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ANALYSIS OF INORGANIC NITROGEN LEACHING IN A BOREAL RIVER BASIN IN NORTHERN FINLAND

Katri Rankinen

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<p>In this study the dynamic, semi-distributed INCA-N model was applied to the boreal Simojoki river basin in northern Finland to outline inorganic nitrogen (N) leaching patterns and N processes in catchment scale. Special emphasis was paid to the quality assurance of the modelling work. The dominant human impacts in the area are forestry, agriculture, scattered settlement and atmospheric deposition. In order to assess the effectiveness of current environmental policies and to implement river basin management plans, it is essential to know the relative significance of the different sources of pollution. INCA-N explained main features of the hydrological pattern and seasonality of inorganic N concentrations in river water when N processes in soil in sub-zero temperatures were included. Over-winter N mineralization processes in soil accounted for 38% of annual N mineralization. The lowest concentrations during the growing season were not reproduced, which indicates that there are some retention processes missing from the model. As summer is typically a low flow period the simulation results are reliable as long as the interpretation is based on daily or annual loads. Loading from the river basin was mostly dependent on annual hydrology and it was concentrated to peaks during the snow melting period. In the upper parts of the river inorganic N load originated mainly from commercial forests. At the outlet of the river anthropogenic sources accounted for more than half of the overall inorganic N load, with agriculture, forestry and scattered settlements making almost equal contributions. Expected changes in atmospheric N deposition would not have any significant effect but the combination of water protection measures both in agricultural and forestry areas and in scattered settlement areas would decrease inorganic N load by 18% of the total load. The expected increase in forest felling of 20% would not have an influence, but increase in agricultural land due to the EU's Common Agricultural Policy might lead to increased N load to the sea.</p>			
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<p>Tässä työssä hahmoteltiin alueellisesti osittain hajautetun INCA-N mallin avulla epäorgaanisen typen huuhtoutumiseen liittyviä prosesseja metsäisellä Simojoen vesistöalueella Pohjois-Suomessa. Erityistä huomiota kiinnitettiin mallinnuksen laatutyöhön. Alueella tärkeimmät ihmistoiminnan vaikutukset ovat peräisin metsätaloudesta, maataloudesta, haja-asutuksesta ja ilmasta tulevasta laskeumasta. Jotta voitaisiin arvioida vallitsevan ympäristöpolitiikan tehokkuutta ja toteuttaa vesienhoitoalueen hoitosuunnitelmia, on eri lähteiden osuus kokonaiskuormituksesta tiedettävä. INCA-N kykeni selittämään kohdealueen hydrologian lisäksi myös jokiveden epäorgaanisen typen konsentraatioiden vuodenaikaisvaihtelun, kun maassa tapahtuvien typen prosessien laskeminen nollan alapuolisissa lämpötiloissa sisällytettiin malliin. Talven aikana mineralisoituneen epäorgaanisen typen osuus oli 38% vuotuisesta mineralisaatiosta. Malli ei kyennyt toistamaan kasvukaudella havaittuja alhaisimpia konsentraatioita, mistä voi päätellä, että mallista puuttuu jokin pidättymisprosessi. Koska kesällä virtaama on pieni, simulointitulokset ovat kuitenkin luotettavia niin kauan kuin niiden tulkinta perustuu vuosikuormiin. Hydrologia vaikutti eniten vuosikuorman suuruuteen ja kuormitushuippu keskittyi kevättulvaan lumien sulaessa. Joen yläjuoksulla epäorgaaninen tyyppi oli peräisin lähinnä talousmetsistä, mutta joen alajuoksulla puolet kokonaiskuormituksesta oli peräisin ihmisten toiminnasta. Maatalouden, metsätalouden ja haja-asutuksen osuudet olivat lähes yhtä suuret. Odotettavissa olevat muutokset ilmasta tulevassa typpilaskelmassa eivät riitä muuttamaan mereen menevää typpikuormaa, mutta yhdistelmä vesien-suojelutoimenpiteitä maa- ja metsätaloudessa sekä haja-asutusalueilla voi vähentää epäorgaanisen typen kuormitusta 18%. Metsänhakkuiden oletettava lisäys 20%:lla ei lisää kuormitusta, mutta EU:n maatalouspolitiikka näyttäisi johtavan maatalousmaan pinta-alan lisääntymiseen ja siten myös kasvavaan epäorgaanisen typen kuormitukseen alueella.</p>		
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LIST OF APPENDED PAPERS

This thesis is based on the following papers:

I Rankinen K., A. Lepistö and K. Granlund 2002. Hydrological application of the INCA (Integrated Nitrogen in CAtchments) model with varying spatial resolution and nitrogen dynamics in a northern river basin. *Hydrology and Earth System Sciences* **6**(3): 339-350.

II Rankinen K., A. Lepistö and K. Granlund 2004. Integrated nitrogen and flow modelling (INCA) in a boreal river basin dominated by forestry: scenarios of environmental change. *Water, Air and Soil Pollution: Focus* **4**: 161-174.

III Rankinen K., T. Karvonen and D. Butterfield 2004. Development of a simple model for predicting soil temperature in snow covered and seasonally frozen soil. *Hydrology and Earth System Sciences* **8**: 706-716.

IV Rankinen K., K. Granlund and I. Bärlund 2004. Modelling of seasonal effects of soil processes on N leaching in northern latitudes. *Nordic Hydrology* **35** (4-5): 347-357.

V Rankinen, K., H. Lehtonen, K. Granlund, and I. Bärlund 2004. Assessing the Effects of Agricultural Change on Nitrogen Fluxes Using the Integrated Nitrogen CAtchment (INCA) Model. Complexity and Integrated Resources Management, Transactions of the 2nd Biennial Meeting of the International Environmental Modelling and Software Society, Manno, Switzerland, iEMSS, 2004.

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VII Rankinen, K., T. Karvonen and D. Butterfield. Application of the GLUE methodology in estimating the parameters of the INCA-N model. *the Science of the Total Environment*, accepted.

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Contribution of the author to Papers from I to VII is as follows:

I: Ms. Katri Rankinen was responsible for the model application in paper I. She parameterised the hydrological submodel for forested areas, and for agricultural areas together with Ms. Kirsti Granlund. Dr. Ahti Lepistö conducted analysis of N fluxes and dynamics in the river water. All authors took part in interpretation of results and writing.

II: Ms. Katri Rankinen had the major responsibility for the model application in paper II. Nitrogen processes for forested areas were parameterised together with Dr. Ahti Lepistö and for agricultural areas together with Ms. Kirsti Granlund. All the authors participated in planning, interpretation of results and writing of the paper.

III: Ms. Katri Rankinen initiated the paper. The physical soil temperature function was developed by Prof. Tuomo Karvonen and Ms. Katri Rankinen and they also wrote the paper. Mr. Daniel Butterfield commented the manuscript and coded the soil temperature function into the INCA-N model.

IV: Ms. Katri Rankinen had the major responsibility for the model application but N processes for agricultural areas were parameterised together with Ms. Kirsti Granlund. All the authors participated in analyzing the results. Ms. Katri Rankinen wrote the manuscript and Ms. Kirsti Granlund and Dr. Ilona Bärlund commented on it.

V: Dr. Ilona Bärlund initiated the paper and Ms. Katri Rankinen and Dr. Heikki Lehtonen planned how to combine models. Ms. Katri Rankinen had the major responsibility for INCA-N application and Dr. Heikki Lehtonen DREMFIA application. Ms. Katri Rankinen and Dr. Heikki Lehtonen wrote the paper and Dr. Ilona Bärlund and Ms. Kirsti Granlund commented on it.

VI: Ms. Katri Rankinen initiated the paper and Mr. Kaarle Kenttämies participated in updating the parameterisation of forestry areas and provided data for that. The scenarios were planned together with Ms. Suvi Nenonen. Ms. Katri Rankinen did INCA-N simulations. Dr. Heikki Lehtonen performed DREMFIA simulations and wrote on the changes in agricultural land use in Simojoki area. Ms. Katri Rankinen had the main responsibility of writing the manuscript and others commented on it.

VII: Prof. Tuomo Karvonen initiated the paper and coded the GLUE-programme. Ms. Katri Rankinen did calculations and had the main responsibility of writing the manuscript. Mr. Daniel Butterfield provided command-line version of INCA-N and commented on the manuscript.

*Voipi se tietenki niinki olla, minen kyllä usko.
(I know what you say is true, but I don't believe it)
– Pohjoisen mies mallinnuksesta –*

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Helsingissä 6.2.2006

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1 INTRODUCTION

1.1 Background

The monitoring of nutrient emissions from diffuse sources is less accurate than that from point sources, the emissions of which can be monitored at the outlet of a factory, sewage treatment plant or other single point. Diffuse pollution, or non-point pollution, can be defined as input of a substance emitted from moving sources, from sources extending over a wide area or from multiple sources such as agriculture, scattered dwellings and natural land. Year-to-year variation in non-point loading is large as the loading is highly dependent on local hydrology and meteorology.

Nutrient flux from catchments can be calculated on the basis of observed river discharge and sampling of water quality determinants (Rekolainen et al. 1995). When estimating emissions from non-point sources based on river monitoring, retention and other generally poorly known losses in the river system should be included (Behrendt and Bachor 1998, Hetling et al. 1999). The total river transport is also influenced by different pathways of nutrient input (Behrendt and Bachor 1998).

In order to assess the effectiveness of current environmental policies and to implement river basin management plans it is essential to know the relative significance of the different sources of pollution. The estimation of the contributions of different sources to overall pollution is known as source apportionment. Different quantification methods vary from complex process-based models to semi-empirical methods. In source oriented approach the emphasis is on nutrient flux entering the river system. Contribution of different diffuse sources are estimated for example by using export coefficients for different land use classes and sources. In load oriented approach the load monitored at the outlet of the river is separated into categories according to the source.

Mathematical modelling of nutrient processes, sources and sinks may help to achieve a better understanding of where water protection actions should be taken. Mathematical models in catchment scale have been used to estimate the origin and/or timing of nutrient loading by e.g. Arheimer and Brandt (2000), Arheimer and Lidén (2000), Krysanova and Becker (2000), Lepistö et al. (2001), Jarvie et al. (2002) and Grizzetti et al. (2005).

1.2 Mathematical modelling of nutrient leaching in catchment scale

Hydrological and nutrient leaching models vary in their methods for handling spatial and temporal scales. In lumped conceptual models the catchment is considered as a whole and described as a simplified physical system. Distributed process-based models represent the most complex structure, in which the catchment is described using a grid structure and the fluxes of water and nutrients are described by partial differential equations. A compromise between conceptual and fully distributed models are the so-called semi-distributed models, which divide the catchments into smaller units based on e.g. land use (Krysanova et al. 1998, Whitehead et al. 1998, Karvonen et al. 1999, Kokkonen et al. 2001, Wade et al. 2002, Krysanova et al. 2005). Spatially distributed models have their advantages in assessment where either detailed outputs or information about processes inside the catchment are needed,

although the simpler models may provide an overall picture of catchment behaviour with less input information (Refsgaard and Knudsen 1996, Krysanova et al. 1999, Lidén et al. 1999, Vachaud and Chen 2002, Alexander et al. 2002).

Improved computational resources have made it possible to develop and apply more and more complex distributed models. The type and spatial resolution of a catchment scale model define the input data needed to calibrate it. A model with a simple structure often does not make the best use of the available data. Conversely, the model structure and parameters cannot be identified accurately if there are too many model parameters but not enough data to test the model performance properly. This problem is known as over-parameterisation of a model (Refsgaard 1997, Beven 2001, Blöschl and Grayson 2002). To avoid over-parameterisation, Refsgaard (1997) proposed limiting the number of parameters subject to adjustments during calibration to as small a set as possible, and that the parameter values should as far as possible be assessed from available field data.

Hydrological systems are typically heterogeneous and hydrological processes are non-uniform and non-linear in time and space. The equations established for small-scale homogeneous systems are not necessarily valid in grid-scale heterogeneous systems. There is no general method to derive parameters from point measurements which are often very variable, and in practice it is impossible to measure all the parameters required for each grid square (Beven 1989, Beven 2001, Blöschl and Grayson 2002). Often different sets of parameters may give equally acceptable results, a problem which Beven (2001) called equifinality. Problems caused by heterogeneity and non-linearity in parameterising a hydrological model have been demonstrated by e.g. Durand et al. (2002).

Calibration and testing of distributed hydrological models should be based not only on runoff data but also on observed spatial patterns of catchment dynamics (Refsgaard 1997, Blöschl and Grayson 2002, Hattermann et al. 2005). If spatially distributed predictions are required, multi-site calibration and validation is needed. Multivariable checks are required if predictions of the individual sub-systems within the catchment are needed (Refsgaard 1997). Seibert and McDonnell (2002) presented a method to include 'soft data,' i.e. the qualitative knowledge of an experimentalist, into the calibration process of a model. Soft data has a high degree of uncertainty itself, a high spatial or temporal variation or it may also include some expert knowledge. They concluded that a hydrological model gave lower runoff efficiency value but more real description of catchment behaviour when it was calibrated against runoff and groundwater levels supplemented with 'soft data'.

Klemeš (1986) emphasized that non-stationarities create special requirements for model validation and suggested the hierarchical scheme of validation tests. The basic idea in this method was to test model behaviour through the required transition regime, for example land use change. In practice this method is difficult to use when applying a model to evaluate future changes, and he suggested to use data from representative catchments which have already undergone similar non-stationarities. When using the calibrated model to simulate the effects of climate change, the model should be validated using historical data of wet and dry periods.

According to Refsgaard and Henriksen (2004), not only validation tests against independent data but also uncertainty assessments especially in model structure and parameter values should be included in a modelling study. Different sensitivity (Hamby 1994) and uncertainty analyses (e.g. Thiemann et al. 2001, Vrugt et al. 2005)

are primarily concerned with the question of how model outputs are affected by the variability of the model parameters and input values and give useful information when these components are not completely known. The Generalised Likelihood Uncertainty Estimation (GLUE) (Beven and Binley 1992) approach defines the performance of possible parameter sets in terms of likelihood measures. This method has been applied to show that several sets of parameters can perform equally well in hydrological models (Durand et al. 2002, Pappenberger et al. 2005).

1.3 Seasonality of N dynamics and loading

Typical hydrological features in the boreal zone in northern Finland are long (5-7 months) winters with continuous snow cover and soil frost. The annual hydrological pattern is dominated by a snowmelt-induced spring flood in late April-May, and most nutrient leaching occurs during this high flow period. Smaller flow peaks occur in autumn due to rainfall. When studying nitrogen (N) leaching to watercourses, it is of great importance to estimate correctly both hydrology and N processes during these flow peaks. Particularly, the early phase of the spring flood is a critical period due to flushing of inorganic N from catchment soils and from melting snow, followed by dilution processes (Arheimer et al. 1996). 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, Williams et al. 2001, Kaste and Skjelkvåle 2002).

The limiting nutrient for boreal forest growth is generally nitrogen, 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, Piirainen et al. 1998). The N cycle is dominated by reactions involving biological material (O'Neill 1993), for example the natural fixation of N from atmosphere is carried out by free-living or symbiotic micro-organisms. According to DeLuca et al. (2002), microbiological N fixation potential in forests of northern Scandinavia and Finland varies between 1.5 and 2.0 kg N ha⁻¹ a⁻¹, mainly due to symbiosis between a cyanobacterium and feather moss. Atmospheric N deposition is in northern Finland less than 4 kg N ha⁻¹ a⁻¹ (Vuorenmaa 2004). Mineralization of soil organic matter by micro-organisms is still the most important source of inorganic N in boreal ecosystems. Typically annual net N mineralization (gross N mineralization minus immobilisation back to organic matter) is one order of magnitude higher than inputs from the atmosphere in northern areas (Stottlemeyer and Toczydlowski 1999a, Stottlemeyer and Toczydlowski 1999b).

Stottlemeyer et al. (1997) found no significant correlation between snow pack ion loss and soil water chemistry in field studies. 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. Stottlemeyer and Toczydlowski (1999b) found that among boreal species types, net N mineralization rates were positively correlated with mean streamwater NO₃⁻ concentrations. In addition, Williams et al. (1996) concluded that mineralization under seasonal snow, rather than snowmelt release of NO₃⁻, may control NO₃⁻ concentrations in surface waters of high-altitude catchments. Accumulated N in the soil under the snow cover would be available for plants in early spring. In boreal coniferous forests 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 (Billbrough et al. 2000).

Traditionally, N mineralization in soil is assumed to take place within a temperature range of 5-35 °C, so that the rate of transformation approximately doubles when the temperature increases by 10 °C (Stanford et al. 1973). On the other hand, there is also evidence for microbial activity and nutrient mineralization at lower temperatures, even at sub-zero temperatures down to -7 °C (Clein and Schimel 1995, Kähkönen et al. 2001, Schimel et al. 2004). In laboratory incubations of arctic soils Elberling and Brandt (2003) reported microbial soil respiration to continue at temperatures down to -18 °C. In a field incubation study carried out in a sub-arctic region in northern Sweden Schmidt et al. (1999) observed both mineralization and immobilisation of N and P in soil in winter.

Snow is an effective insulator of heat and thus the soil temperature is typically only a few degrees below zero in snow-covered soil, which enables N transformation processes to continue throughout the winter. Soil microorganisms are more active in upper layers than in deeper soil layers. For example, Persson and Wirén (1995) found that on average 78% of the net N mineralization occurred above a depth of 10 cm and 22% occurred in the 10-50 cm layer in forest soils. In order to model N mineralization and other microbial processes in soil, an accurate method for the calculation of soil surface layer temperatures in winter is needed. A suitable method for accurately calculating soil temperature in seasonally frozen soil could be to couple the differential equations for water and heat flow. These equations can be solved with numerical methods as given e.g. by Harlan (1973), Taylor and Luthin (1978), Guymon et al. (1980) and Jansson and Karlberg (2001).

1.4 Anthropogenic non-point pollution sources

Nowadays in Finland nutrient release from non-point sources (agriculture, forestry and scattered settlement) exceeds the industrial and municipal loads including peat mining and fish farming (Statistics Finland 2004). Agriculture comprises the greatest single source of nutrients to surface waters (Rekolainen et al. 1992, Vuorenmaa et al. 2002) by changing the hydrological and nutrient status of soils. The effects of forestry differ from those of agriculture, as some practices have a strong effect but of brief duration whereas others have a long-lasting effect.

In their natural state peatlands act as sinks but when drained they become sources of inorganic nutrients. Both low flows and peak runoff tend to increase due to drainage (Seuna 1988, Seuna 1990). The increase in runoff ends after about 15-20 years when the water-carrying capacity of ditches decreases and transpiration of growing trees compensates for decreased evaporation (Seuna 1990, Laine et al. 1995, Kenttämies 1998). Both first-time drainage and ditch-cleaning with supplementary drainage were observed to increase inorganic N, mainly $\text{NH}_4\text{-N}$, concentrations in runoff waters (Hynninen and Sepponen 1983, Laine et al. 1995, Joensuu et al. 2001, Åström et al. 2002).

Forest felling, especially clear cuts, increases annual runoff and sharpens peak runoffs in spring and summer due to faster melting of snow, decreased evaporation and interception. This increase is not long-lasting as new vegetation starts to recover

soon after felling (Kenttämies 1998). In clear fellings logging waste is generally left on the site and when it decomposes inorganic nutrients are mineralized. Inorganic nutrients may be leached to groundwater and surface waters (Likens et al. 1970) and increased nitrate concentrations in soil water and in groundwater was found in several studies (Kubin 1998, Soveri et al. 2001, Piirainen et al. 2002, Rusanen 2002). Nitrification is also found to increase in forest felling sites in coniferous forests (Tamm et al. 1974, Dahlgren and Driscoll 1994, Paavilainen and Smolander 1998).

Peat mining causes elevated leaching and concentrations of nutrients and suspended solids in downstream waters (Heikkinen 1990, Klöve 2001). Furthermore, about 19% of Finnish citizens are not connected to municipal sewer networks but have on-site wastewater systems. Private wastewater treatment is usually defective, especially in old single-family houses. Loading of both phosphorus and nitrogen per person is clearly higher from private systems than from sewer networks (Rontu and Santala 1995).

1.5 Environmental policy

Currently, attention is paid in both international and national regulations to decreased loading of nutrients. The Convention on Long-range Transboundary Air Pollution (CLRTAP) under the UN Economic Commission of Europe (UN/ECE) has been the driving force for reducing transboundary air pollution in Europe. In 1999 a protocol to abate acidification, eutrophication and ground-level ozone was signed in Gothenburg (UN/ECE 1999). In EU policy the Water Framework Directive sets new challenges for integrated river basin management. Good ecological and chemical status of surface waters is to be achieved within 15 years after implementing the Directive, which entered into force in December 2000.

In Finland the Finnish Council of State issued in 1998 a Decision-of-Principle on water protection targets to 2005. By the year 2005 anthropogenic phosphorus (P) load should be reduced by 45% and anthropogenic N load by 40% from the estimated loads at the beginning of the 1990s (Ministry of the Environment 1998). The objectives were set individually for different sectors, for example for industry the reduction target was 50% for both P and N. For municipal waste water treatment the target was 35% for P, and in cases when N is the minimum nutrient in receiving waters 50% N removal is also needed. Reduction targets of point pollution should be achieved by implementing environmentally friendly methods of production and by improving waste water purification.

In the beginning of 2004 a treaty came into operation which aimed to prevent deterioration of water quality by setting minimum requirements to the purification efficiency of private wastewater disposal systems. On-site waste water treatment in single-family houses should be improved to fulfill current standards by 2014 (Anon. 2003). Peat mining, like any other industrial production, needs an environmental permit in which limits to emissions are set. Both N and P loads from peat mining areas should be reduced by 30% according to the Decision-of-Principle on the water protection targets to 2005.

The Finnish Agri-Environmental Programme (FAEP), which started in 1995, is the most important policy measure for controlling agricultural nutrient loading (Valpasvuo-Jaatinen et al. 1997, Ministry of Agriculture and Forestry 2004a). In

2002 it covered about 92% of Finnish farms and 93% of the arable area (Ministry of Agriculture and Forestry 2004b). In FAEP environmental support is paid to farmers who undertake 'basic' and 'additional measures', such as preparing a farm environmental management plan, establishing filter strips on the sides of main ditches and water courses and conforming to targeted levels of fertilizer and manure application. In turn, 'special measures' require more efficient environmental protection, e.g. establishment and management of 15 m wide buffer zones, wetlands and sedimentation ponds. In forestry, buffer zones, wetlands and sedimentation ponds are recommended measures against erosion and nutrient leaching caused by forestry practices. Wetlands and sedimentation ponds are a part of environmental permission terms to reduce suspended sediment and nutrient leaching from peat mining areas.

Only fish farming had almost achieved the objectives set in the Decision-of-Principle on water protection targets to 2005 already by 2000. Municipal waste water treatment has now achieved the objective of N reduction, and P reduction should be achieved by the year 2008. Industry has achieved the objective of P reduction but not that of N reduction. In other activities water protection measures need to be further intensified (Silvo et al. 2002, Ministry of the Environment 2005).

1.6 Objectives of this study

The main objective of this work was to study the timing and origin of inorganic N loading from a boreal river basin in northern Finland by the dynamic, semi-distributed INCA-N (Integrated Nutrients in Catchments – Nitrogen) model. The structure of the INCA-N proved to be suitable to describe processes in boreal forests and to allow effective use of the data available from the area (Rankinen 2003). Further, different scenarios of water protection measures and environmental change were studied.

So far the modelling studies of nutrient leaching from Finnish large river basins have been more empirical than process based (e.g. Viro 1955, Bilaletdin et al. 1994, Lepistö et al. 2001) but also dynamic, semi-distributed nutrient leaching models have been applied on river basin scale (Alasaarela et al. 1995, Francos et al. 2001, Granlund et al. 2004). In these early applications the emphasis has been mainly in model calibration and less in quality assurance with phases of validation and uncertainty analysis.

The study area, the Simojoki river basin, is located in Lapland in northern Finland. The river Simojoki represents a Fennoscandian natural river habitat type in the NATURA 2000 network. The dominant human impacts in the area are forestry, agriculture, scattered settlement and atmospheric deposition, and the only industrial activity is peat mining. As the anthropogenic loads are low in the river basin, it provides a suitable research area to study the behaviour of the model and its capability to simulate N processes in northern environments.

No decreasing trends were observed in nutrient (both P and N) concentrations in the river water from 1975 to 2000 (Räike et al. 2003) although improved water protection methods were applied in peat mining and agriculture during the last decade. Standards of sewage treatment in private dwellings has stayed constant

during last decade (Perkkiö 1995, Nenonen 2004). First time forest drainage was most intensive in 1960s and 1970s, and now the emphasis is on ditch cleaning and supplementary drainage (Perkkiö 1995). In 1990s the area of peat harvesting fields did not increase significantly but water protection measures improved (Perkkiö 1995, Vääränen 2000). Lately the main changes in N loading were due to changes in agricultural practices and agricultural land use as FAEP was established to ensure the change in agricultural practices towards higher sustainability (Valpasvuo-Jaatinen et al. 1997).

Scenarios of environmental change were included to assess increasing pressures for nitrogen leaching due to human impact in future. Scenarios of environmental change were included in order to assess the possibilities to improve water protection in the river basin. The scenarios are based on current legislation and they were planned together with local authorities to represent preferred development in the river basin. The main questions for water managers in the Simojoki region are to enhance its favourable status of conservation both by restoration and by improving water quality by diminishing diffuse loading from the river basin. The emphasis in implementing water protection methods in the Simojoki river basin is also in improving the state of the Gulf of Bothnia in the Baltic Sea. Nitrogen from clearly P limited Bothnian Bay flows south to N limited Bothnian Sea where eutrophication has been observed to continue throughout 1980s and 1990s (Pitkänen 2004).

Particular attention in this work was paid to applicability of the model to the study area. In statistics the term internal validity refers to the validity that the relationship between two variables is causal, and the term external validity to the validity that the presumed causal relationship can be generalized to and across alternate measures of the cause and effect (e.g. Cook and Campbell 1979). In this study this approach is broadly adopted so that attention is paid both to correctness of the model structure, and to how well the simulated results correspond to the results of other studies. Simulated annual inorganic N process fluxes and specific loads were compared to values found in the literature or to studies of small research catchments. In this study the GLUE methodology is combined together with the concept of soft data (empirical knowledge) in order to study model behaviour and parameter identifiability.

The main objectives of this study are:

1. Is it possible to follow rigorous calibration and validation scheme in river basin scale model set-up?
2. Is it possible to enhance reliability of model simulations by correct use of all available data?
3. Does the INCA-N model explain the main features of hydrological pattern and seasonality of inorganic N concentrations in river water in the boreal zone?
4. Does over-winter N mineralization explain the relatively high inorganic N concentrations in late winter/early spring?
5. Is it possible to decrease diffuse pollution by allocating suitable water protection measures to all possible sources?
6. How will the expected land use changes in future change total inorganic N load in the Simojoki river basin?

2 MATERIAL AND METHODS

2.1 Simojoki river basin

The river Simojoki discharges to the Gulf of Bothnia in the Baltic Sea (Fig. 1). The river basin (3160 km²) can be subdivided into nine sub-basins (Ekholm 1993). Over the period 1961-1975, annual precipitation varied between 650 and 750 mm and annual runoff between 350 and 450 mm. The total length of the river is 193 km and the elevation drop is 176 m. There are several rapids in the river Simojoki. The combined length of the rapids is 36 km, and most of them are located in the middle and lower reaches of the river. The rapids were cleaned out and canalized during the 1950s in order to facilitate log floating.

The river Simojoki is an unregulated Atlantic salmon river without any major point pollution sources. The streambed belongs to the NATURA 2000 network representing the Fennoscandian natural river habitat type. Groundwater resources in the area are very restricted. Bedrock is covered by ground moraine, mainly sandy till. In the lower reaches of the river there are also river deposits, and some clay areas are located in the river valley. The Simojoki river basin is located partly in the mid-boreal and partly in the northern boreal vegetation zone. Main tree species is pine (*Pinus sylvestris*), but in coastal area there are dense vegetation of alder and willow. Composition of tree stand in sub-catchments is presented in Table 1.

The dominant human impacts in the area are forestry, agriculture, scattered settlement and atmospheric deposition. Peatlands and peatland forests are common in the region. Forest drainage was most extensive during the 1960s and 1970s, and by 1991 over 30% of the total river basin area had been drained. 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 0.1% and agricultural areas 2.7 % of the catchment area (Perkkiö et al. 1995). Grass cultivation for animal husbandry is the most common form of agricultural production.

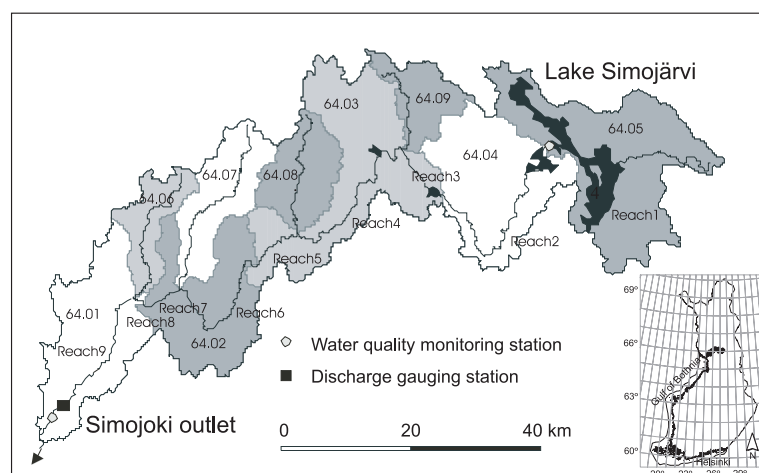


Figure 1. Location of the Simojoki river basin in northern Finland.

Table 1. Distribution of tree species in the sub-catchments of the Simojoki river basin

	64.01 [%]	64.02 [%]	64.03 [%]	64.04 [%]	64.05 [%]	64.06 [%]	64.07 [%]	64.08 [%]	64.09 [%]
Pine	46	49	58	74	82	53	59	47	82
Spruce	3	3	4	3	3	3	3	5	3
Coniferous	5	5	2	1	0	4	2	6	0
Mixed	46	43	36	22	15	40	36	42	15

There are two discharge gauging stations in the river Simojoki, one at the outlet of the river and one in the Hosionkoski rapids. Mean daily flow at the outlet was $37.2 \text{ m}^3 \text{ s}^{-1}$ ($11.8 \text{ l s}^{-1} \text{ km}^{-2}$) during the period 1965-1990. The maximum discharge was $730 \text{ m}^3 \text{ s}^{-1}$ and the minimum discharge $3 \text{ m}^3 \text{ s}^{-1}$. There is one snow measurement line in the river basin but no groundwater monitoring station.

The N concentration data for the river Simojoki were obtained from the water quality data base operated by the Finnish Environment Institute (SYKE) and regional environment centres. The median concentration of $\text{NH}_4\text{-N}$ was $16 \mu\text{g l}^{-1}$ and that of $\text{NO}_3\text{-N}$ $42 \mu\text{g l}^{-1}$ during the period of 1988-2000. Median concentration of total N was clearly higher ($460 \mu\text{g l}^{-1}$) during the same period so that considerable part of the total N load from the Simojoki river basin can be expected to be in the form of organic N. Sampling frequency at the outlet of the Simojoki river was 12-17 samples a^{-1} during the study period 1995-1999.

2.2 Small research catchments

Small catchments are often used when studying hydrology and in particular diffuse loading, because the loads measured at the outlets are assumed to represent loads from areas of corresponding soil type and land use. Kortelainen et al. (1997) estimated long term leaching from forested study catchments in Finland based on daily measured discharge and water quality sampling (approximately 12 samples per year). These 22 study catchments represented typical forest management practises, like forest ditching, clear felling, scarification and low levels of N fertilization up to 110 kg N km^{-2} . Vuorenmaa et al. (2002) estimated agricultural losses of dissolved inorganic N from agricultural catchments in southern Finland in 1991-1995.

The purpose of the network of small research catchments maintained by SYKE is to gather information on the climate and soil conditions typical for the country, including both forested and agricultural catchments. Originally the measurements were only hydrological (Mustonen 1963, Mustonen 1971) and monitoring of water quality was launched in 1962 (Seuna 1983).

The other research basin networks are the five forested catchments of the VALU research project (Finér et al. 1997) and six forested catchments of the Nurmes project (Ahtiainen and Huttunen 1999), located in eastern Finland. These research catchments are used to study the effects of silvicultural treatments using the paired-catchment method (Sallantausta 1986). In this method forestry treatment are carried out in a small catchment, which has a representative control catchment. Both catchments are preferably monitored several years by same methods to get relationships between

different state variables in catchments. Thus it is possible to filtrate out influences caused by other factors than forestry, so the change in water quality can be expressed as specific loading caused by the forestry measure. The Nurmes study started in 1978. Four of the catchments are treated and two are control catchments with a calibration period of 5 years. The VALU research was started in 1991 and the calibration period was 6 years.

In this work estimated long-term inorganic N leaching from these research catchments was used to derive ranges in which simulated inorganic N leaching from different land use classes can vary. Catchments locating in northern part of Finland with representative land use and climate for Simojoki river basin were selected. Only catchments representing agricultural land use are located in southern Finland. In the Hovi catchment (60°42', 24°38') 100% of the area was cultivated and in the Löytäneenoja catchment (61°27', 22°25') 68%, correspondingly.

Two of the study catchments, Kotioja and Ylijoki (both 66°14', 26°15'), are located in the Simojoki river basin in sub basin 64.03. Peatlands cover over 50% of the area in these catchments. Forest drainage was carried out mainly in the 1960s, so that drainage percentages reached 26% and 30%, correspondingly (Seuna 1982, Kortelainen et al. 1997). The two forested catchments where mineral soils are dominating and which are located close to the Simojoki river basin are Vähä-Askanjoki (66°55', 27°69') and Kuusivaaranpuro (66°76', 28°13').

Koivupuro (63°52', 28°30') and Suopuro (63°52', 28°30'), located in north eastern Finland, were used to derive specific inorganic N loading from forest drainage areas. The area of the Koivupuro catchment is 118 ha, of which 57% is mire. In 1983 27% of the area was ditched. During the ten years after treatment the increase in specific loading of inorganic N from treated areas was 0.97 kg N ha⁻¹. The area of the Suopuro catchment is 113 ha, of which 70% is mire. In 1983, 13% of the area was ditched leaving a 10 m wide protective zone between the ditched area and the brook. During the ten years after treatment the increase in specific loading from treated areas was 0.26 kg N ha⁻¹ a⁻¹.

Specific inorganic N loading from forest felling sites was estimated from loading of the small catchments Iso-Kauheja (63°53', 28°37'), where >50% of the area is peatland, and Kangasvaara (63°51', 28°58'), where the area of peatlands is <10%.

2.3 The INCA-N model

The dynamic, process-based and semi-distributed INCA-N (Integrated Nutrients in Catchments - Nitrogen) model (Whitehead et al. 1998, Wade et al. 2002, Wade 2004) integrates hydrology, catchment and river N processes. The term semi-distributed is used, as it is not intended to model catchment land surface in a detailed manner, but to use a land-use class in a sub-basin as a basic modelling unit. River, soil water and groundwater NO₃-N and NH₄-N concentrations and fluxes are produced as daily time series. Three components are included: the hydrological model, the catchment N process model and the river N process model. Sources of nitrogen include atmospheric deposition, the terrestrial environment and direct discharges from point sources.

The INCA-N model can be calibrated to observed discharge and NO₃-N and NH₄-N concentrations along the river. In addition, observed NO₃-N and NH₄-N concentrations

in soil and groundwater in different land use classes and sub-catchments can be used in calibration. The INCA-N model also calculates annual N process loads from every land use class, which may then be compared to experimental data or literature values for these processes in order to estimate whether the N process loads are at the correct level.

Hydrological processes

Hydrological processes in the soil are simulated in the hydrological sub-model. Hydrological input is given to the model as daily time series of hydrologically effective rainfall (*HER*). *HER* defines that part of total incident precipitation which reaches stream channels as runoff, and it is used to drive water flow and nitrogen fluxes through the catchment system. Hydrology within a catchment (either the whole river basin or a sub-catchment) is modelled using a simple two-box approach, with reservoirs of water in the reactive soil zone and deeper groundwater zone. The flow from the zones is calculated by the following equations:

Soil zone

$$\frac{dx_1}{dt} = \frac{1}{T_1} (HER - x_1) \quad (1)$$

Groundwater

$$\frac{dx_2}{dt} = \frac{1}{T_2} (BFIx_1 - x_2) \quad (2)$$

where x_1 and x_2 are output flows of the soil zones ($m^3 s^{-1}$). T_1 and T_2 are time constants (i.e. residence times in days) associated with the zones, and the base flow index (*BFI*) is the proportion of water being transferred to the groundwater zone. Since the hydrological mass balance equations are based on a 1 km² cell, the output is multiplied by the land area within a sub-catchment to calculate the water volume entering the stream.

The river flow model is based on mass balance of flow and uses a multi-reach description of the river system (Whitehead et al. 1998). Flow variation within each reach is determined by a non-linear reservoir model.

$$\frac{dS(t)}{dt} = I(t) - Q(t) \quad (3)$$

where I is inflow ($m^3 s^{-1}$), Q is outflow ($m^3 s^{-1}$), S is storage ($m^3 s^{-1}$) and t is time.

$$S(t) = T(t) * Q(t) \quad (4)$$

where T is a travel time parameter

$$T(t) = \frac{L}{v(t)} \quad (5)$$

where L is the reach length (m) and v is the mean flow velocity in the reach ($m s^{-1}$). Mean flow velocity is related to discharge, Q , through

$$v(t) = aQ(t)^b \quad (6)$$

where a (m^2) and b (-) are constants.

Catchment N processes

The mass balance equations for NO_3 -N and NH_4 -N in both the soil and groundwater zones are solved simultaneously with the flow equations. The key N processes that are solved in the soil water zone in all six land use classes are nitrification, denitrification, mineralization, immobilisation, N fixation and plant N uptake. In the groundwater zone it is assumed that no biochemical reactions occur. The daily inputs to the catchment N process model are Soil Moisture Deficit (SMD), Air Temperature (T_{AIR}) and Actual Precipitation (P).

The change in NO_3 mass in soil, x_3 ($kg\ N\ km^{-2}$) and groundwater, x_4 ($kg\ N\ km^{-2}$) stores are given as:

Soil store

$$\begin{aligned} \frac{dx_3}{dt} = & U_2 \cdot 100 - \frac{x_1 x_3 \cdot 86400}{V_r + x_{11}} - C_3 S_2 \frac{x_3}{V_r + x_{11}} 10^6 \\ & + C_4 \frac{x_5}{V_r + x_{11}} 10^6 - C_1 \frac{x_3}{V_r + x_{11}} 10^6 + C_2 \cdot 100 \end{aligned} \quad (7)$$

Groundwater store

$$\frac{dx_4}{dt} = \frac{BFLx_1 x_3 \cdot 86400}{V_r + x_{11}} - \frac{x_2 x_4 \cdot 86400}{x_{12}} \quad (8)$$

where x_3 is the NH_4 mass in the soil store ($kg\ N\ km^{-2}$), U_2 is the input rate of NO_3 ($kg\ N\ ha^{-1}\ day^{-1}$), V_r and x_{11} are the retention and drainage volumes in the soil ($m^3\ km^{-2}$), and x_{12} is the groundwater drainage volume ($m^3\ km^{-2}$). The parameters C_1 , C_2 , C_3 and C_4 are the rates of denitrification (day^{-1}), fixation ($kg\ N\ ha^{-1}\ day^{-1}$), plant NO_3 uptake (day^{-1}) and nitrification (day^{-1}) and S_2 is seasonal plant growth index. The N balance equations are also based on a $1\ km^2$ cell and therefore the output from the equations is multiplied by the land area within a sub catchment to calculate the N mass entering a reach of the stream.

The change in NH_4 mass in soil, x_5 and groundwater, x_6 stores ($kg\ N\ km^{-2}$) is given as:

Soil store

$$\begin{aligned} \frac{dx_5}{dt} = & U_3 \cdot 100 - \frac{x_1 x_5 \cdot 86400}{V_r + x_{11}} - C_7 S_2 \frac{x_5}{V_r + x_{11}} 10^6 \\ & - C_4 \frac{x_5}{V_r + x_{11}} 10^6 + C_5 100 - C_6 \frac{x_5}{V_r + x_{11}} 10^6 \end{aligned} \quad (9)$$

Groundwater store

$$\frac{dx_6}{dt} = \frac{BFLx_1x_5 \cdot 86400}{V_r + x_{11}} - \frac{x_2x_6 \cdot 86400}{x_{12}} \quad (10)$$

where U_3 is the input rate of NH_4 load ($\text{kg N ha}^{-1} \text{ day}^{-1}$). The constants C_5 , C_6 and C_7 are the rates of mineralization ($\text{kg N ha}^{-1} \text{ day}^{-1}$), immobilisation (day^{-1}), which covers both microbial immobilisation and retention due to cation exchange, and plant NH_4 uptake (day^{-1}), respectively. Details of catchment N processes are given in Wade et al. (2002) and in Wade (2004).

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

$$f(SMD) = \frac{SMD_{\max} - SMD}{SMD_{\max}} \quad (11)$$

where SMD is the daily soil moisture deficit (mm) and SMD_{\max} the maximum soil moisture deficit (mm). Temperature response (Bunnel et al. 1977) for all temperature-dependent processes is calculated as:

$$f(T_Z) = t_{Q10}^{(T_Z - t_{Q10bas})/10} \quad (12)$$

where t_{Q10} (-) and t_{Q10bas} ($^{\circ}\text{C}$) are parameters and T_Z ($^{\circ}\text{C}$) is soil temperature. The parameter t_{Q10} is the factor change in rate with a 10°C change in temperature and the parameter t_{Q10bas} is base temperature for N processes at which the response is 1. Microbial processes are assumed to cease when soil temperature decreases below a limiting value which can be defined independently for each process.

A simplified method for calculating soil temperature in seasonally frozen soil is based on coupling the differential equations for water and heat flow (III). The basic assumptions behind the two equations are the law of conservation of mass and energy, and water and heat flow as a result of gradients in water potential (Darcy's Law) and temperature (Fourier's law). The calculations of water and heat flows are based on soil properties. However, the combined solution of water and heat flow is impractical due to wide spatial and temporal heterogeneity in the soil properties, so three simplifications were made. Firstly, soil water content was assumed to be constant throughout the computation. Secondly, no heat flow was assumed at the bottom of the profile. Thirdly, the effect of snow density on thermal conductivity of snow was neglected in the model. Soil temperature is calculated as:

$$T_*^{t+1} = T_Z^t + \frac{\Delta t K_T}{C_A (2Z_S)^2} [T_{AIR}^t - T_Z^t] \quad (13)$$

where T_*^{t+1} is the unknown soil temperature at depth Z_s at time $t+1$, T_Z^t is the calculated soil temperature from the previous day ($^{\circ}\text{C}$), K_T ($\text{W m}^{-1} \text{ }^{\circ}\text{C}$) is soil thermal conductivity, and T_{AIR}^t ($^{\circ}\text{C}$) is measured air temperature. The apparent heat capacity term C_A ($\text{J m}^{-3} \text{ }^{\circ}\text{C}^{-1}$) includes both volumetric specific heat of soil and energy released when water is frozen or consumed when frozen soil melts (III). The influence of snow cover is taken into account by an empirical equation (Rankinen et al. 2004) where T_Z^{t+1} is calculated soil temperature ($^{\circ}\text{C}$):

$$T_Z^{t+1} = T_*^{t+1} e^{-f_s D_s} \quad (14)$$

corrected by snow depth D_s (m) which can be calculated by a simple degree-day model, and f_s is an empirical damping parameter (m^{-1}).

River N processes

The NO_3 , x_{23} and NH_4 mass stored, x_{24} (kg N) in the reach are given by:

$$\frac{dx_{23}}{dt} = S_5 - \frac{x_{22} x_{23} \cdot 86400}{x_{29}} - \frac{C_{11} a_{5,t-1} x_{29}}{1000} + \frac{C_{10} a_{6,t-1} x_{29}}{1000} \quad (15)$$

$$\frac{dx_{24}}{dt} = S_6 - \frac{x_{22} x_{24} \cdot 86400}{x_{29}} - \frac{C_{10} a_{6,t-1} x_{29}}{1000} \quad (16)$$

where S_5 and S_6 are the input mass (kg N) from upstream, x_{22} is reach flow ($\text{m}^3 \text{ s}^{-1}$), x_{29} is the reach volume (m^3), and C_{10} (day^{-1}) and C_{11} (day^{-1}) are the in-stream nitrification and denitrification rates, respectively. In-stream ammonium and nitrate concentrations (a_5 and a_6 mg l^{-1}) are taken from the previous time step.

2.4 Input data

Hydrological input data (HER , T_{AIR} , SMD , P) were taken from the output of the WSFS (Watershed Simulating and Forecasting System). The WSFS has been widely used in Finland since 1990 (Vehviläinen 1994). It includes model applications for 20 watersheds ranging from 600 km^2 to 60000 km^2 and covers 85% of the country. The watersheds are divided into small homogeneous sub-basins according to elevation, land use, snow distribution and lakes. Each model consists of 10-100 independent sub-basins ($50\text{-}500 \text{ km}^2$) with simulations of areal precipitation, temperature, water equivalent of snow, soil moisture, changes in subsurface and groundwater storage, and formation of runoff. 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 in common use for flood forecasts, water resources management and supervision purposes.

Atmospheric N deposition was calculated using the regional nitrogen transport and deposition model DAIQUIRI (Syri et al. 1998, Kangas and Syri 2002). The model incorporates transfer matrices calculated from the results of the EMEP Lagrangian transport model. Each two-dimensional matrix describes the deposition in every grid cell due to emissions in one European country. Total deposition is obtained by summing the deposition fields caused by each country. The DAIQUIRI calculation grid has a resolution of $\frac{1}{4}^{\circ} \times \frac{1}{8}^{\circ}$, which is about 14 km*14 km in southern Finland.

Land use classes were derived from the satellite image-based land use and forest classification of Finland (Vuorela 1997), and supplemented with satellite image-based maps of final cuttings on mineral and organic soils, provided by the Finnish Forest Research Institute. Land use data covered the period 1986-1994. Forest land was assumed to represent commercial forests where typical silviculture is practiced (drainage, thinning, regeneration or clear cut), but it has been at least 10 years since the last measures were made. New treatment areas (1-10 years old) were simulated separately. Forest on mineral soil (ForMin) covers 35%, new 1-10 years old clear felling (ForMinCu) on mineral soil 4%, forest on organic soil (ForOrg) 52%, new 1-10 years old clear felling (ForOrgCu) on organic soil 1%, agriculture (Agri) 2% and open surface water 6% of the whole Simojoki river basin area. The forest areas on organic soil were assumed to be drained during the 1960s or 1970s. Ground vegetation was assumed to start to recover in new (1-10 years old) forest felling areas. As forest fertilization in the Simojoki area came to the end in early 1980's no forest fertilization was assumed. Land use classes, atmospheric deposition and number of houses outside municipal sewer networks are presented in Table 2.

Table 2. Land use, atmospheric deposition and number of dwellings outside municipal sewer network in the sub-catchments of the Simojoki river basin

Parameter	Dimension	Sub-catchment									
		64.01	64.02	64.03	64.04	64.05	64.06	64.07	64.08	64.09	
Area	[km ²]	412	375	557	446	630	147	245	201	147	
NO ₃ deposition	[kg N ha ⁻¹ a ⁻¹]	1.5	1.5	1.5	1.4	1.4	1.5	1.5	1.5	1.4	
NH ₄ deposition	[kg N ha ⁻¹ a ⁻¹]	0.8	0.8	0.8	0.7	0.7	0.8	0.8	0.8	0.7	
Dwellings outside municipal sewage treatment	[number]	789	372	713	219	634	4	34	75	89	
ForMin	[%]	37	29	37	32	43	28	26	40	41	
ForMinCu	[%]	4	3	4	3	3	2	4	6	3	
ForOrg	[%]	52	63	52	59	35	67	68	52	47	
ForOrgCu	[%]	1	1	0	0	0	1	1	0	0	
Arable	[%]	4	2	3	1	1	0	0	1	1	
Water	[%]	2	2	4	5	18	2	1	1	8	

All the fields were under perennial grass ley. All the farmers were assumed to follow the same typical cultivation practices which were derived from an FAEP interview studies of years 1995-1999 (Palva et al. 2001, Pyykkönen et al. 2004). Maximum fertilization levels either for mineral (154 kg N ha⁻¹) or organic fertilizers (165 kg N ha⁻¹) were based on the recommended fertilization according to the FAEP basic measures and the FAEP interview study.

2.5 Effluent time series

Forest drainage (Paper VI)

Between the years 1988 and 1998 14 700 ha of forests were ditched, which covered about 5% of the total catchment area. The areas of first-time drainage and ditch cleaning and supplementary ditching presented in Table 3. were summed. Data from the Koivupuro catchment was used in basic calibration and data from the Suopuro catchment in a scenario run of water protection measures.

Calculated specific load from treated areas was added as a percentual increase to simulated daily leaching from the land use class 'Forest on organic soil'. Discharge from ditched areas was assumed to increase by 0.6% per ditching percentage throughout the 10-year period (Seuna 1988, Seuna 1990) which was assumed to be the duration of increased loading. Inorganic N loading from ditched areas was assumed to be only in the form of NH₄-N (Laine et al. 1995).

Peat mining (Paper VI)

Inorganic N loads from peat mining areas were taken from annual reports of pollution load prepared for the water authorities (Vääränen 2000, Kaikkonen and Salo 2004). Loading percentages of the non-productive season from the Lumiaapa peat mining area were adapted to other peat mining areas which were monitored only in summer. For the years 1995-1997 peat mining areas (1365 ha) reported in the year 1995 were used, and for the years 1998 and 1999 peat mining areas (1409 ha) reported in 1999 were used.

Table 3. Forest areas ditched between 1988 and 1998

Year	First-time drainage [ha]	Ditch cleaning and supplementary drainage [ha]
1988	2076	968
1989	1928	1037
1990	518	644
1991	800	408
1992	1282	860
1993	274	954
1994	162	1018
1995	136	336
1996	86	373
1997	62	527
1998	0	208

Scattered settlement (Paper VI)

Catchment scale N loading from the areas of scattered settlement was based on research results of nutrient loading from single-family houses with different on-site waste water systems (Vilpas et al. 2005). Numbers of persons and houses not included in the municipal sewerage system were taken from the official statistics of the year 2000 (Anon. 2000). Information of on-site waste water system types was based on surveys conducted by the local authorities (Perkkiö et al. 1995, Nenonen 2004).

2.6 Scenarios of land use change and water protection measures

Evaluating agricultural land use (Papers V, VI)

Impacts of EU's agricultural policy reforms on agricultural land use were assessed by the DREMFIA model (Lehtonen 2001, Lehtonen 2004). The model can be used to evaluate the effects of different agricultural policies simultaneously on land use, animal production and agricultural income. The output of the DREMFIA model includes year-to-year variation in the total area of agricultural land as well as in the area of main crops and in the fertilization levels used as input to the INCA-N model (V). The total area of agricultural land is limited by the existing available agricultural land, including some marginal land currently not fully utilised and by mineral soil types suitable for agriculture.

Since 1995, the Finnish agricultural support measures have been based on the Common Agricultural Policy (CAP) of the EU. CAP was reformed in the Agenda 2000 agreement, in which producer prices of cereal crops, milk and beef were reduced. In June 2003 CAP was further reformed and a major part of CAP supports was decoupled from production. Finland has decided to move over to the reformed CAP in 2006.

The Base (BASE) scenario followed Agenda 2000 which was assumed to stay unchanged until 2020. National support, investment support and environmental support were assumed to stay at the level of 2004. EU prices of dairy products as well as the producer price of milk (exogenous in the DREMFIA model) were assumed to decrease by 15%, whereas domestic prices (endogenous in the model) decreased by 12%. In the BASE scenario the total utilised area of agricultural land will increase by 25% (up to 5560 ha) by 2010 from level in 1995. After 2010 it will slowly decrease back to the 1995 level by 2020. Grass cultivation will remain the main production form accounting for 98% of the field area.

The Mid Term Review (MTR) scenario followed the CAP reform agreed in 2003. In the MTR scenario the quota system for milk was assumed to remain, but the producer price in the EU was assumed to decrease by 22% by 2007 while domestic milk prices remained slightly higher due to reduction of aggregate milk production in Finland. In the MTR-scenario, the total area of agricultural land will increase by 35% (up to 6390 ha) by the year 2010 and then level off. Grass cultivation stays the main form of agricultural production at the Simojoki river basin (93% of the agricultural area). However, by the year 2020 milk production will have decreased by 40% and consequently the area of grass cultivation decreases (down to 52% of agricultural land) and the area of green fallow increases (up to 46%). This happens as a result of decreasing milk price and de-coupled payments decrease dairy investments while set-aside land is eligible for de-coupled payments. Also, some farms change to grain production (2% of the field area).

Changes in forest management areas (Papers II, VI)

An assumed increase of 20% in forest felling areas (CUT scenario) is based on Finland's National Forest Programme (Ministry of Agriculture and Forestry 1999). In the forest management plan of Lapland the volume of thinnings is planned to increase but the volume of clear fellings will remain at the present level in the years 2001-2005 (Riissanen and Härkönen 2001). On the other hand demand for forest drainage, especially for ditch cleaning and supplementary drainage, is seen in the future. The planned area for remedial forest drainage is 10 700 ha for the whole of Lapland over the years 2001-2005. However, as no official targets for forest drainage in the Simojoki river basin were available, the total area of forest drainage was assumed to remain at the present level in the near future.

Water protection measures in agriculture, forestry and scattered settlement (Paper VI)

Crop production in the Simojoki river basin is mainly perennial grass, and thus there is vegetation cover on the fields throughout the year. Farmers are also assumed to follow both recommended fertilization levels and times to spread manure according to FAEP basic measures. In this case the most effective water protection measures are buffer zones and wetlands which belong to special measures of FAEP.

In agricultural areas 15 m wide buffer zones established along rivers may decrease inorganic N leaching in surface water by 30%-50% (Puustinen 1999) and wetlands from upper parts of the catchment by up to 30% (Koskiaho and Puustinen 2004). As most of the agricultural land in the Simojoki river basin is located along the river, both buffer zones and wetlands were assumed to be established on about 15% of the fields. This combination would lead to max. 10% reduction in inorganic N leaching from agricultural areas. As current version of the INCA-N model does not include buffer zones and wetlands explicitly, their influence was taken into account in Catchment N sub-model by increasing immobilisation and denitrification in land use class 'Agriculture'.

The effects of buffer zones on forest drainage areas were assumed by calculating the specific load of inorganic N using the values of the Suopuro research catchment where the buffer zones were applied. No special water protection measures on new (1-10 years old) forest felling sites were assumed.

As the dwellings in the Simojoki river basin are relatively old, inhabitants in single-family houses were assumed to renew their sewage treatment system to reach the current recommended purification capacity, but not to change the system type. The loading from renewed subsurface disposal systems was assumed to decrease by 30%.

Atmospheric N deposition (Paper II)

Expected future N deposition (UNE) in 2010 was calculated from the emission reduction obligations agreed within the UN/ECE (1999). The maximum technically possible reduction of N deposition was estimated using the Maximum Feasible Reductions (MFR) emission scenario compiled by IIASA (International Institute for Applied Systems Analysis). The average N deposition decreased from 2.3 kg ha⁻¹ a⁻¹ to 2 kg ha⁻¹ a⁻¹ (UNE scenario) and to 1.4 kg ha⁻¹ a⁻¹ (MFR scenario).

2.7 Uncertainty analysis

The parameter uncertainty was evaluated by a Bayesian Monte Carlo method, GLUE (Beven and Binley 1992). In this method a large number of model runs are made with many different randomly chosen parameter combinations selected from predefined probability distributions. The acceptability of each run is evaluated against observed values, and if the acceptability is below a certain threshold, the run is considered to be ‘non-behavioural’ and it is removed from further analysis. Outputs from the retained runs are likelihood-weighted and ranked to form a cumulative distribution of the output variable.

In a Bayesian method each parameter set is associated with a likelihood:

$$L(\Theta|y) = L_1(y|\Theta)L_0(\Theta)/C \quad (17)$$

where $L_0(\Theta)$ is a prior likelihood of parameter set Θ , $L_1(y|\Theta)$ is the likelihood measure calculated for the simulation of the observed variable y by the parameter set Θ and $L(\Theta|y)$ a posterior likelihood of the parameter set Θ given the new observations y , and C is the scaling factor.

In this study minimum and maximum values were specified for all parameters and uniform a priori distributions were used. The Nash and Sutcliffe coefficient of efficiency (Nash and Sutcliffe 1970) was used as a goodness-of-fit criterion and as the likelihood measure:

$$R_{eff} = \frac{S_M - S_E}{S_M} \quad (18)$$

$$S_M = \sum_{i=1}^{N_D} (M_i - M_{mean})^2$$

$$S_E = \sum_{i=1}^{N_D} (M_i - C_i)^2$$

where M_i are measured values and C_i simulated values. R_{eff} is the likelihood measure in the case that only hard data (runoff and inorganic N concentrations) is used in the GLUE analysis.

The parameters studied in the uncertainty analysis were selected on the basis of expert judgement and a previous sensitivity analysis study (Rankinen et al. 2002). Three different combinations of 20 parameters were included in the GLUE analysis. In the first combination, parameters defining N mineralization, immobilisation, plant NO_3 uptake and plant NH_4 uptake in five land use classes (ForMin, ForMinCu, ForOrg, ForOrgCu, Arable) in the catchment N model were analyzed. In the second combination mineralization, nitrification, denitrification, N fixation, along with the limiting minimum temperature for these processes, were selected. In the third combination the parameters a and b defining river flow velocity in the hydrological model, and nitrification and denitrification rates in the river N model were analyzed.

Given the relatively large number of parameters, the information contained in runoff and inorganic N concentrations (hard data) is insufficient for the identification of parameter values through calibration. Consequently, parameter uncertainty would

be expected to be high. Empirical knowledge of the system behaviour (soft data) enable additional judgment of model simulations in more process-based ways than using only the available hard data. In this study observed inorganic N leaching, vegetation N uptake and N mineralization in the main land use types were used as soft data. Soft data was used to evaluate aspects of the model simulations for which there is no hard data available.

When comparing model simulations or parameter values with soft data, there may be a relatively wide range of acceptable simulations or values. Furthermore, there may be a range of values that fall between “fully acceptable” and “not acceptable” based on the experimentalist’s experience in the field and other synoptic measurements. Fuzzy measures of acceptance can be used to consider these ranges. For each soft data type a trapezoidal function (Eq. 19) was defined to compute the degree of acceptance from the corresponding simulated quantity or parameter value. This trapezoidal function is a simple way to map experimentalist experience into a quantity, which can then be optimized (Seibert and McDonnell 2002):

$$R_{soft} = \begin{cases} 0 & \text{if } x \leq a_1 \\ \frac{x - a_1}{a_2 - a_1} & \text{if } a_1 \leq x < a_2 \\ 1 & \text{if } a_2 \leq x < a_3 \\ \frac{a_4 - x}{a_4 - a_3} & \text{if } a_3 \leq x < a_4 \\ 0 & \text{if } x > a_4 \end{cases} \quad (19)$$

where x is a simulated value, a_2 and a_3 are limits for the range in which the simulated value is fully acceptable and limits a_1 and a_4 define the range in which the simulated value is partly acceptable. If the simulated value falls outside both ranges it is either too low or too high and thus unacceptable.

If the simulated value falls outside both ranges it is either too low or too high and thus unacceptable. Soft data limits for different land use classes and processes are based on preconceptions originating from other, different studies and these limits are listed in Table 4. These limits should not be selected to be too narrow because then the best model might be discarded. It should be pointed out that in this study soft data is basically hard data extracted from previous field studies. In this way soft data, which is usually considered to be non-numerate observations of the system, is quantified and it can be used both in model calibration and in the GLUE analysis.

The total efficiency R_{tot} is used both as an optimization criterion and likelihood measure for discharge and inorganic N concentrations when soft data criteria is included

$$R_{tot} = R_{eff} R_{soft,1} R_{soft,2} \dots R_{soft,n} \quad (20)$$

Equation 20 implies that if any of the soft criteria go to zero the total goodness-of-fit is also zero.

Table 4. Soft data limits for different land use classes and processes

Land use class	Process	a1 [kg N ha ⁻¹ a ⁻¹]	a2 [kg N ha ⁻¹ a ⁻¹]	a3 [kg N ha ⁻¹ a ⁻¹]	a4 [kg N ha ⁻¹ a ⁻¹]	Reference
ForMin	NO ₃ leaching	0	0.07	0.15	1	Kortelainen et al. (1997)
ForOrg		0	0.1	0.3	1	Kortelainen et al. (1997)
Arable		0	5	15	30	Vuorenmaa et al. (2002), Turtola and Kempainen (1998)
ForMin	NH ₄ leaching	0	0.05	0.3	1	Kortelainen et al. (1997)
ForOrg		0	0.06	0.4	1	Kortelainen et al. (1997)
Arable		0	1	3	5	
ForMin	N uptake	10	15	40	100	Mälkönen (1974)
ForOrg		10	15	40	100	Finér (1989)
Arable		50	70	270	300	statistics of yields
ForMin	N mineralization	0	20	80	120	Persson and Wirén (1995)
ForOrg		0	20	70	120	
Arable		20	60	150	250	Lindén et al. (1992), Sippola (2000)

3 RESULTS AND DISCUSSION

3.1 Calibration of the INCA-N model against observed discharge and inorganic N concentrations

3.1.1 The INCA-N model set-up (Papers I, II, VI)

The original calibration described in papers I and II was updated for the current model version (INCA1v9) and improved to use more exact data of N leaching from small research catchment studies (VI). The original model set-up (II) was calibrated against discharge and inorganic N concentration data for the years 1994-1996 and validated by the traditional split-sample method using data of the years 1993 and 1997. The model behaviour was not tested through the required transition regime (Klemeš 1986) because in reality it was not possible to find a representative river basin which had already undergone similar land use changes as studied in the Simojoki river basin. Multi-site calibration to individual sub-catchments (Refsgaard 1997) was not used as monitoring data from representative sub-catchments was not available and land use in different sub-catchments was relatively homogeneous.

In paper VI the input data for the years 1995-1999 was used in simulations as these years corresponded both to the land use classification and forest drainage data, and covered the first programme period of FAEP. Average discharge during this period ($40 \text{ m}^3 \text{ s}^{-1}$) was close to the long term average. All the results presented in this study were updated to follow simulations for the period 1995-1999.

Updating of the model calibration followed the procedure described in paper II so that the values of model parameters were adjusted to obtain simulated and observed discharge and inorganic N concentrations close to each other. Snow-line measurements were used to calibrate snow depth. Forest drainage tends to cause long-lasting increase in runoff and therefore new forest felling areas (1-10 years old) were separated from new forest drainage areas. Inorganic N loading data from forest drainage areas were added as effluent time series, as well as loading from scattered settlement and peat mining areas which were updated to use more accurate data than in paper II. In order to allow full use of non-uniform additional data, no separate validation process was conducted at this stage.

Simulated annual inorganic N process fluxes and specific loads from different land use classes were compared to values from literature and from small research catchment studies. Particular attention was paid to literature studies carried out in boreal environments, but two sites in southern Sweden from the study of Persson and Wirén (1995) were also included. They were assumed to represent northern conditions, for example annual mineralization was in the range reported by Smolander et al. (1998a) and Smolander et al. (1998b).

In this study net mineralization as the difference between gross mineralization and microbial immobilisation was used in calibration, as very little measured values of microbial immobilisation corresponding different land uses was available. Small values of immobilisation rates was allowed to cover retention of $\text{NH}_4\text{-N}$ due to cation exchange. Mineralization and immobilisation was assumed to cease at -5°C , nitrification at -2°C and denitrification at 0°C .

The land use change and water protection scenarios were run for the same period 1995-1999 than the basic model calibration. The relative importance of these scenarios was indicated by the percentage change in annual inorganic N fluxes compared to basic model calibration run above.

3.1.2 Calibration of the hydrological submodel (Papers I, VI)

The structure of the INCA-N model allows hydrological input data to be added to the INCA-N model as one common data set for the whole catchment (lumped input data) or as separate data sets for each of the nine sub-basins (semi-distributed input data). These two approaches to adding input data were compared in paper I.

The simulated discharges based on semi-distributed input data fitted better to the observed values than those based on lumped input data. The annual water balances over the three-year period improved when semi-distributed data was used. Seasonal water balances were adequate during the spring months April-June in each calibration year. The timing of flow peaks was simulated rather well with both approaches, although the semi-distributed input data gave a more realistic simulation of low flow periods and the magnitude of spring flow peaks. Under present climatic conditions most of the N export occurs during periods with high flow, and therefore the timing and magnitude of high flow peaks are important when simulating N fluxes. Also Krysanova et al. (1999) found that the estimated flow was improved when applying the distributed version of the HBV model instead of the lumped version.

Observed and simulated discharge using semi-distributed input data at the outlet of the river Simojoki are presented in Figure 2a. The fit between simulated and observed discharge was good with a Nash and Sutcliffe efficiency R_{eff} of 0.76 (VI). The discharge observations during late autumn/early winter may be uncertain due to ice formation in the river (Hyvärinen 1986). Ice damming, measured as increase in water level, is maximal when discharge is high at the beginning of winter and decreases towards the end of the winter. In most rivers damming is a rather random process which varies during the winter and between years. Although the observations are corrected manually on the basis of information from the local observer, a periodical overestimation of river discharge cannot be excluded.

The hydrological inputs relate to the whole river basin or to the sub-basins rather than to the individual land use classes. The velocity of the water passing through the soil layers can be adjusted by residence times, and the distribution of water between the soil reactive zone and the groundwater zone is adjusted by the base flow index. However, these parameters do not influence the actual amount of water discharging from a certain land use class. Care should be taken when land use changes causing changes in runoff are simulated. In paper VI, loading from forest drainage areas was added as an effluent time series because forest drainage is known to have a long-lasting effect on runoff.

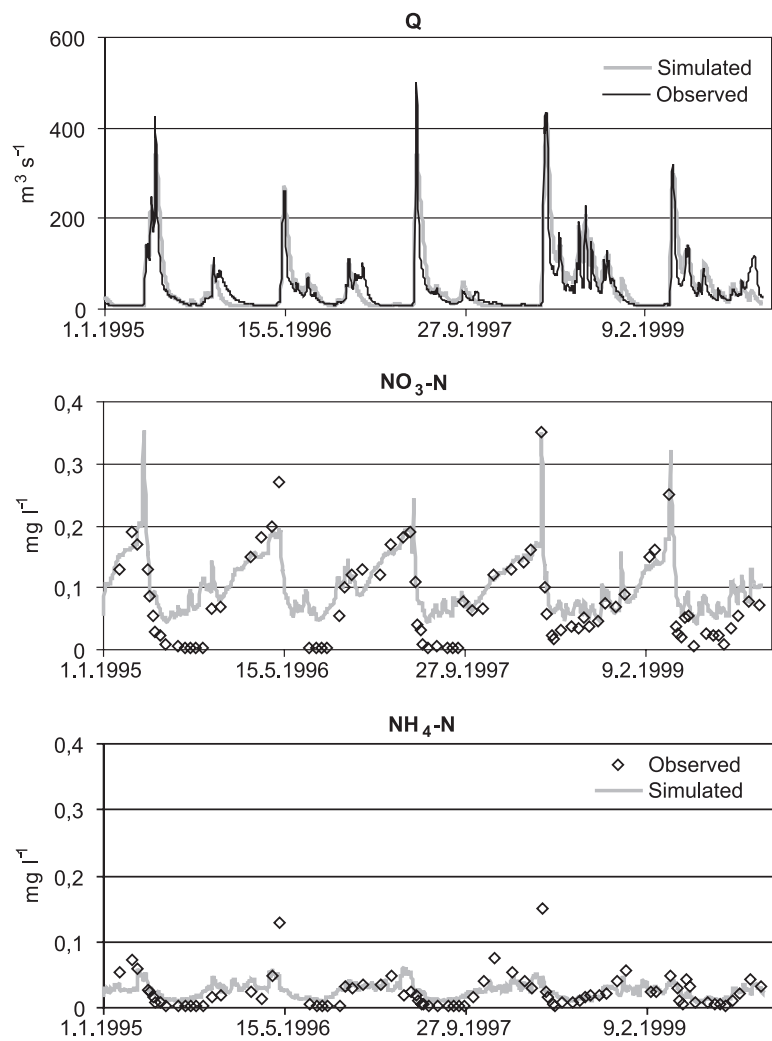


Figure 2. Simulated and observed discharge and inorganic N concentrations at the outlet of the river Simojoki

3.1.3 Seasonal inorganic N concentrations (Papers I, III, IV, VI)

The INCA-N model represented the seasonal dynamics in $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations with R_{eff} efficiencies of 0.61 and 0.30, respectively (Fig. 2b,c). The fit between simulated and observed inorganic N concentrations during the dormant season was good, but during the growing season observed inorganic N concentrations in the river water decreased down to the detection limit presumably due to high vegetation uptake and retention processes, whereas simulated concentrations remained at a higher level (VI).

In long time series of inorganic N concentrations in the river Simojoki, both the median concentration of $\text{NO}_3\text{-N}$ and the inter-annual variation were highest in April and $\text{NO}_3\text{-N}$ concentration decreased during the growing season to an almost negligible level (I). The seasonal pattern of $\text{NH}_4\text{-N}$ concentrations was similar to that of $\text{NO}_3\text{-N}$, except that the concentration levels were lower. On average, inorganic N concentrations in the river water were low ($\leq 0.5 \text{ mg l}^{-1}$).

In INCA-N simulations, atmospheric deposition was assumed to accumulate in the snow pack during winter and to flush to soil water when the snow began to melt (IV). Accumulated deposition during winter was not sufficient to alter river water inorganic N concentrations significantly. When N processes in sub-zero temperatures down to $-5 \text{ }^\circ\text{C}$ were included the INCA-N model was able to simulate the increasing inorganic N concentrations in the river water in winter (IV). This limiting temperature value coincides with the temperatures at which microbial activity was found in similar vegetation and soil types (Schmidt et al. 1999, Kähkönen et al. 2001).

Further, a physically based method to calculate soil temperature (III) was included in the INCA-N model to improve simulated soil temperatures in winter under the snow pack. Three simplifications were made to derive the proposed soil temperature model from coupled water and heat flow equations. Firstly, soil water content was assumed to be constant throughout the computation. Secondly, no heat flow was assumed at the bottom of the profile. Thirdly, the effect of snow density on thermal conductivity of snow was neglected in the model. This simple model simulated soil temperature well in the uppermost soil layers where most of the nitrogen processes occur. The small number of parameters enabled it to be included easily in catchment scale models.

3.2 Simulated N process fluxes and loads

3.2.1 N balance of the Simojoki river basin (Papers I, II, VI)

Simulated annual N process fluxes were within the range reported in the literature (Table 5). The main N input in undisturbed forested areas was from N mineralization, but agricultural areas were assumed to be fertilized. Simulated annual net N mineralization, 47 kg ha^{-1} in the whole river basin, is one order of magnitude higher than atmospheric inorganic N deposition, which in the Simojoki river basin varied between 2.1 and $2.3 \text{ kg N ha}^{-1} \text{ a}^{-1}$ (II). Simulated N fixation in forested areas was less than 1 kg N ha^{-1} which is in range reported by Granhall and Lindberg (1978), Granhall and Lindberg (1980) and Granhall (1981), but lower than N fixation potential ($1.5\text{-}2 \text{ kg N ha}^{-1}$) reported by DeLuca et al. (2002). Denitrification and nitrification rates in boreal forests on mineral soil are reported to be negligible except in forest cut areas (Martikainen 1984, Smolander et al. 1998b).

Table 5. Simulated N fluxes from different land use classes and observed N fluxes from corresponding land use types

Process/Land use	Simulated [kg ha ⁻¹ a ⁻¹]	Observed [kg ha ⁻¹ a ⁻¹]	Reference
<i>N uptake</i>			
ForMin	23-31	16-51	Mälkönen (1974), Helmisaari (1995), Finér et al. (2003)
ForMinCu	18-25	11-23	Mälkönen (1974)
ForOrg	21-28	26-42	Finér (1989)
ForOrgCu	21-26		
Agriculture	208-227	225 ^a	
<i>Mineralization</i>			
ForMin	47-61	15-120	Persson and Wirén (1995) ^b , Smolander et al. (1998a)
ForMinCu	48-61		
ForOrg	38-49		
ForOrgCu	53-61		
Agriculture	83-107	210	Lindén et al. (1992), Sippola (2000)
<i>Nitrification</i>			
ForMin	0.4-0.5	0-7	Martikainen et al. (1994), Persson and Wirén (1995) ^b
ForMinCu	2.4-3.0	13	Persson and Wirén (1995) ^b
ForOrg	2.8-3.4		
ForOrgCu	8.5-9.7		
Agriculture	84-96		
<i>Denitrification</i>			
ForMin	0.1	<1	Gundersen and Bashkin (1994)
ForMinCu	0.3-0.4	<0.5	Smolander et al. (1998b)
ForOrg	0.1-0.2	0-4.7	Martikainen (1984), Martikainen et al. (1994) Regina et al. (1996)
ForOrgCu	2.2-2.7	<1	Nieminen (1998)
Agriculture	17-21	1-19	Saarijärvi et al. (2004), Pihlatie (2001)
<i>N Fixation</i>			
ForMin	0.14	0.31-3.8	Granhall and Lindberg (1978), Granhall and Lindberg (1980)
ForMinCu	0.14	0.27	Granhall and Lindberg (1980)
ForOrg	0.13-0.14	<1-2	Granhall (1981)
ForOrgCu	0.13-0.14	<1-2	Granhall (1981)
Agriculture	-	~4	Rekolainen et al. (1992)

^a Based on statistical information of yields^b Low-leaching sites Farabol 1 and Skogaby

Simulated inorganic N leaching was within the range of observed leaching from small research catchment studies (Table 6). Leaching of inorganic N from study catchments representing natural background areas is typically lower than leaching from catchments representing managed commercial forests as investigated in this study. For example, long-term NO₃-N leaching was less than 0.04 kg ha⁻² a⁻¹ and NH₄-N leaching was below 0.03 kg ha⁻² a⁻¹ from unmanaged study catchments in eastern Finland (Ahtiainen and Huttunen 1999).

Simulated inorganic N leaching from agricultural areas was somewhat lower than measured values in areas representing more southern conditions and different cultivation practices. According to Turtola and Kemppainen (1998), inorganic N leaching from perennial grass ley fertilized by slurry and mineral fertilizer was approximately 7 kg ha⁻¹ a⁻¹ in a field scale experiment in northern Finland.

The simulated average NO₃-N load was 0.48 kg ha⁻¹ a⁻¹ and NH₄-N load was 0.12 kg ha⁻¹ a⁻¹ from the whole river basin during the period 1995-1999. This corresponded a mean value of 0.13 kg km⁻² d⁻¹ for NO₃-N load and 0.03 kg km⁻² d⁻¹ for NH₄-N load. Minimum values were 0.01 kg km⁻² d⁻¹ and 0 kg km⁻² d⁻¹, and maximum values 1.63 kg km⁻² d⁻¹ and 0.32 kg km⁻² d⁻¹, correspondingly. Simulated leaching from terrestrial part of sub-catchments (Catchment N sub-model) is presented in Table 7.

Table 6. Simulated inorganic N leaching from different land use classes and observed inorganic N leaching from small research catchments

Land use class	Simulated		Observed		Catchment	Reference
	NO ₃ -N [kg ha ⁻¹ a ⁻¹]	NH ₄ -N [kg ha ⁻¹ a ⁻¹]	NO ₃ -N [kg ha ⁻¹ a ⁻¹]	NH ₄ -N [kg ha ⁻¹ a ⁻¹]		
ForMin	0.12–0.16	0.17–0.19	0.14	0.11	Vähä-Askanjoki	Kortelainen et al. (1997)
			0.12	0.09	Kuusivaaranpuro	
			0.27	0.11	Average of 9 catchments	
ForMinCu	0.23–0.30	0.18–0.19	0.12*	0*	Kangasvaara	unpublished
ForOrg	0.20–0.27	0.16–0.18	0.32	0.06	Kotioja	Kortelainen et al. (1997)
			0.33	0.19	Ylijoki	
			0.19	0.29	Average of 13 catchments	
ForOrgCu	0.50–0.56	0.39–0.40	0.21*	0.12*	Iso-Kauheaa	unpublished
Agriculture	5.88–6.42	0.82–0.89	8.8**		Hovi	Vuorenmaa et al. (2002)
			9.8**		Löytäneenoja	

Table 7. Simulated inorganic N leaching from terrestrial part (Catchment N model) in sub-catchments

Parameter	Dimension	Sub-catchment								
		64.01	64.02	64.03	64.04	64.05	64.06	64.07	64.08	64.09
NO ₃ -N	[kg N a ⁻¹]	18272	11595	20212	9932	12812	2834	5482	4667	3875
NH ₄ -N	[kg N a ⁻¹]	8787	7001	10534	7621	10549	2556	4472	3672	2721
NO ₃ -N	[kg N ha ⁻¹ a ⁻¹]	0.44	0.31	0.36	0.22	0.20	0.13	0.22	0.23	0.26
NH ₄ -N	[kg N ha ⁻¹ a ⁻¹]	0.21	0.19	0.19	0.17	0.17	0.17	0.18	0.18	0.19

3.2.2 Seasonality of inorganic N load (Papers III, IV, VI)

During the growing season simulated concentrations remained at higher levels compared to the observations but this did not have significant effect on daily inorganic N loads at the outlet of the river (Fig. 3) because the growing season is typically a low flow period. Loading from the river basin was concentrated to peaks during high flow periods, especially the snow melting period in spring. Inorganic N load to the sea was 41% higher in the wet year 1998 (annual discharge 62 m³ s⁻¹) and 29% lower in the dry year 1997 (annual discharge was 32 m³ s⁻¹) than the average load of the five-year period (VI).

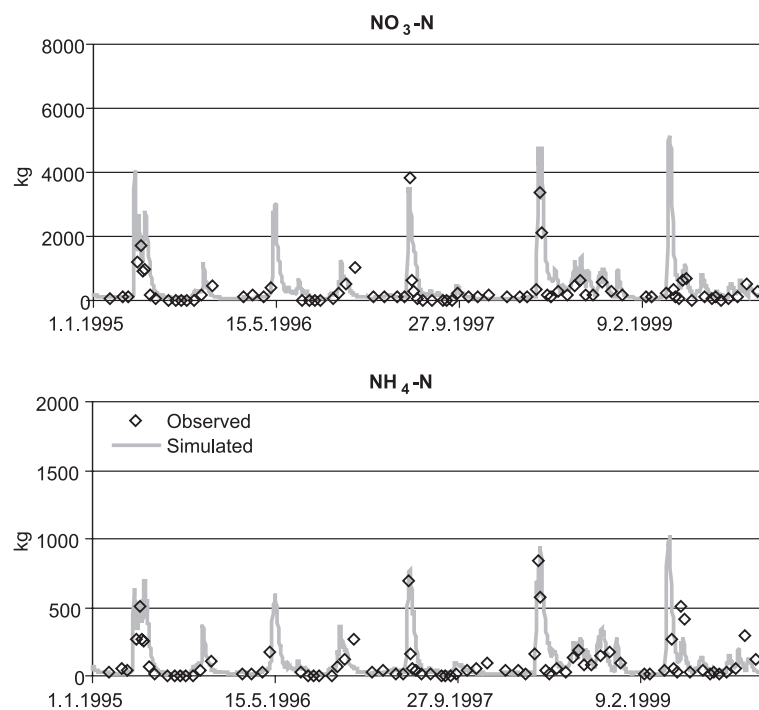


Figure 3. Simulated and estimated inorganic N load at the outlet of the river Simojoki

The beginning of the snow melt period is critical for N load (Fig. 4). On average 52% of the total load of inorganic N occurred in April-May. In April the relationship of inorganic N load to runoff was highest. The reason for this was not only increased runoff but also higher inorganic N concentrations in river water due to over-winter N mineralization processes in soil (IV).

Seasonal net N mineralization in the land use class 'Forest on mineral soil' is presented in Figure 5. Monthly mineralization rates were highest in early summer when the soil was warm but soil moisture did not limit process rates, declined later in the summer, but increased again in autumn. Mineralization rates were stable in winter reflecting stable soil temperature and moisture under the snow pack. Seasonal net mineralization in agricultural fields followed the pattern of that in forests, but the rates were higher (IV).

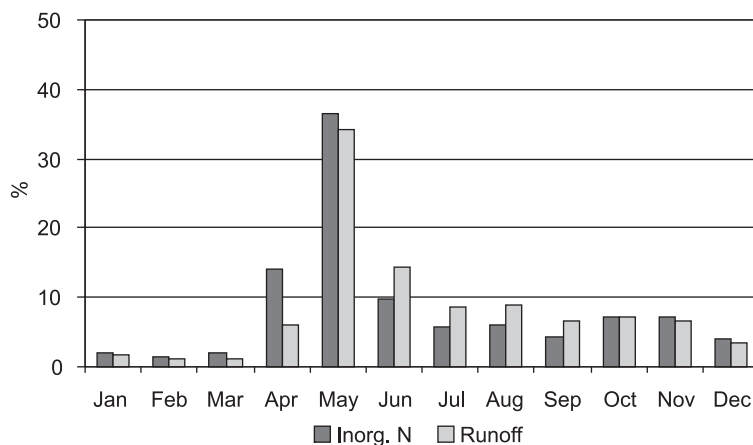


Figure 4. Seasonality of simulated runoff and inorganic N loading in the Simojoki river basin

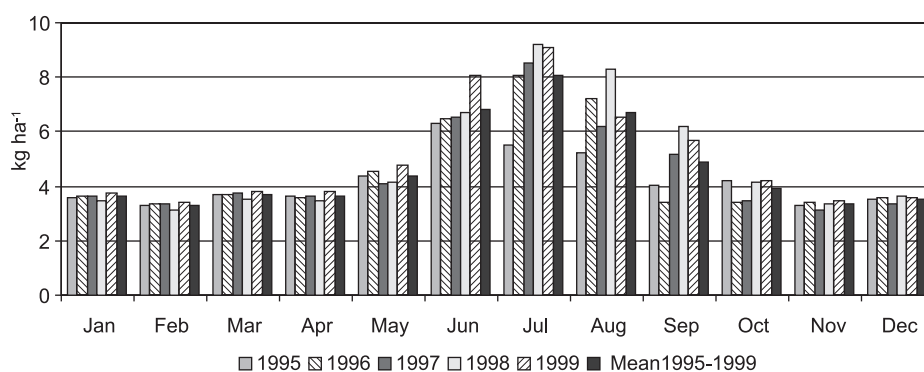


Figure 5. Simulated seasonal net N mineralization in the land use class Forest on mineral soil

Net N mineralization occurring during the dormant season when soil was assumed to be mainly frozen (November – April) accounted for 38% of the annual N mineralization. Division of the calendar year into two halves is a simplification of the actual growing seasons, which may vary considerably, both temporally from year to year and spatially within a large river basin.

The importance of over-winter microbial processes for N balance in boreal environments were supported by findings of Stottlemeyer and Toczydowski (1999b) that net N mineralization peaked in early summer, and 40% of the annual N mineralization occurred in winter. Further, Elberling and Brandt (2003) reported that winter soil respiration accounted for about 40% of the annual microbial soil respiration in tundra soils.

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. 1992, Stottlemeyer et al. 1995, Stottlemeyer and Toczydowski 1999b, Schimel et al. 2004). Gross mineralization and immobilisation of N have partly different regulating factors, so that gross and net mineralization rates are not necessarily correlated (Stottlemeyer and Toczydowski 1999a, Schimel et al. 2004). In this study net mineralization together with N retention on soil particles proved to be sufficient to simulate N processes (IV, VI).

Both peat mining and forest drainage can be assumed to increase inorganic N load during spring runoff as they were found to increase spring peak runoff (Seuna 1988, Seuna 1990). Concentration of $\text{NH}_4\text{-N}$ in runoff water from forest drainage areas was observed to be higher during the dormant season than during the growing season (Hynninen and Sepponen 1983, Laine et al. 1995). Almost one third of the annual loading from the Lumiaapa peat mining area which was monitored throughout the year since 1995, was observed to occur during a few weeks in spring (Kaikkonen and Salo 2004). In constructed effluent time series based on these references, inorganic N load from peat mining and forest drainage areas was highest in spring.

Runoff waters from large peat harvesting areas are conducted to one outlet where water quality and discharge are monitored. This justifies the treatment of peat harvesting areas as point sources, even though monitoring is conducted mainly in summer. The loading percentages of Lumiaapa peat mining area were generalised to other peat harvesting areas even though differences in water treatment methods etc. were not taken into account. When adding loading from forest drainage areas as percentual increase to loading from land use class 'Forest on organic soil' it is assumed that the seasonal dynamics was captured. However, more accurate method would be to simulate both discharge and N concentrations in a land use class in INCA-N.

3.2.3 Source apportionment of inorganic N load (Paper VI)

Simulated inorganic N gross load (total load from a land use class or a forest treatment area) in the river Simojoki was partitioned according to the source (Fig. 6). Load oriented approach was used in which losses in the river system were taken into account. In the upper parts of the river inorganic N load originated mainly from untreated commercial forests, but the load from other anthropogenic sources increased in the lower parts of the river (VI). This was in accordance with ecological

studies of the river Simojoki. According to both a bottom fauna survey (Liljaniemi 2003) and a diatom survey (Miettinen 2003) the ecological state of the river Simojoki was good. Upper parts of the river were almost in natural state, but the influence of anthropogenic sources could be seen in the lower reaches.

More than 50% of the inorganic N load at the outlet of the river originated from anthropogenic sources. Scattered settlements and agriculture were the greatest anthropogenic sources of inorganic N. Load from forest drainage areas was clearly higher than from forest felling areas. Peat mining had no significant role in inorganic N load to the sea.

As the INCA-N model simulates N processes in up to six land use classes it can be used to get an overview of the most important processes in catchment or river basin scale. Further, river basin scale model applications typically rely on satellite figures and statistics, which often can not distinguish different surface treatments in detail, or location of different treatments is not accurate. Other modelling methods are needed to study the effects of land use changes more closely. For example, to separate the effects of first-time drainage from the effects of supplementary drainage, or effects of forest harvesting from those of final cutting, not only a more detailed model is needed but also more detailed input data.

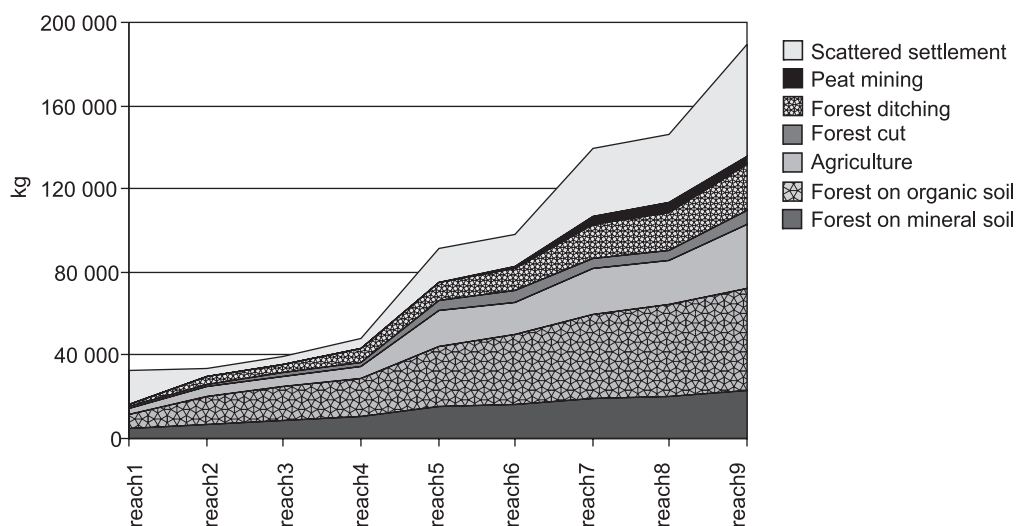


Figure 6. Source apportionment of simulated inorganic N gross load along the river Simojoki

3.3 Scenarios of environmental change

3.3.1 Changes in land use (Papers II, V, VI)

If the total area of agricultural fields were to stabilise to the extent of that in 2010 according to CAP scenarios, the inorganic N load would increase by 4%-5% from the simulated baseline level 190 000 kg a⁻¹. In a combined scenario of forest felling and MTR agricultural land use the inorganic N load would increase by 6%. By the year 2020 the increase in inorganic N load would cease because grass cultivation gave way to green fallow and set-aside land.

According to the CAP scenarios the area of agricultural land would increase from the 1995 level (VI). This development is well in line with the already observed 11% average increase in agricultural area in northern Finland in 1996-2003 (Information Centre of the Ministry of Agriculture and Forestry 2003). The assumption that in DREMFA simulations agricultural area was limited only by mineral soil types might even lead to underestimation of utilised agricultural area, as Mylly and Sinkkonen (2004) estimated the area of cultivated organic soils to be about 30% in northern Finland.

Agricultural activities in this northern river basin were clearly policy driven. Any significant reduction in milk price and de-coupling of agricultural support from production was likely to decrease the intensity and scale of production. This was contradictory to the results of Winter and Gaskell (1998) and Wier et al. (2002), who did not find any significant environmental effect of the Agenda 2000 CAP reform in Great Britain and Denmark. While the EU's agricultural policy has focused attention on controlling agricultural nutrient loading, it has also increased the risk of N loading through promoting changes in agricultural land-use and concentrating animal farming into larger units.

Increased forest felling by 20% (of annually felled area) would not change the inorganic N load to the sea. The area treated annually by forest fellings was small (~0.5% of the catchment area), and the locations of forest felling areas were scattered throughout the river basin. Furthermore, in the simulations ground vegetation was assumed to start to recover. Westling et al. (2001) estimated that if the treated area does not exceed 10% of the total area, water chemistry is mainly defined by the natural variation from untreated areas. The direct and local effects of forest treatments can be much higher (Ahtiainen and Huttunen 1999).

According to the latest studies the levels of 2-3 mg NO₃-N l⁻¹ are considered to endanger biodiversity (Giles 2005). Inorganic N concentrations in the river Simojoki stay lower than that in every land use change scenario and not even in peaks reach these values.

3.3.2 Effects of environmental protection measures on inorganic N load (Papers II, VI)

Buffer zones and wetlands established at agricultural fields would decrease inorganic N load to the sea by 2%. Forest drainage combined with buffer zones would decrease inorganic N load to the sea by 6%. Renewed subsurface disposal systems would decrease inorganic N load to the sea by 7%. Combination of the different measures would lead to reduction by 18% (VI). So, water protection measures properly

established in agriculture, forestry and scattered settlement area have the potential to better decrease the anthropogenic part of the inorganic N load to the Gulf of Bothnia than concentrating such measures to one land-use class only.

Decreasing atmospheric deposition according to the UNE scenario decreased N flux by <1% and according to the MFR scenario decreased N flux by 2% (II). Atmospheric N deposition in the area is relatively low and the small expected changes in N deposition did not have an important role in N leaching.

3.4 Uncertainty analysis

3.4.1 Equifinality of different parameter sets (Paper VII)

The results of the GLUE analysis can be presented as groups of dots where the x-axis shows the parameter value and the y-axis shows the goodness-of-fit measure of the output. Each dot represents one run of the model with parameter combinations selected from predefined distributions.

According to the GLUE analysis many parameter sets can perform equally well (problem of equifinality). Typically the parameters showed a pattern in which the same goodness-of-fit value was reached with several combinations of parameter values.

In simulations with hard data, discharge and inorganic N concentrations in river water at the outlet of the river only (Combination 1) the parameters defining mineralization rate did not find any optimal range (Fig. 7a,b,c). When the soft data, cumulative inorganic N leaching, plant N uptake and mineralization, was included an optimal range could be determined for the parameter defining mineralization rate in the land use class 'Arable land' (Fig. 7f). For mineralization rates in the land use classes 'Forest on mineral soil' and 'Forest on organic soil', acceptable parameter values smaller than $0.6 \text{ kg ha}^{-1} \text{ d}^{-1}$ gave higher likelihood values (Fig. 7d,e).

In the second combination rates of mineralization, nitrification, denitrification and N fixation together with temperature limits for these processes from the Catchment N submodel were selected for GLUE analysis. When hard data only was used in the analysis, positive values of the parameter defining minimum temperature for N mineralization gave lower likelihood values compared to negative parameter values (Fig. 8b), indicating that minimum temperature for mineralization should be negative. This pattern was not seen for nitrification or immobilisation (Figs. 8 a,c). When soft data was included, this pattern became more clear so that on the basis of GLUE analysis negative values for two other parameters – limiting temperatures for nitrification and immobilisation – appeared to give better results (Fig. 8d,f). These results support earlier modelling studies in which soil N mineralization proved to be the largest input of N to forested areas and over-winter N mineralization in sub-zero temperatures in soil proved to be significant when modelling N leaching in a northern area (II, IV).

When analyzing the river N model parameters (river flow velocity parameters *a* and *b*, and nitrification and denitrification rates) a clear optimum range of values was found for one parameter only. This parameter was the constant *a* value for reaches 5-9 (Fig 9a). The constant *a* in equation 6 used to calculate flow velocity is important as it influences the residence time in the river during low flow conditions and this is significant for nutrient processes. An optimum range was not found for the constant *b* or for nitrification and denitrification rates (Figs. 9b,c,d).

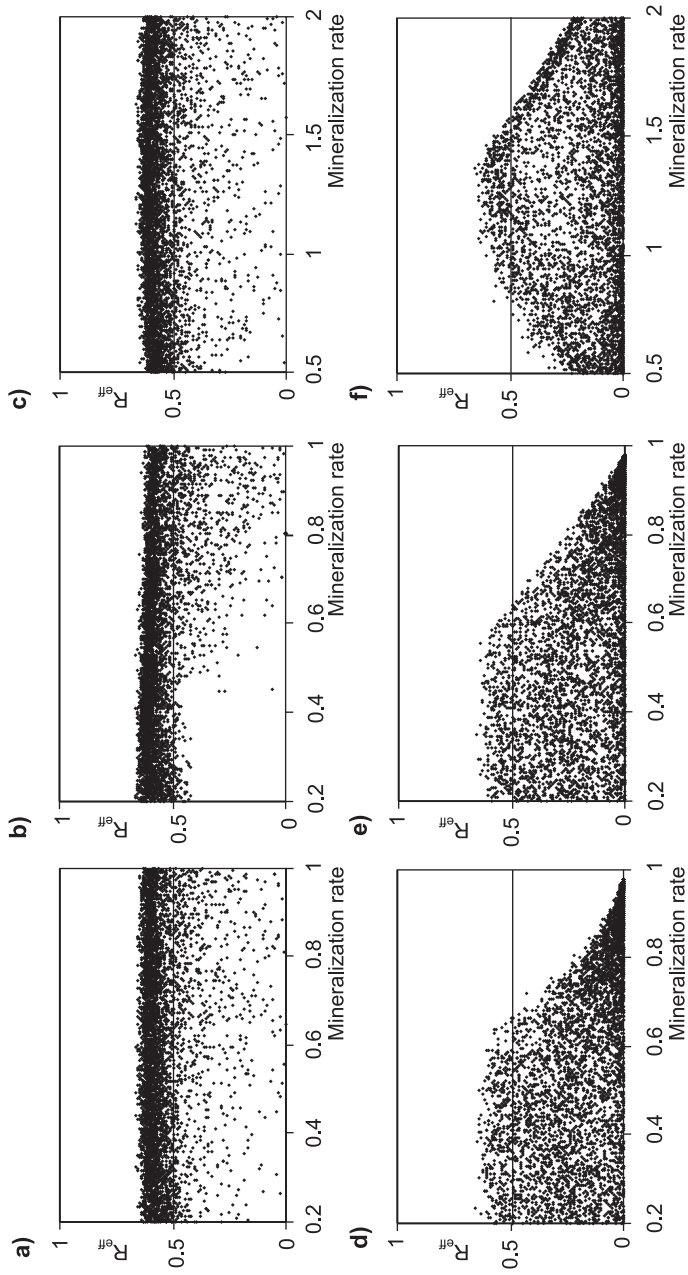


Figure 7. The catchment N submodel parameter values (combination 1) against likelihood measure.
a) Mineralization rate (kg ha⁻¹ day⁻¹), ForMin, hard data only b) Mineralization rate (kg ha⁻¹ day⁻¹), ForOrg, hard data only
c) Mineralization rate (kg ha⁻¹ day⁻¹), Agriculture, hard data only
d) Mineralization rate (kg ha⁻¹ day⁻¹), ForMin, soft data e) Mineralization rate (kg ha⁻¹ day⁻¹), ForOrg, soft data
f) Mineralization rate (kg ha⁻¹ day⁻¹), Agriculture, soft data

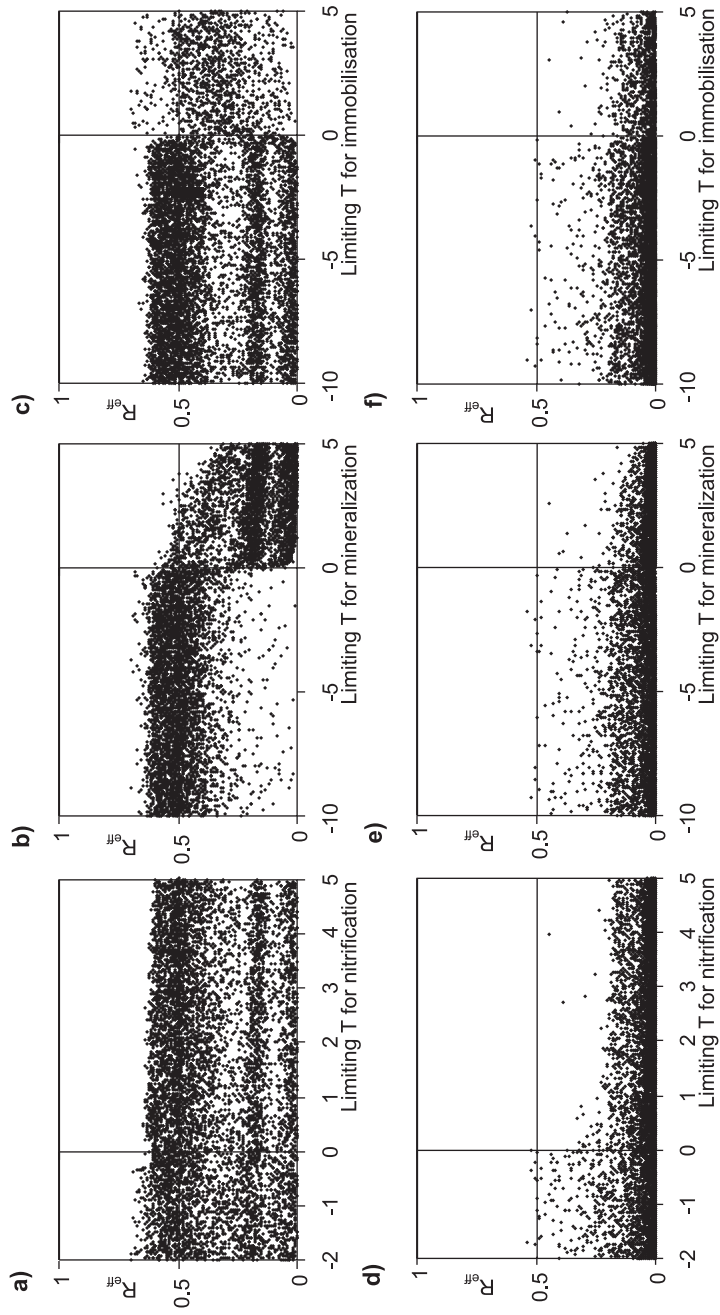


Figure 8. The catchment N submodel parameter values (combination 2) against likelihood measure.
 a) Limiting T for nitrification ($^{\circ}\text{C}$), hard data only b) Limiting T for mineralization ($^{\circ}\text{C}$), hard data only c) Limiting T for immobilisation ($^{\circ}\text{C}$), hard data only d) Limiting T for nitrification ($^{\circ}\text{C}$), soft data e) Limiting T for mineralization ($^{\circ}\text{C}$), soft data f) Limiting T for immobilisation ($^{\circ}\text{C}$), soft data.

The information contained in hard data (discharge and inorganic N concentrations in river water at the outlet of the river) was not sufficient for identification of parameter values by calibration. In this INCA-N model application to the Simojoki river basin the number of parameters needed was over 350, which led to overparameterization of the model when it was calibrated against data from one observation station only.

Another reason for the observed equifinality is that nitrogen-limited vegetation tends to take all the available nitrogen in northern areas. In the GLUE analysis it is possible to find parameter combinations in which catchment N processes compensate each other, i.e. leaching of N from terrestrial environment is about the same magnitude if mineralization is low and plant uptake is low, or if mineralization is high and plant uptake is also high.

Soft data proved to be one possible way to reduce the equifinality of different parameters. In this study cumulative inorganic N leaching, plant N uptake and mineralization proved to be useful soft data. Some more parameters might be identified if soft data of other catchment N processes, e.g. nitrification or N fixation, had been available.

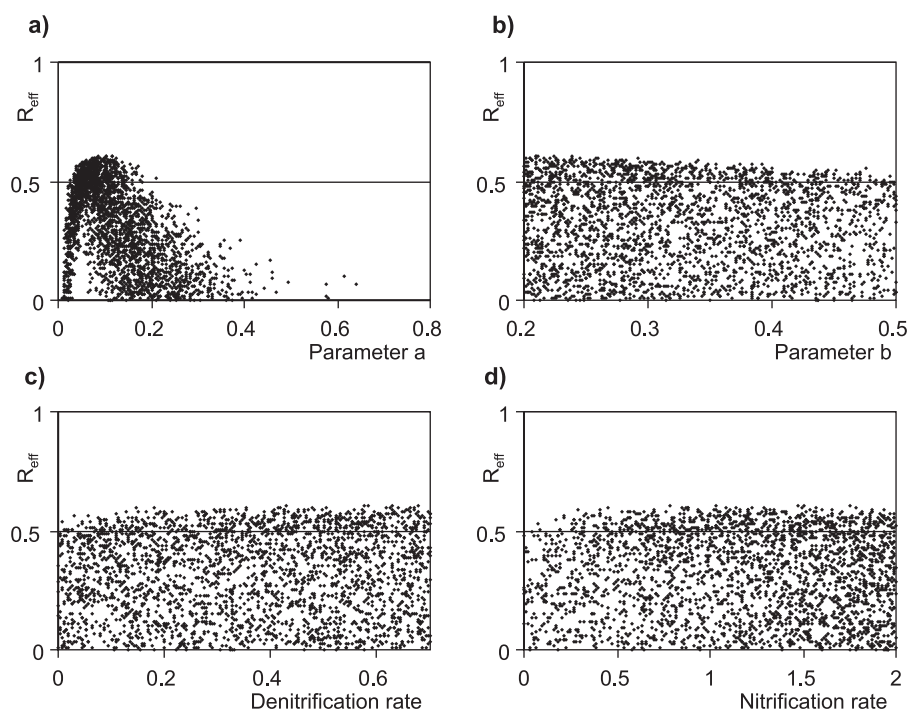


Figure 9. The river submodel parameter values (combination 3) against likelihood measure.

a) param a (m^{-2}) b) param b (-) c) denitrification rate (day^{-1}) d) nitrification rate (day^{-1})

3.4.2 Probability estimations of output range (Paper VII)

The probability estimation of 5% and 95% limits for simulated inorganic N concentrations in river water were calculated on the basis of the GLUE analysis. The phenomenon in which uncertainty limits fail to enclose the measured peak values can in general be seen as a sign that some processes are not adequately described in the model.

The catchment N submodel appeared to define $\text{NO}_3\text{-N}$ concentrations in river water in winter when soil N processes in sub-zero temperatures were allowed (Fig. 10). Some improvement was seen also in simulations of $\text{NH}_4\text{-N}$ observations. The highest $\text{NH}_4\text{-N}$ concentrations were individual peaks during low flow periods and their importance in the total load was not significant (VI).

During the growing season the observed inorganic N concentrations fell below 5% limit in simulations in which the first and second parameter sets of the Catchment N submodel were used. When the analysis was made using the third parameter set (river flow velocity and N processes in river water) most of the observations were above the 5% limit as this combination of parameters also allowed more variation in denitrification in the river during the growing season (Fig. 11).

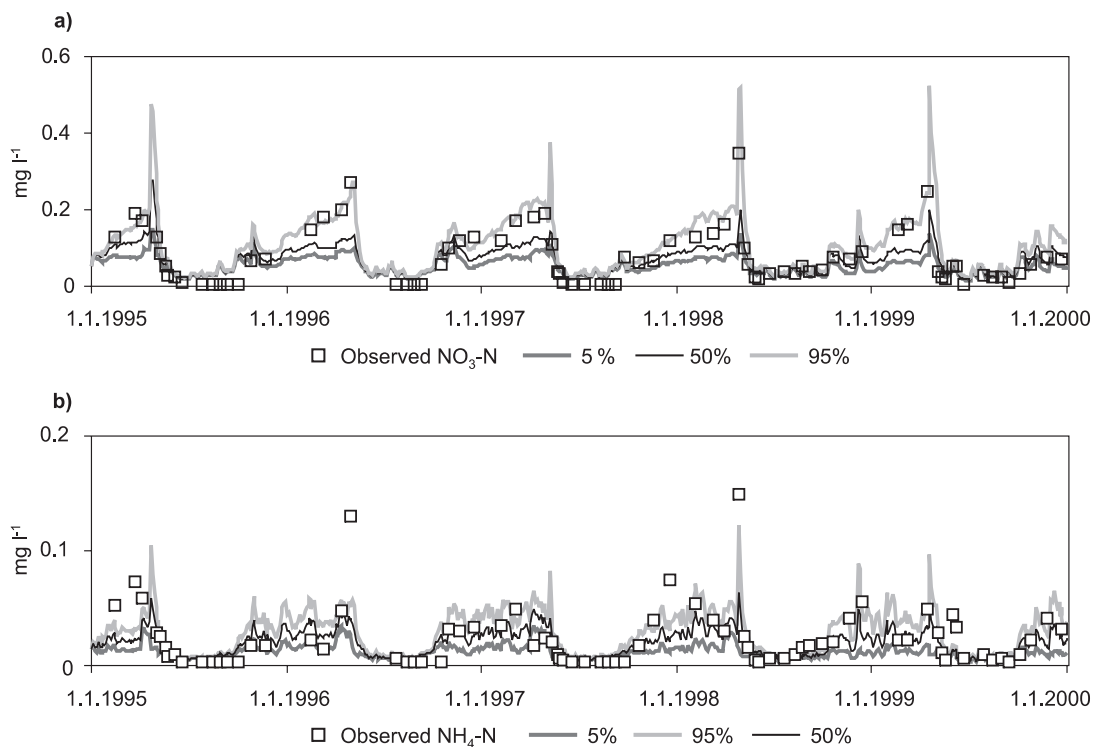


Figure 10. The 5% and 95% confidence limits for simulated inorganic N concentrations, a) $\text{NO}_3\text{-N}$ +b) $\text{NH}_4\text{-N}$. Catchment N submodel combination 2.

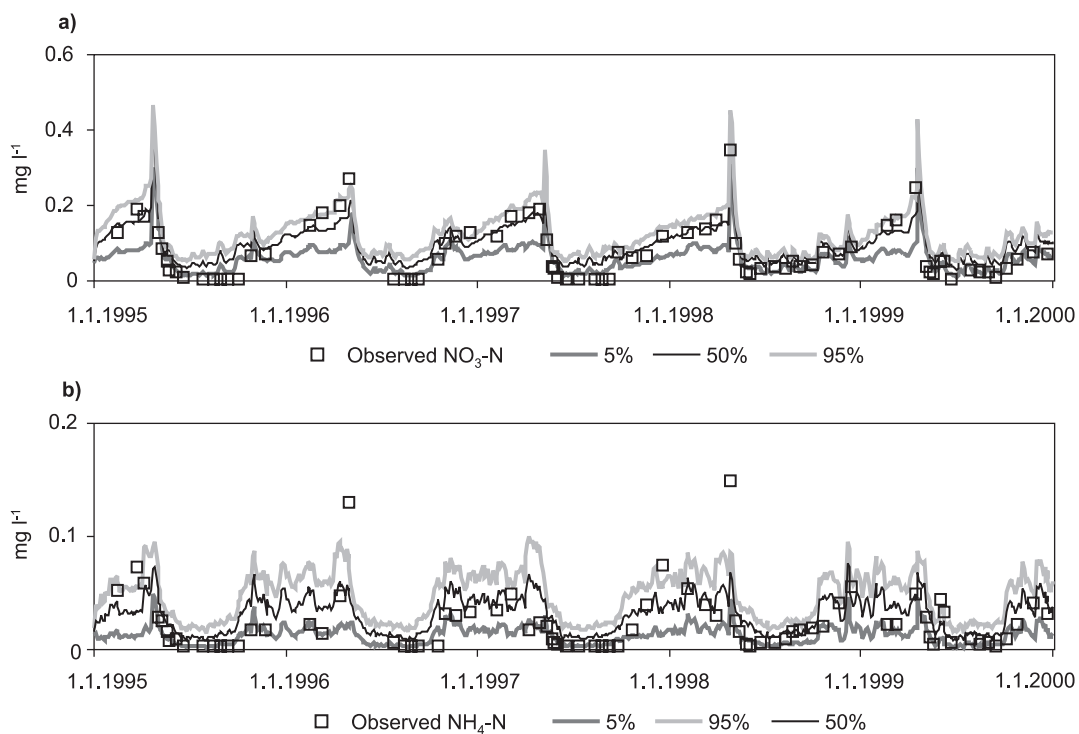


Figure 11. The 5% and 95% confidence limits for simulated inorganic N concentrations, a) $\text{NO}_3\text{-N}$ b) $\text{NH}_4\text{-N}$. River submodel combination 3.

The lowest concentrations during the growing season were not reproduced, which indicates that there are some retention processes either in peatland/wetland areas or in rivers which were not included in the INCA-N model. One such process could be the in-stream nitrogen uptake of aquatic biota during the growing season. Jarvie et al. (2002) concluded this missing process to be the reason for occasional overestimation of $\text{NO}_3\text{-N}$ concentration in the river Tweed in Scotland. In the lowest reach of the river Simojoki, which has a length 35900 m, the excess in simulated inorganic N compared to the observed value was around 200 kg N d^{-1} from 1 June to 15 September 1995. This could be accounted for by a plant N uptake of $100 - 300 \text{ kg N ha}^{-1}$ growing season $^{-1}$ assuming a surface area of 70 - 200 ha. As the river is not totally covered by aquatic plants, this process alone is not sufficient to explain low summertime concentrations.

One reason for low summer-time concentrations is that in Finnish forested areas, runoff draining from mineral soils flows to rivers via peatland/wetland areas which retain nutrients effectively during the growing season, whereas during winter and snow melt periods most of the runoff flows over the frozen land. Moreover, Deelstra et al. (2004) measured agricultural N losses at three spatial scales, i.e. plot, drainage field and catchment scale. Agricultural N losses measured on a catchment scale were 28% lower than the measured losses on the drainage field scale. They assumed that this was due to different hydrological pathways and retention processes, such as denitrification in open streams.

4 CONCLUDING REMARKS

The emphasis in implementing water protection methods in the Simojoki river basin is in improving the state of the Gulf of Bothnia in the Baltic Sea. Nitrogen from clearly P limited Bothnian Bay flows south to N limited Bothnian Sea where eutrophication has been observed to continue throughout 1980s and 1990s (Pitkänen 2004). No decreasing trend were observed in N concentrations in the river water from 1975 to 2000 (Räike et al. 2003) although improved water protection methods were applied in peat mining and agriculture during the last decade.

The general objective of this work was to outline the timing and origin of inorganic N loading from a boreal river basin in northern Finland by the dynamic, semi-distributed INCA-N (Integrated Nutrients in Catchments – Nitrogen) model. An interesting question is the role of terrestrial organic N in total N load to the sea because organic N concentration in the river Simojoki is higher than inorganic N concentrations. According to latest studies chromophoric dissolved organic matter decomposes in coastal waters mainly due to photodegradation relatively rapidly (Vähätalo and Wetzel 2004). In this way organic N may become available to food webs in N limited sea areas. To get a general view of N loading more emphasis should be paid on organic N leaching.

The INCA-N model was validated by the traditional split-sample method using independent data from years which were not used in calibration as the more rigid validation scheme turned out difficult to implement. Instead simulated annual inorganic N process fluxes were compared to values found in small catchment studies and literature. In the uncertainty analysis GLUE (Beven and Binley 1992) the information contained in hard data (discharge and inorganic N concentrations in river water at the outlet of the river) was not sufficient for identification of parameter values by calibration. Soft data (in this case experimental knowledge of N processes) was used to evaluate aspects of the model simulations for which no hard data was available. The use of experimental knowledge of N processes helped to identify parameter values and reduce equifinality. Thus the use of soft data is recommended during calibration of the INCA-N model.

Inorganic N load to the sea was mostly dependent on annual hydrological and meteorological conditions. The discharge in the Simojoki river basin was dominated by a snow-melting peak in spring and inorganic N loading was concentrated to this peak. On average 50% of the total load of inorganic N occurred in April-May. Due to higher N concentration in the river water derived from over-winter N mineralization, inorganic N load in the beginning of the snowmelting period was higher than expected based on contemporary runoff level. During the growing season inorganic N concentrations decreased almost to the detection limit, presumably due to different retention processes in catchment.

This modelling work suggests that over-winter N processes in soil play important role in N leaching from boreal forests. So far very little empirical information about N process rates in Finnish climate and land use types in winter is available. Due to global climate change winter time runoff in Finland is assumed to increase (Carter and Kankaanpää 2005). This may lead to increased inorganic N leaching from this kind of basin where N loading is concentrated on spring flow peaks and mainly controlled by annual hydrology. Further, to be able to describe nutrient leaching from

river basins in full, more information about different retention processes in aquatic and terrestrial environments is needed. For example Deelstra et al. (2004) measured clearly lower agricultural N losses on catchment than on drainage field scale.

Agricultural activities in the Simojoki river basin were clearly policy driven. While the EU's agricultural policy has focused attention on controlling agricultural nutrient loading, it has also increased the risk of N loading through promoting changes in agricultural land-use and concentrating animal farming into larger units. Expected changes in forestry land use would not increase inorganic N loading to the Gulf of Bothnia, and water protection measures related to forest drainage areas could actually decrease it.

The water protection scenarios in this study should be considered indicative, because they are to a large degree based on expert opinions. More empirical research of the effect of water protection methods and different treatments in forestry and agriculture is needed. Nevertheless, water protection measures properly established in agriculture, forestry and scattered settlement area have the potential to better decrease the anthropogenic part of the inorganic N load to the Gulf of Bothnia than concentrating such measures to one land-use class only.

5 SUMMARY

In this study the dynamic, semi-distributed INCA-N model was applied to a boreal river basin in northern Finland in order to outline inorganic N leaching patterns and N processes in catchment scale. The discharge in the study area is dominated by a snow-melting peak in spring. Inorganic N concentrations in the river water increase throughout the winter and decrease almost to the detection limit during the growing season. The Simojoki river discharges to the Gulf of Bothnia in the Baltic Sea. Nitrogen from clearly P limited Bothnian Bay flows to south to N limited Bothnian Sea where eutrophication has been observed to continue.

In this work some changes were suggested to the structure of the INCA-N model. The structure of the model was changed so that hydrological input can be given to each of the sub-catchments instead of the whole river basin. This improved simulation of discharge and water balance clearly.

A new simple method to calculate soil temperature was developed that can be used to calculate daily values for soil temperature at various depths in both frozen and unfrozen soils. The model requires four parameters: average soil thermal conductivity, specific heat capacity of soil, specific heat capacity due to freezing and thawing and an empirical snow parameter. Precipitation, air temperature and snow depth (measured or calculated) are needed as input data. This simple model simulates soil temperature well in the uppermost soil layers where most of the nitrogen processes occur. The small number of parameters required means that the model is suitable for addition to catchment scale models.

Further, a method to include experimental knowledge, 'soft data' to GLUE uncertainty analysis was introduced. The Generalized Likelihood Uncertainty Estimation (GLUE) approach defines the performance of possible parameter sets in terms of likelihood measures. Soft data (in this case experimental knowledge of N processes) was used to evaluate aspects of the model simulations for which no hard data (discharge and inorganic N concentrations in river water at the outlet of the river) was available. These values were taken from field studies conducted in corresponding climate, land use and geological conditions.

Based on this work the following conclusions can be made:

1. Is it possible to follow rigorous calibration and validation scheme in river basin scale model set-up?

The INCA-N model was calibrated successfully to the Simojoki river basin. The fit between simulated and observed discharge was good (R_{eff} of 0.76) and the model represented the seasonal dynamics in $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations (R_{eff} of 0.61 and 0.30, respectively). The fit between simulated and observed inorganic N concentrations during the dormant season was good, but during the growing season observed inorganic N concentrations in the river water decreased down to the detection limit whereas simulated concentrations remained at a higher level (VI).

The model was validated by the traditional split-sample method using independent data from years which were not used in calibration (II). The model behaviour was not tested through the required transition regime neither was multi-site calibration to individual sub-catchments used, because monitoring data from

representative areas was not available. Instead simulated annual inorganic N process fluxes were compared to values found in small catchment studies or literature.

Simulated annual inorganic N process fluxes were in the range reported in the literature. Leaching losses from commercial forests, in which the most recent forestry treatments were made more than 10 years ago were less than one kg N ha⁻¹, because the vegetation tended to take up all the available N. Inorganic N leaching from new (less than 10 years old) forest treatment areas was higher, and leaching of N from agricultural areas was clearly higher than from forested areas.

2. Is it possible to enhance reliability of model simulations by correct use of all available data?

The Generalised Likelihood Uncertainty Estimation (GLUE) approach defines the performance of possible parameter sets in terms of likelihood measures. In the GLUE analysis it was possible to find parameter combinations which can perform equally well (problem of equifinality). The information contained in hard data (discharge and inorganic N concentrations in river water at the outlet of the river) was not sufficient for identification of parameter values by calibration but led to overparameterization of the model. Further, it is possible to find parameter combinations in which catchment N processes compensate each other, i.e. leaching of N from the terrestrial environment is about the same magnitude if mineralization is low and plant uptake is low, or if mineralization is high and plant uptake is also high. The use of experimental knowledge of N processes (soft data) is recommended during calibration of the INCA-N model as this helps to identify parameter values and thus reduce equifinality. In this study cumulative inorganic N leaching, plant N uptake and mineralization proved to be useful soft data (VII).

In the GLUE analysis most of the observations fell between simulated 5% and 95% confidence limits indicating that the model structure was able to describe the most influential processes in the river basin.

3. Does the INCA-N model explain the main features of hydrological pattern and seasonality of inorganic N concentrations in river water in the boreal zone?

In the INCA-N model the hydrological inputs relate to the whole river basin or the sub-basins rather than to the individual land use classes. Hydrological data given separately to each of the subcatchments improved both simulated discharge and annual water balances compared to the lumped input data which was the same for the whole river basin (I). Loading from land use changes which have long-lasting effects on runoff, such as forest drainage in paper VI, can be added as an effluent time series to the model if sufficient observation data is available.

The INCA-N model was able to simulate the increasing inorganic N concentrations in the river water in winter when N processes in sub-zero temperatures together with a physically based method to calculate soil temperature were included in the model (III, IV). In the GLUE analysis the catchment N submodel appeared to explain inorganic N concentrations in river water in winter when soil N processes in sub-zero temperatures were allowed (VII).

When the GLUE analysis was performed by allowing river flow velocity and N processes in river water to change most of the low concentrations in summer were explained (VII). The lowest concentrations during the growing season were not reproduced, which indicates that there are some retention processes either in peatland/wetland areas or in the river which were not included in the INCA-N model. Simulated higher concentration levels compared to observations during the growing season had no significant effect on daily or annual inorganic N loads at the outlet of the river because the growing season is typically a low flow period. The INCA-N model can be applied to a northern river basin in the boreal zone and the results are reliable as long as the interpretation is based on daily or annual loads (VI, VII).

In the upper parts of the river inorganic N load originated mainly from commercial forests, but the load from other anthropogenic sources increased in the lower parts of the river. The proportion of this load at the outlet of the river was more than half, so that agriculture, forestry and scattered settlement had almost equal amounts of the total load (VI).

Inorganic N load to the sea was mostly dependent on annual hydrological and meteorological conditions, so that the load was 41% higher in the wet year 1998 (annual discharge $62 \text{ m}^3 \text{ s}^{-1}$) and 29% lower in the dry year 1997 (annual discharge $32 \text{ m}^3 \text{ s}^{-1}$) than the average load of the studied five year period when the average discharge was $40 \text{ m}^3 \text{ s}^{-1}$ (VI). Loading from the river basin was concentrated to peaks during high flow periods, especially the snow melting period in spring (II, VI). On average 50% of the total load of inorganic N occurred in April-May. Both peat mining and forest drainage were found to increase spring peak runoff and inorganic N loading (VI).

4. Does over-winter N mineralization explain the relatively high inorganic N concentrations in late winter/early spring?

The main N input in forested areas was from soil N mineralization, and simulated annual net N mineralization was clearly one order of magnitude higher than atmospheric inorganic N deposition (II). Over-winter nitrogen mineralization processes in soil accounted for about 38% of annual N mineralization (IV). Soil temperature under the snowpack in winter remains mainly above $-5 \text{ }^\circ\text{C}$ which is high enough for soil decomposition processes to continue. In this work a new function in which soil freezing and thawing processes were included was developed to calculate soil temperature under the snow cover (III). Due to the higher N concentration derived from over-winter N mineralization (IV), inorganic N load in the beginning of the snow melting period was higher than expected based on contemporary runoff level (VI).

5. Is it possible to decrease diffuse pollution by allocating suitable water protection measures to all possible sources?

The water protection measures (buffer zones and wetlands established at agricultural fields, forest drainage combined with buffer zones, renewed subsurface disposal systems) would decrease inorganic N load to the sea up to 7% when acting individually. Combination of the water protection measures both in agricultural and forestry areas and in scattered settlement areas would decrease

the load of inorganic N more effectively (about 18% of the total load) than when concentrating on one source only (VI).

Change in N deposition according to current legislation (UNE scenario) did not play an important role in N load (II). Atmospheric N deposition in the area is relatively low and expected changes in N deposition, which are small, did not have an important role in N leaching.

6. How will the expected land use changes in future change total inorganic N load in the Simojoki river basin?

Increased forest felling by 20% of forestry land would not change the inorganic N load to the sea, but increase in agricultural land by 25%-33% (of agricultural land) may lead to increased N load to the sea by 2010 (VI). Agricultural activities were clearly policy driven and agricultural area tends to increase due to the EU's CAP policy. If the total area of agricultural fields would stabilise to that in 2010 according to CAP scenarios, the inorganic N load would increase by 4%-5%. By the year 2020 the increase in inorganic N load would cease because grass cultivation gave way to green fallow and set-aside land (MTR scenario) or the area of agricultural land decreased (BASE scenario).

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APPENDIX 1: LIST OF SYMBOLS

a	constant to calculate flow velocity [m^{-2}]
a_1	lower limit for the range in which the simulated value is partly acceptable in fuzzy rule
a_2	lower limit for the range in which the simulated value is fully acceptable in fuzzy rule
a_3	upper limit for the range in which the simulated value is fully acceptable in fuzzy rule
a_4	upper limit for the range in which the simulated value is partly acceptable in fuzzy rule
a_5	in-stream ammonium concentration [mg l^{-1}]
a_6	in-stream nitrate concentration [mg l^{-1}]
b	constant to calculate flow velocity [-]
BFI	base flow index [-]
C	scaling factor in GLUE
C_A	apparent heat capacity term [$\text{J m}^{-3} \text{ }^\circ\text{C}^{-1}$]
C_i	simulated values
C_1	denitrification rate in soil [day^{-1}]
C_2	fixation rate in soil [$\text{kg N ha}^{-1} \text{ day}^{-1}$]
C_3	plant NO_3 uptake rate in soil [day^{-1}]
C_4	nitrification rate in soil [day^{-1}]
C_5	mineralization rate [$\text{kg N ha}^{-1} \text{ day}^{-1}$]
C_6	immobilisation rate [day^{-1}]
C_7	plant NH_4 uptake rate [day^{-1}]
C_{10}	in-stream nitrification rate [day^{-1}]
C_{11}	in-stream denitrification rate [day^{-1}]
D_S	snow depth [m]
f_S	empirical damping parameter [m^{-1}]
HER	hydrologically effective rainfall [mm]
I	inflow to reach [$\text{m}^3 \text{ s}^{-1}$]
K_T	soil thermal conductivity [$\text{W m}^{-1} \text{ }^\circ\text{C}^{-1}$]
L	reach length [m]
$L_0(\theta)$	a prior likelihood of the parameter set θ in GLUE
$L(\theta y)$	a posterior likelihood of the parameter set θ in GLUE
$L_1(y \theta)$	likelihood measure calculated for the simulation of the observed variable y by the parameter set θ in GLUE
M_i	measured values
P	actual precipitation [mm]
Q	outflow from reach [$\text{m}^3 \text{ s}^{-1}$]
R_{eff}	the Nash and Sutcliffe coefficient of efficiency
R_{soft}	degree of acceptance from the corresponding simulated quantity or parameter value
R_{tot}	total efficiency
S	storage in reach [$\text{m}^3 \text{ s}^{-1}$]
S_2	seasonal plant growth index [-]
S_3	NO_3 input mass from upstream [kg N]

S_6	NH_4 input mass from upstream [kg N]
SMD	daily soil moisture deficit [mm]
SMD_{max}	maximum soil moisture deficit [mm]
t_{Q10}	factor change in rate with a 10 °C change in temperature [-]
t_{Q10bas}	base temperature for N processes at which the response is 1 [°C]
T	travel time parameter [s]
T_{AIR}	air temperature [°C]
T_Z	soil temperature [°C]
T_1	time constant associated with the soil water zone [day]
T_2	time constant associated with the groundwater zone [day]
T_Z^{t+1}	soil temperature at depth Z_s at time $t+1$ [°C]
T_Z^t	calculated soil temperature from the previous day [°C]
T_*^{t+1}	uncorrected soil temperature [°C]
U_2	input rate of NO_3 load to soil store [$\text{kg N ha}^{-1} \text{ day}^{-1}$]
U_3	input rate of NH_4 load to soil store [$\text{kg N ha}^{-1} \text{ day}^{-1}$]
v	mean flow velocity in the reach [m s^{-1}]
V_r	retention volume in the soil [$\text{m}^3 \text{ km}^{-2}$]
x	simulated value in fuzzy rule
x_1	output flow of water of the soil zone [$\text{m}^3 \text{ s}^{-1}$]
x_2	output flow of water of the groundwater zone [$\text{m}^3 \text{ s}^{-1}$]
x_3	NO_3 mass in soil stores [kg N km^{-2}]
x_4	NO_3 mass in groundwater stores [kg N km^{-2}]
x_5	NH_4 mass in the soil store [kg N km^{-2}]
x_6	NH_4 mass in groundwater store [kg N km^{-2}]
x_{11}	drainage volume in the soil [$\text{m}^3 \text{ km}^{-2}$]
x_{12}	groundwater drainage volume [$\text{m}^3 \text{ km}^{-2}$]
x_{22}	reach flow [$\text{m}^3 \text{ s}^{-1}$]
x_{23}	NO_3 mass stored in the reach [kg N]
x_{24}	NH_4 mass stored in the reach [kg N]
x_{29}	reach volume [m^3]
y	new observation in GLUE
Z_s	soil depth [m]
Θ	parameter set in GLUE