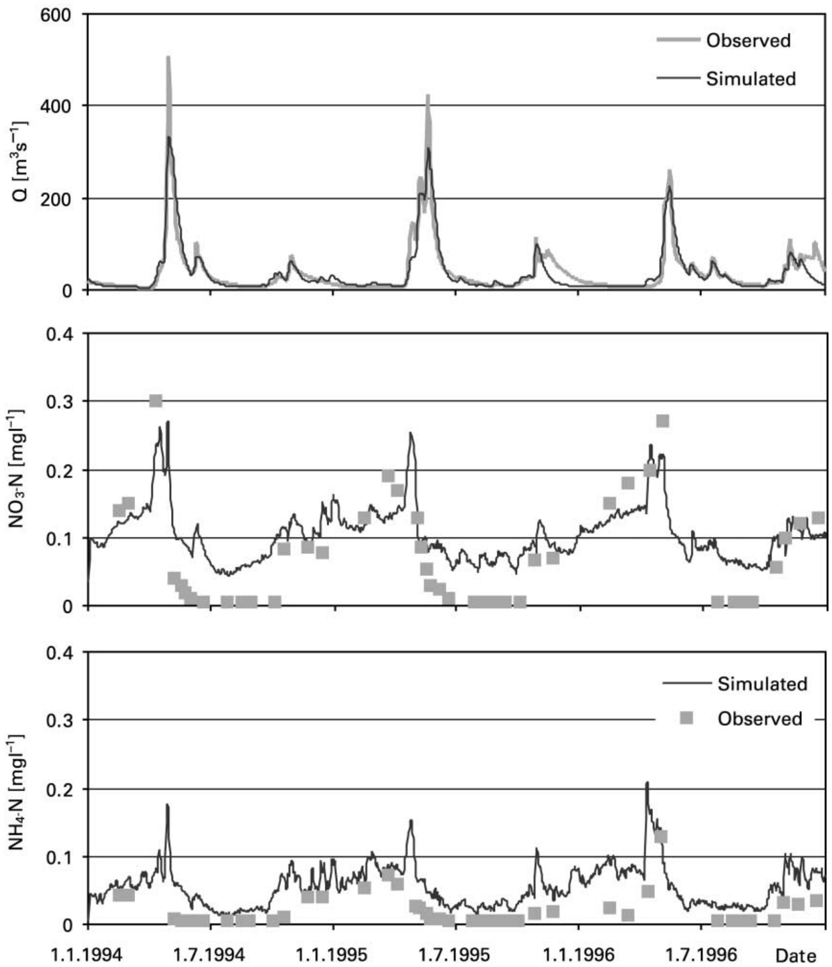

Figure 3 Simulated and measured discharge and inorganic N concentrations in the River Simojoki outlet

late winter/early spring just before snow melt. In long time series (30 years) of inorganic N concentrations in the River Simojoki both the median concentration of NO_3-N (220 $\mu g\,l^{-1}$) and the inter-annual variation were highest in April (Rankinen *et al.* 2002). NO_3-N concentration decreased during the growing season to an almost negligible level. The seasonal pattern of NH_4-N concentrations was similar to that of NO_3-N, except that the concentration levels were lower. A similar pattern of inorganic N concentrations during the growing season compared with the dormant season was also observed by Arheimer *et al.* (1996), Williams *et al.* (1996, 2001) and Kaste and Skjelkvåle (2002). Observed high N concentrations in late winter or early spring were often explained by the release of N from the snow pack and/or soil (Arheimer *et al.* 1996; Stottlemyer *et al.* 1997; Williams *et al.* 1996, 2001).

In this work atmospheric deposition was assumed to accumulate in the snow pack during winter and to flush to soil water when the snow started to melt. Accumulated deposition during winter is on average 0.53 $kg\,N\,ha^{-1}$, which is not sufficient to change river water inorganic N concentrations significantly. Stottlemyer *et al.* (1997) also failed to find any significant correlation between snow pack ion loss and soil water chemistry. They concluded that soil processes such as over-winter nitrification and mineralization, ion exchange and biological uptake are probably major factors modifying melt water chemistry.

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Stottlemyer and Toczydlowski (1999b) found that, among boreal species types, net N mineralization rates were positively correlated with mean streamwater NO_3^- concentrations. In addition, Williams *et al.* (1996) concluded that mineralization under seasonal snow, rather than snowmelt release of NO_3^-, may control NO_3^- concentrations in surface waters of high-elevation catchments.

The INCA model was not able to simulate the high inorganic N concentrations in the river water until N processes in sub-zero temperatures down to -5°C were included in the model. This limiting temperature value coincides with the temperatures at which microbial activity was found in similar vegetation and soil types (Kähkönen *et al.* 2001; Schmidt *et al.* 1999). Temperature response of soil N mineralization for all temperatures was calculated by using the same temperature coefficients. In laboratory incubation studies of CO_2-production, temperature coefficients (Q_{10} values, i.e. the factor change in rate following a 10-degree change in temperature in exponential relationships) are found to increase abruptly with freezing of soil (Mikan *et al.* 2002; Elberling and Brandt 2003). Respiration coefficients may be considered as conservative estimates of temperature effects on N mineralization (Rustad *et al.* 2001). However, Elberling and Brandt (2003) found that the temperature-dependent soil respiration for a field can be rather well described by a single temperature coefficient value, when soil temperature is above approximately -9°C and other influential factors like soil moisture are taken into account.

Annual and seasonal mineralization rates

Simulated annual net N mineralization, 35 kg ha^{-1} in forest on mineral soil, is one order of magnitude higher than atmospheric inorganic N deposition. Persson and Wirén (1995) measured 35–58 kg ha^{-1} annual net N mineralization in acid forest soils in Southern Sweden in low leaching sites, which could represent northern conditions. Stottlemyer and Toczydlowski (1999b) measured annual net N mineralization to be 15 kg ha^{-1} in a study site located in a transitional zone between boreal forest and northern deciduous forest. In Finland the measured above-ground vegetation requirement of N in boreal forests was 28–51 kg ha^{-1} a^{-1} on mineral soil (Mälkönen 1974) and 26–42 kg ha^{-1} a^{-1} on organic soil (Finér 1989).

Simulated annual net N mineralization is 109 kg ha^{-1} in agricultural fields. In Sweden Paustian *et al.* (1990) calculated annual N budgets from measurements for barley and grass ley. Annual net N mineralization was estimated to be 80–90 kg ha^{-1} in barley fields and 210 kg ha^{-1} in grassland. Higher N mineralization rate corresponded with higher carbon turnover and microbial activity in grass ley.

Seasonal net N mineralization in the land use class of forest on mineral soil is presented in Figure 4. Monthly rates are highest in early summer when the soil is warm but soil moisture does not limit process rates. Monthly rates decline in summer, but increase again in autumn. Mineralization rates are stable in winter, probably reflecting stable soil temperatures under the snow pack. The lowest mean monthly N mineralization in February is clearly higher than the atmospheric deposition accumulating in the snow pack during the winter.

The year is divided into the dormant season when soil is assumed to be mainly frozen (November–April) and the growing season when soil is assumed to be unfrozen (May–October). This is a simplification of the actual growing seasons, which may vary considerably temporally from year to year and spatially within a large river basin. The lengths of the growing seasons at three nearby observation stations of the Finnish Meteorological Institute (Figure 1) are presented in Table 2. Net N mineralization occurring during the dormant season accounts for 43% of the annual N mineralization. These percentages are consistent with the findings of Stottlemyer and Toczydlowski (1999b) that net N mineralization peaked in early summer and 40% of the annual N mineralization occurred in winter.

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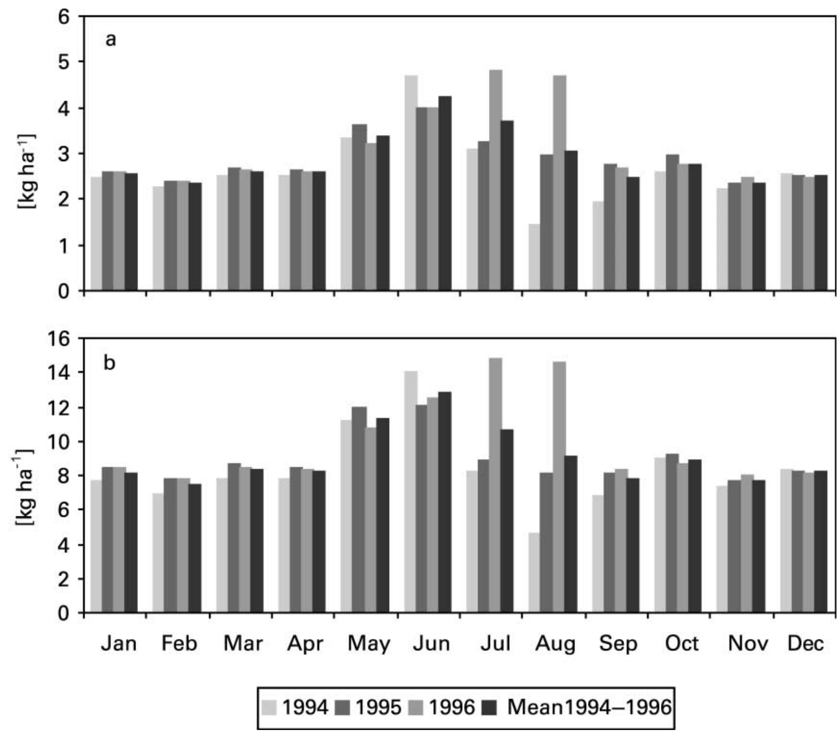

Figure 4 Monthly mineralization rates in the land use classes (a) forest on mineral soil and (b) agricultural fields

Seasonal net mineralization in agricultural fields follows the pattern of that in forests (Figure 4), but the rates are higher. Simulated net N mineralization during the cropping season is 60 kg ha^{-1} and during the dormant season 48 kg ha^{-1}. The simulated daily mean net N mineralization is 0.33 kg ha^{-1} d^{-1} during the growing season. Lindén *et al.* (1992a) found that on average net N mineralization in cropped (barley), unfertilized soil from early spring to yellow ripeness was 41 kg ha^{-1}, and that the daily mean was 0.4 kg ha^{-1} d^{-1}. In Finnish barley fields Sippola and Yläranta (1985) measured inorganic N contents of 15–78 kg ha^{-1} in spring before the start of the growing season. The average mineralization rate from harvesting to early spring in barley field was 11 kg ha^{-1} with a range of 0–30 kg ha^{-1}.

In addition to temperature and moisture there are several other factors which influence net N mineralization including soil type, vegetation type and species richness among a certain vegetation type, fertilization, and C and N dynamics (Lindén *et al.* 1992b; Stottlemyer *et al.* 1995; Stottlemyer and Toczydlowski 1999b; Schimel *et al.* 2004). Gross mineralization and

Table 2 Growing periods at the observation stations of the Finnish Meteorological Institute

Station	Coordinates	Year	Start	End
Oulu	64°56, 25°22	1994	25.4	11.10
		1995	18.5	18.10
		1996	22.5	13.10
Kuusamo	65°59, 29°13	1994	6.5	24.9
		1995	21.5	27.9
		1996	1.6	25.9
Sodankylä	67°22, 26°37	1994	30.5	25.9
		1995	24.5	13.9
		1996	31.5	25.9

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immobilization of N have partly different regulating factors, so that gross and net mineralization rates are not necessarily correlated (Stottlemyer and Toczydlowski 1999a; Schimel *et al.* 2004). In this work net mineralization rate, which represents the difference between immobilization and gross mineralization, was sufficient to calibrate the model. However, measured mineralization rates would be needed to completely calibrate the simulated mineralization process.

Conclusions

The INCA model version in which N processes in sub-zero temperatures were included was able to simulate the high inorganic N concentrations in river water in winter. Net N mineralization rate proved to be sufficient to calibrate the N leaching model, as it represents the difference between immobilization and gross mineralization. The same temperature coefficients for all temperatures were used in calibration. Simulated annual and seasonal net N mineralization appeared to be at the correct level in the two different land use modes studied. However, measured mineralization rates would be needed to completely calibrate the simulated mineralization process. Despite all the simplifications made in this work, the importance of over-winter N processes in low temperatures in soil was clearly indicated. These processes should be taken into account in modelling studies of nutrient leaching in northern areas.

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