HUMAN IMPACTS AND WEATHER-DEPENDENT EFFECTS ON WATER BALANCE AND WATER QUALITY IN SOME SWEDISH RIVER BASINS

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BY
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DISSERTATION

Doctoral dissertation to be publicly defended in the Committee-room of the Royal Institute of Technology (Kollegiesalen, KTH), Valhallavägen 79, Stockholm, on December 7th, 1990, at 10:00 a.m.

Abstract

The weather has a great effect on the water balance and, indirectly, affects water quality of river systems. At the same time, man-made changes in the landscape and other human activities have a great impact. To be able to distinguish the human impacts from the effects of natural weather fluctuations we need observations and measurements but also analysis tools.

In this thesis the PULSE and HBV hydrological models have been used as the analysis tools. Examples are given from forest management, in particular clear-cutting, drainage and biomass increase, and from mining and agricultural activities. The models include conceptual descriptions of the most significant hydrological processes and are capable of coping with weather-dependent fluctuations. Observed air temperature, precipitation and an estimate of the potential evapotranspiration are input data to the models.

Simple hydrochemical and nitrogen leaching subroutines have been linked to the PULSE water balance model. These subroutines have been used to quantify weather-dependent and human effects on pH downstream from a mine tailings deposit and on nitrogen leaching from different non-point sources, especially from arable land.

The applications illustrate the advantage of this type of model for analysis of man-made impacts and short-term climatological fluctuations. As the models are restricted to stationary conditions they cannot be used for forecasting of long-term changes due to changes in atmospheric deposition, land use or climate, unless the local effects of these changes are known.

Other methods of analysing effects of man-made changes have also been tested, such as conventional comparative investigations, regression analysis and trend analysis. The use of these methods is exemplified by an analysis of human effects on erosion and sediment transport. It was found to be much more difficult to quantify effects with these simpler methods.

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Key words

Water balance modelling, nitrogen model, human effects on runoff, water quality.

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C O N T E N T S

PREFACE

The thesis is mainly based on the following papers dealing with four environmental issues in connection with the hydrology of the rural landscape.

<u>Effects of forest management:</u>

I.   Brandt, M., Bergström, S., and Gardelin, M., 1988. Modelling the Effects of Clearcutting on Runoff - Examples from Central Sweden. Ambio, Vol. 17, No. 5: 307 - 313.

This paper is a result of team-work between the authors, of whom Sten Bergström is responsible for the initial ideas. Marie Gardelin made the simulation of pre-harvested and post-harvested water balance for one of the clearcutted small basins. I am responsible for the remaining simulations of clearcutted basins, the model application to a larger basin, and the analysis of the results.

<u>Effects of mine tailing deposits:</u>

II a. Allard, B., Karlsson, S., Lohm, U., Sandén, P., Bergström, S., and Brandt, M., 1987. Environmental Impacts of an Old Mine Tailings Deposit - Hydrochemical and Hydrological Background. Nordic Hydr. 18: 279 - 290.

II b. Brandt, M., Bergström, S., and Sandén, P., 1987. Environmental Impacts of an Old Mine Tailings Deposit - Modelling of Water Balance, Alkalinity and pH. Nordic Hydr. 18: 291 - 300.

Paper II a serves as an introduction to three papers based on field data from the Bersbo mine area. Paper II b is one of these papers. In paper II a I contributed with the hydrological description. In paper II b Sten Bergström is responsible for the original ideas concerning modelling. Per Sandén has (together with Stefan Karlsson) contributed with data collection and the analysis programme. I am responsible for model development, the simulations of water balance and hydrochemistry, sensitivity analysis, and conclusions.

<u>Suspended and dissolved material:</u>

III.   Brandt, M., 1990. Generation, Transport and Deposition of Suspended and Dissolved Material - Examples from Swedish Rivers. Geogr. Ann. 72A (3-4), (in press).

<u>Non-point source pollution from agricultural land:</u>

IV.   Bergström, S., Brandt, M., and Gustafson, A., 1987. Simulation of runoff and nitrogen leaching from two fields in southern Sweden. Hydr. Sciences Journal 32: 191 - 205.

The nitrogen model described in paper IV is my first real effort to construct a conceptual model. Sten Bergström initiated the project, taught me how to develop the model structure, and followed up the results. Arne Gustafson has

contributed with the data base and the knowledge of nitrogen turnover, which I tried to describe empirically in the model.

V.   Brandt, M., 1990. Simulation of Runoff and Nitrate Transport from Mixed Basins in Sweden. Nordic Hydr., 21: 13 - 34.

VI.   Brandt, M., 1990. Modelling Nitrogen Transport from a Basin of Mixed Land Use - Example from the Lake Ringsjön drainage area, Sweden. Vannet i Norden, Vol. 23, No. 3: 39 - 57.

In the dissertation summary, reference to these papers will be given by their Roman numerals. Some results not included in these papers will also be presented.

ABSTRACT

The weather has a great effect on the water balance and, indirectly, affects water quality of river systems. At the same time, man-made changes in the landscape and other human activities have a great impact. To be able to distinguish the human impacts from the effects of the natural weather fluctuations we need observations and measurements but also analysis tools.

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DISSERTATION SUMMARY

INTRODUCTION

The weather affects the water balance and the transport of different water-carried and dissolved substances both directly and indirectly. Man has also a great influence through changes of land use, regulation of water systems, pollution and other activities. Today these impacts are accelerating in the world due to increasing population and urbanization, industrial development, increasing energy need, and more effective management of agriculture and forestry. Significant changes of land use have occurred over very long time periods, and it can be argued that very few areas can really be termed natural.

To be able to interpret short-term variations and discern any change caused by human influence, we must always take weather fluctuations into account. Sometimes the effect of human influence or weather fluctuations is so large that it is easy to see the difference, but often there is a complex interaction. It is important that we learn to distinguish man-made changes from natural effects, otherwise the former can grow to such extent that major damage will occur. This is also important for the evaluation of the effects of pollution control programmes.

Climate is not constant in the long run. We have had drastic fluctuations in the past and are now discussing a possible global climate change due to growing atmospheric concentration of carbon dioxide and other greenhouse gases. A climate change will affect the way man can use land, which may indirectly have climatic repercussions. This aspect is, however, out of the scope of this thesis.

OBJECTIVES OF THE STUDY

The overall objectives of the research presented in the papers (I - VI) can be summarized as follows:

- to find methods of distinguishing effects of human influence from natural fluctuations (mostly weather-dependent) on water balance, water quality and transport of different water-carried and dissolved substances.

- to develop an operational nitrogen leaching model, applicable for small fields up to larger basins with mixed land use and lakes.

- to give examples of natural weather-dependent and human influences on water balance and transport of different water-carried substances in Swedish river basins.

It would be too far-reaching to try to cover all conceivable human impacts on the water cycle. The emphasis is put on the rural system and some issues which have

been very much in focus in the hydrological debate in Sweden during the 1980s. Effects of urbanization are not included.

The locations of the river basins studied in papers I - VI and in this summary are found in Figure 1 except the stations of the nationwide sediment transport network (see Figure 3 in paper III).

Figure 1. Locations of the principal basins and rivers studied in the research projects discussed in papers I - VI and in this summary. The Roman numerals of the papers are also indicated, where they occur.

METHODS OF DISTINGUISHING HUMAN INFLUENCE FROM NATURAL WEATHER-DEPENDENT FLUCTUATIONS

Comparative investigations

One common way of studying effects of human impacts on the water cycle is to make comparative investigations in different drainage basins. The basins can differ in, for example, land use, river regulation, mining, and wastewater pollutions. The same method can be used to study natural influences on water balance and water quality, such as effects of different morphology, geology and lake percentage of the drainage areas. In paper III this methodology is used to estimate effects of land use and other human impacts on erosion and sediment transport. The drawback of this method is that the results are an integration of several effects. To improve the analysis, relationships between runoff or water quality and basin characteristics are sought. This can be done by dividing the basins into geographical regions, homogeneous groups based on runoff or water quality characteristics, or groups based on basin characteristics. The Frend project (Gustard et al. 1989) has demonstrated the use of regionalization and multiple-regression analysis. I have used this method for regionalization of concentration and transport of suspended and dissolved material in Swedish river systems, but there is a scaling problem. I have also tried to find relationships between sediment transport and basin properties with multiple-regression. The problem was to identify and to choose convenient independent basin characteristics, and the attempt actually failed (Brandt 1982).

Pair-basin approach

Another way to improve the information from comparative investigations is to use a pair-basin approach with one control and one treated basin with a pre-treatment period of some years. This method is often used, for example, to study effects of clearcutting and drainage on runoff. The pre-treatment period is used to find a regression equation between the variables that are of interest, for example runoff in the two basins, and to analyse how this is changed after treatment in one of the basins. Instead of one treated and one control basin, streamflow from several partial treated basins can also be compared to an untreated basin. The first clearcutting study in Sweden used the pair-basin method. It was performed in the 1920s but published much later (Kihlberg 1958). The same method was used later by Grip (1982), Rosén (1984), and in Finland by Seuna (1988) to determine changes in runoff after clearcutting.

Double-mass curves can be an alternative to regression when analysing the effect of a treatment in a pair-basin approach (Liebscher 1980). A treatment effect is then indicated by a break in the trend line.

The pair-basin approach is often an efficient way of detecting, for example, change in water yield after a treatment. The weaknesses of the method are high costs, sometimes difficulties in generalizing the results, and inability to provide the reasons for the changes (Seuna 1989). Many of the analyses are based on partly treated basins, and the relationship found is often too vague for a firm conclusion.

I have noticed two specific problems when working with pair-basin data sets and regression methods. The first one is the reliability of the control basin data. The record must be homogeneous for the whole period. In the first Swedish study of clearcutting at Himmelsberget (Kihlberg 1958), for example, there was a problem with increasing water leaching under the dam in the control basin. This was documented in notes from the investigation period, and the project seems to have been stopped partly due to this. There was also a problem with icedamming during snowmelt periods. Inhomogeneities in the control data set may result in misleading conclusions from the whole study.

Another problem is that the weather fluctuations between years can disturb the results quite substantially. In the clearcutting study at Kullarna (Rosén 1984) the pre-treatment period was dry and followed by some wet years after treatment, which implied that the regression relation had to be extended outside the range of observations and was thus uncertain. The length of observation period needed is therefore dependent on the variability of the climate. The larger the fluctuations, the longer the observation period that must be used. Pre-treatment and post-treatment periods of at least three years are required in our climate. A longer post-treatment period is needed if long-term effects, such as forest regrowth, are studied.

Reference model simulation

The above problem with weather-dependent fluctuations may be overcome, at least to some extent, by conceptual hydrological modelling. Instead of a gauged reference basin, the modelled runoff or any other variable is then used as a reference. Calibration of the model is done for the pre-treatment period. Hydrologically based model analysis is more capable of coping with weather-dependent fluctuations, but it is also more susceptible to inconsistencies in climatological records, and this has to be controlled. The problem of homogeneity is transferred from the runoff record of the control data set to the climatological data used by the model.

The model approach is more informative than the pair-basin method, since it relates streamflow or other variables to the factors that influence them. It is, of course, of fundamental importance that the model manages to describe the processes in an acceptable way and that the model is properly calibrated. If not, it can give a totally misleading result. This means that the processes and the most important factors that influence the processes must be known. The model approach is less expensive because only one basin is needed.

The treated small basin can be used to identify suitable model coefficients for pre- and post-treatment conditions. The model can then be used to extrapolate known results from a small research site to an integrated part of a large basin.

The water balance model

The PULSE water balance model is used to study the effects of clearcutting on runoff with the same data as Rosén used. The results are transferred to a large basin (I). Another application is found in paper II b, where the effects of mine tailings deposit on runoff and pH are studied. A modification of the same model is also used in the studies of non-point source pollution of nitrogen (IV, V and VI).

The PULSE model (Carlsson et al. 1987, Brandt 1987 a) is a modification of the HBV runoff model (Bergström and Forsman 1973, Bergström 1975, and 1976) which is being used in several countries for hydrological forecasting and for estimation of design floods.

Both models can be classified as conceptual runoff models. They are empirical, which means that some of the coefficients (parameters) have to be found by calibration and can not be subject to too far-reaching physical interpretation.

The models include conceptual descriptions of the most significant hydrological processes, such as precipitation, snow accumulation and melt, soil moisture storage and evapotranspiration, runoff generation and routing of the flow down the river system. Input data are point measurements of daily precipitation and temperature together with monthly standard values of potential evapotranspiration.

The models convert point precipitation into areal average values by the use of fixed weights for each station, and also take account of increase of precipitation with altitude as well as temperature decrease. Homogeneous and representative meteorological stations are essential (Brandt 1987 b). The homogeneities of the stations have been tested by double-mass plotting in all studies of this thesis.

The PULSE model is further described in paper I. It is normally structured into submodels, defined by the outlet points of significant lakes, to consider the effects of the lakes on the shape of the hydrograph in a more physically correct way. This is not possible when a lumped structure is applied. This substructure of the model limits the demand on model calibration as concerns recession coefficients (Bergström et al. 1985 a) and provides the model with a realistic time distribution of the flow contributions from different parts of the basin. The latter is a very important feature when analysing the various consequences of forest management practices (I).

Alkalinity and pH-routine

In the alkalinity-pH routine linked to the PULSE-model (Bergström et al. 1985 b and paper II b) no account is taken of the variations of the acidity in the precipitation, except for the direct precipitation on water surfaces. Instead it is assumed that the water quality will be determined by chemical processes in the soil, and that a considerable water exchange occurs as the water passes through the unsaturated zone. This will level out the effect of temporary variations in the composition of the precipitation and also the variations in the dry deposition. The model is therefore not suitable for forecasts of long-term changes due to acid precipitation. The aim of the model is to analyse short-term variations only (Bergström and Lindström 1989).

The routine is based on alkalinity (or acidity if negative). The fundamental concept is that the alkalinity of the water is determined by the location or depth in the model aquifer from where the water drains. In early versions (used in paper II b) there is a seasonal variation in the depth/alkalinity relationship with higher alkalinity in summer than in winter for a given groundwater level. As a final step, the alkalinity is transformed into pH by a fixed relationship.

The water balance model together with the alkalinity routine will result in a general variation pattern of flow and pH, as illustrated in Figure 2.
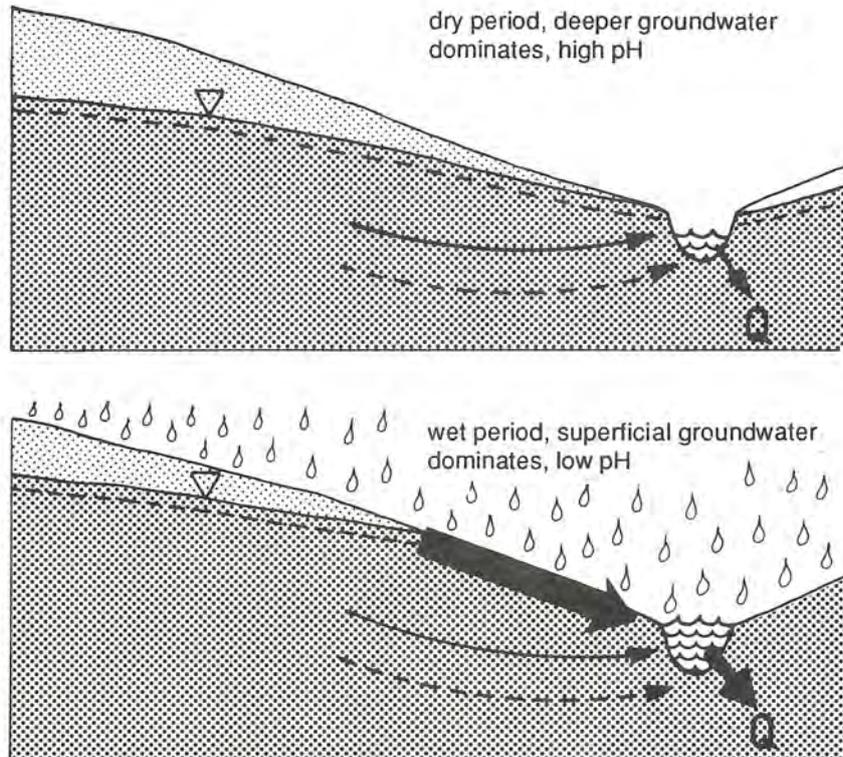

Figure 2. The dominating runoff contributions during dry and wet conditions and their effect on pH (from Bergström 1985).

Nitrogen routine

The development of a nitrogen model is a major part of the work behind this thesis. The initial aim was to develop a model structure which includes the most important factors influencing nitrogen (nitrate) turnover and leaching, but with a complexity that does not exclude application to areas with normal data coverage.

In contrast to some other models of nitrogen turnover in arable land (Hansen and Aslyng 1984, Johnsson 1990, Knisel 1980) our approach is areal. This means that the drainage basin and not the soil profile is the subject of our study. One problem with a soil profile approach is the transfer to areal conditions due to spatial variabilities in particular in larger basins of mixed land use. We have therefore accepted more crude empirical routines when describing nitrogen turnover than is normal in the more process-oriented models.

The loss of nitrate from the soil is an integrated result of a number of processes, all more or less controlled by physical environmental factors, such as soil humidity, soil temperature, and water movement, but also of management factors, such as

fertilization, crop rotation, and cultivation. The first step was to develop a model that described this quantitatively and to test it against long-term measurements from small agricultural fields (IV). The main sources considered in the model are fertilization, mineralization, and atmospheric deposition, and the main sinks are uptake by plants, leaching, and denitrification.

The processes are conceptually driven by hydrological conditions in the soil, estimated by the water balance model. Two mechanisms control the leaching of nitrate in the model - transport by percolating water through the unsaturated zone and washout below the groundwater level.

The coefficients of the water balance model were first calibrated against runoff data from the agricultural field. After that the nitrate part of the model was calibrated and validated against water quality measurements.

The next step was to develop a nitrate turnover model for a middle-sized basin with mixed land use and lakes. This model version is described in paper V. The aim was to find an operational model for calculation of the nitrate transport from a river system with mixed land use. Now the nitrate turnover model for each specific sub-area had to be treated in a rather crude manner to keep the complexity of the model under control. The model structure for each land use category is illustrated in Figure 3. In the lake box, contributions from all areas upstream are collected and mixed.
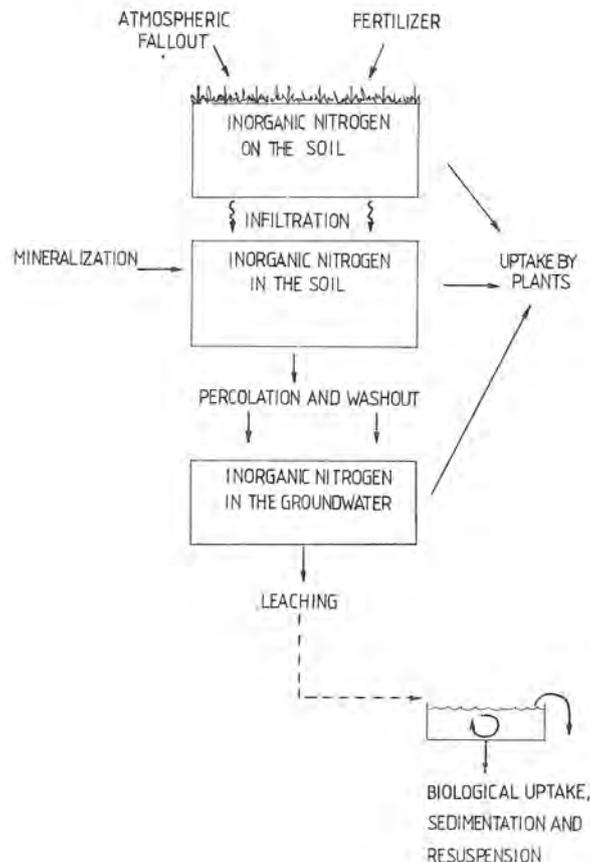

Figure 3. General sketch of the nitrogen model structure (from paper V).

The nitrate sources considered are atmospheric deposition, mineralization and fertilization (as recommended ratios), and the sinks uptake by plants, leaching and uptake in lakes. All processes in lakes, including biological uptake, sedimentation and resuspension, exchange between organic and inorganic nitrogen, and denitrification, are summarized in a simple exponential loss function, based on temperature and nitrate storage in the lake.

The empirical coefficients of the nitrate models for different land-use areas were calibrated against measurements in small homogeneous fields and basins. These calibrated coefficients were then used for a whole basin with mixed land use but without lakes, and their correspondence was tested against observations. Lastly the lake model coefficients were calibrated for a larger basin with lakes and verified against an independent period.

The main drawback of this type of conceptual model is the large number of empirical coefficients, which have to be found by calibration, and several uncertain input data. Therefore, the nitrate turnover model was abandoned and a new simplified version was developed and tested in larger drainage basins (V, VI). This simplified model approach takes care of both nitrate and inorganic nitrogen, and is based on typical monthly concentration values obtained from monitoring programmes in small homogeneous areas. Nitrate especially has a developed annual regime with high concentrations from late autumn to early spring and low concentrations in summer due to both hydrological and pedological controls (Webb and Walling 1985). The daily leaching from each land-use area in a basin is computed as the product of runoff from the area calculated by the water balance model and the corresponding typical concentration. The total nitrogen transport for the whole basin is then computed by adding leachings from all land-use areas. The nitrogen retention in lakes was modelled with the same simple routine as above. In one application (VI) it is based on the typical change of the phytoplankton quantity in the water body, and an exponential decay function.

It was shown (paper V) that the simplified approach gave results that compared well with those of the more complex model in medium sized basins of mixed land use.

If the nitrogen model is calibrated or verified for a basin before a treatment and the control program continues, the model can be used to quantify the effects in the stream or in the lake and to distinguish them from natural weather-dependent effects. The model can also be a useful tool for selecting convenient control strategies and for estimating their effect on nitrogen transport to the sea, if their local effects are known (Ryding and Rast 1989).

Model calibration, validation and sensitivity analysis

In all modelling work behind this thesis emphasis has been put on control against observed data. As the models are empirical and adjusted by calibration a good correspondence does not guarantee that the processes are described correctly. "It must work well for the right reason" (Klemeš 1986). Good model performance on repeated applications to different data sets is a help to support our confidence in the basic processes and the linking of these in the model. Without this control it

would be very difficult to say anything about the success or failure of our modelling exercise even if the model performance is no absolute guarantee of proper process description.

The modelling processes consist of several phases. First input data, such as daily mean air temperature, daily precipitation, mean monthly potential evapotranspiration, daily runoff, and observations of different water quality variables have to be prepared. The basin has to be divided into sub-areas. Calculations of altitude distribution, sizes of different land-use areas and of each sub-area have to be made from maps. Land use and crop distribution (for the nitrogen model) can be found by map analyses, classification by Landsat satellite images, or agricultural statistics (VI).

The second step is calibration of the model. This means that the empirical coefficients of the model, its parameters, are adjusted so that the simulation agrees with observations. It is important that the observed record includes enough variations for this procedure. This means that there must be significant events like floods, dry periods and changes in concentration. In our model all calibrations are based on visual comparisons between graphs with some support from statistical criteria.

The third step is model validation (sometimes named verification). This means that the calibrated model is run over an independent test period with data sets that are not used in the calibration process. If the model is made too complex with too many coefficients, it is likely to show worse performance over the validation period than over the period used for calibration. The model is then overparameterized and modelling has turned into curve-fitting.

In most of the studies of this thesis we have tried to save an independent period for model validation. Sometimes the records have been taken from too short a period for this procedure, which means that the risk for overparameterization is there. We have, however, stated if the simulations refer to calibration or independent data when presenting the results in the papers (II b, IV - VI).

The combination of two semi-empirical models, one for water balance and one for pH or nitrogen loss, makes it difficult to generalize the model coefficients. If the water balance model is recalibrated and its optimum coefficients are adjusted, we may very well need to recalibrate the hydrochemistry routine as well.

The fourth step in a model study is a sensitivity analysis of the response of the model to changes in all its assumptions and coefficient values. This is particularly important if the effect of a coefficient is vague and if there is strong interaction between components of the model. A complete sensitivity analysis is hardly possible, however, because of the large number of possible combinations. Examples of sensitivity analyses are given in papers II b and IV.

It is often stated that a more physical description of the system reduces the need for calibration (see, for example, Refsgaard et al. 1989). Also in formulations as complex as the SHE model there are, however, inevitably approximations in the representations of the physical processes, which lead to calibration coefficients (Bathurst 1986, Beven 1989).

If a conceptual model is applied to a great number of basins, experience from calibration of its coefficients is sometimes used as the foundation of uncalibrated simulations of runoff. This technique has been adopted by SMHI to estimate runoff data for recipient control programmes (Johansson 1986). An overview of suitable coefficient sets for the HBV model is given by Bergström (1990). It has to be remembered, however, that the modelling based on generalized coefficients yield runoff values of lower quality than those from a properly calibrated model.

EXAMPLES OF WEATHER-DEPENDENT FLUCTUATIONS AND OTHER NATURAL EFFECTS

Natural effects on the water balance

The weather fluctuations between years can be quite considerable. This has become particularly clear in the sometimes extreme summer and winter weather of recent years. As mentioned above, a conceptual hydrological model based on climatological records can manage to describe the effects of weather fluctuations on the water balance of a basin.

Rain or snow can be intercepted, evaporated, stored as snowpack or in temporary ponds on its way down, and, if infiltrated into the ground, delayed due to temporary storage in the soil or taken up by vegetation and transpired. The contribution from rain or snowmelt to runoff, called effective precipitation, differs from the precipitation, both as concerns volumes, seasonal patterns and extremes. This is illustrated in Figure 4 which shows results from a simulation by the HBV model applied to the Blankaström basin in river Emån. Maximum daily values of areal precipitation, areal snowmelt and areal effective precipitation are extracted from the model together with the soil moisture conditions for each day of the year. The plots are of the same scale, and the dominating effect of the soil moisture deficit is obvious. All the simulated areal soil moisture deficit plots for the years 1944-85 are shown in the figure. Between years the deficit can vary from about 50 to 120 mm at the same time of the year.

The Blankaström study above has been repeated for 25 larger basins in Sweden with the HBV model in connection with the Swedish spillway design investigation (Brandt et al. 1987 b, Bergström et al. 1988). The data base covers almost 500 station-years of hydrometeorological data (precipitation, temperature and runoff). Figure 5 shows the minimum soil moisture deficit for each day of the year and for each basin. The differences between northern and southern Sweden are quite obvious. In southern Sweden the minimum soil moisture deficit in summer is at least about 100 mm. The average deficit is, of course, considerably higher.

Extreme meteorological and hydrological events have been in focus during the last years due to the work on new guidelines for spillway design (see the Swedish Committee on Spillway Design 1990). It is concluded that floods in Sweden are generally caused by a critical combination of precipitation, low soil moisture deficit and snowmelt, rather than by extreme precipitation alone (Lindström 1990).
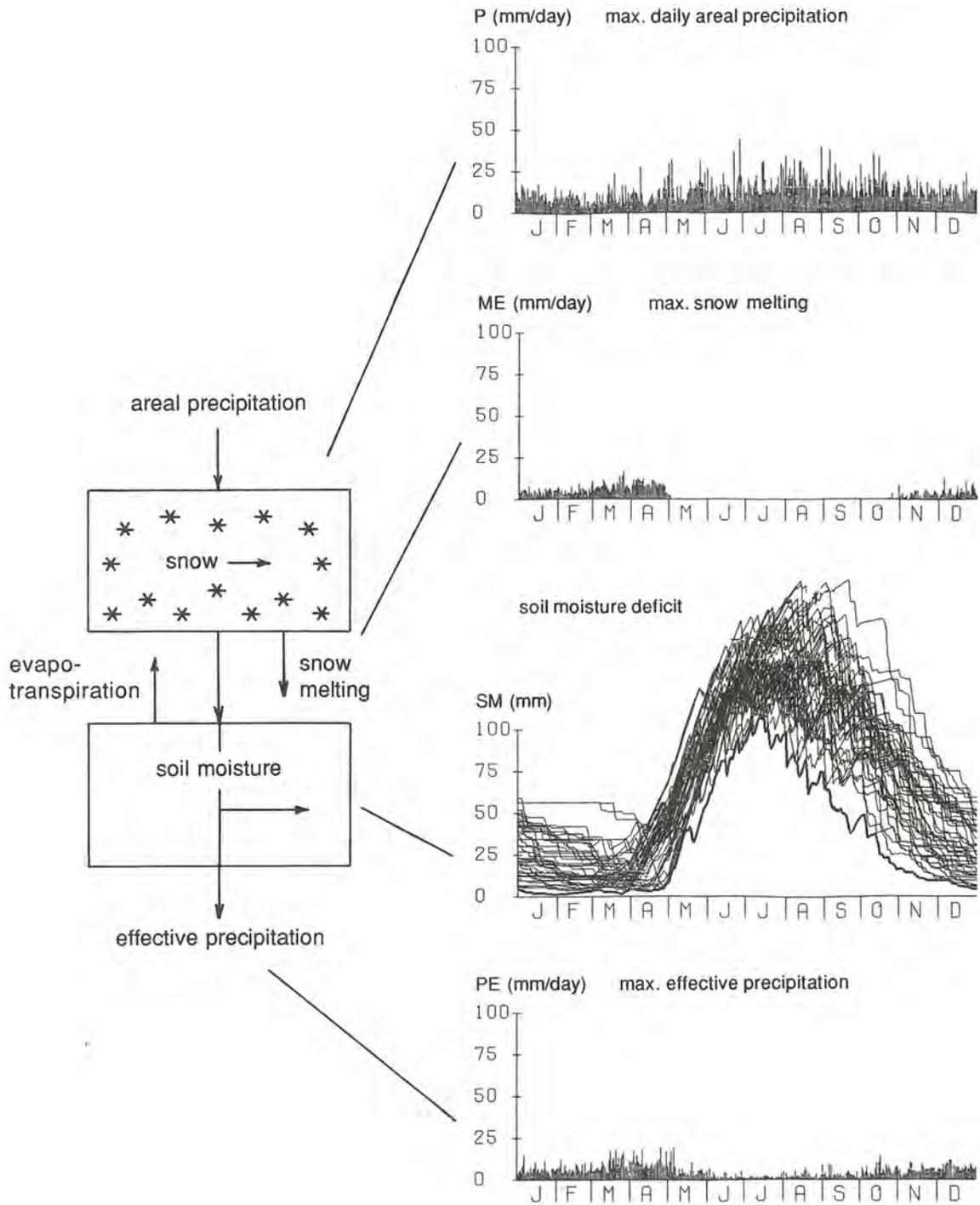
Figure 4. Extreme water balance components for the Blankaström basin (3 446 km²) over the years 1944-85. The extraction from the model is illustrated to the left. Note that the plots are of identical scale (Brandt et al. 1987b).
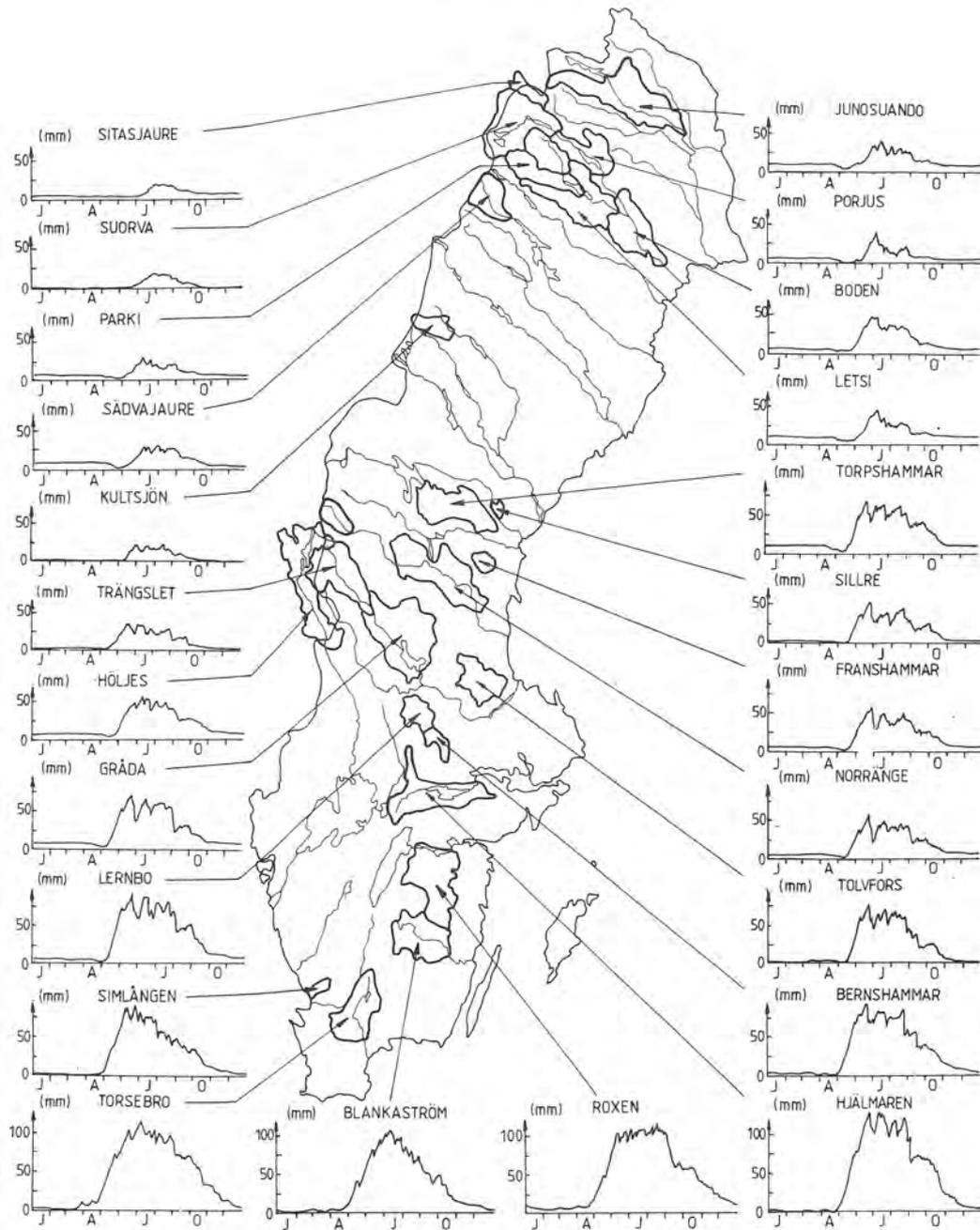
Figure 5. Minimum areal soil moisture deficit for larger river basins in Sweden as computed by the HBV model (Brandt et al. 1987 b)

Brandesten (1987) has shown by analysis of principal component that the variations in daily runoff from eight small basins (up to 10 km²) were mostly governed by the climate. Only 5 % of the variations could be related to the percentage of forest cover and to the annual variation in the evapotranspiration of the trees.

Lakes have a damping effect on the runoff in a river system. This is illustrated in Figure 6 by the simulated runoff from a sub-basin without lakes and the several times larger lake-rich basin to which the sub-basin belongs (from paper II a).
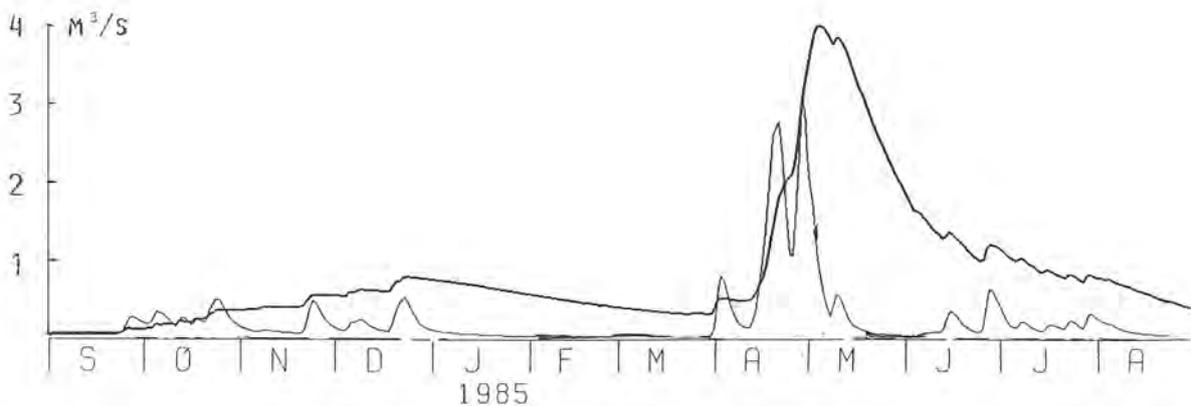
Figure 6. Model simulated hydrographs from a forested area without lakes (thin line) and the outlet of a large lake further downstream in the system (thick line) (from paper II a).

Runoff from arable land also depends on many natural effects in the basin, of which climate, soil type, and morphology are the most important. A comparison of effects of a sandy and a clayey soil in small drained arable fields in south Sweden shows that the runoff from a clayey soil is much more fluctuating and has a quicker response to precipitation than the runoff from a sandy soil (Gustafson et al. 1984).

Natural effects on water transport of substances

Effects on erosion and sedimentation

Erosion of particles by water is a process of detachment and transport of soil particles by raindrop impact and surface runoff. In well vegetated areas without human impact the sheet and rill erosion are normally very low. In streams fluvial erosion can occur either as bank cutting or bed scouring. Fluvial erosion is dependent on bank material, water velocity and vegetation shelter. In steep mountain regions above the timberline it can be quite large and even in rivers cut down into easily eroded sediments. Larger particles deposit easily as water velocity decreases. The deposition of larger particles in lakes, building up of deltas and other depositions, are well documented. In calm parts of a river system and in lakes even finer materials are sedimented (examples of documentations from Swedish rivers: Arnborg 1959, Axelsson 1967, Hjorth 1972, Hjulström 1935, Sundborg 1956, Céwe and Norrbin 1965). In paper III an example (from the River Dalälven) of deposition of suspended material in a lake system is discussed. The sedimentation effect is higher at larger flows, when we have peaks in transport of coarser materials, but it is very difficult to quantify the effect of lake sedimentation and to generalize it to other river systems from this type of comparison of sediment yields at only two places.

The concentration of suspended material is dependent on water velocity which increases as runoff increases. To calculate sediment transport a simple regression relationship (eq.1) is often used (Miller 1951, Nilsson 1971, Walling 1977, Jansson 1982, and 1985):

$$T = a \; Q^b \qquad (1)$$

where T = the sediment transport,
        Q = the runoff,
        a, b = empirical coefficients.

Eq. 1 is site-specific and the measurements often show great scatter around the regression line. Nevertheless, it shows that the seasonal and interannual variations in sediment transport in larger basins in Sweden are mostly determined by hydro-meteorology and runoff, as illustrated by Figure 5 in paper III.

There is often an interaction between geology, soil, and land use. Till soil areas are often covered by forest and sediment areas are usually cultivated. It is therefore difficult to separate the effects. Two examples mostly influenced by geology are shown in Figure 7 in paper III.

Effects on alkalinity and pH

Natural short-term variations in alkalinity and pH in streams in a forest basin can be explained to a great extent by temporary variations in the hydrological situations (Bergström et al. 1985 b, Monitor 1989, paper II b Figure 7). This process has to be separated from long-term acidification caused by exposure of the system to acid precipitation. It has been shown that the short-term variation pattern can be described very well by a simplified empirical hydrological and hydrochemical model, like the PULSE model if it is properly calibrated to an adequate data base (Bergström et al. 1985 b). This model has the advantage of very limited data requirement, and it can be used as a tool to separate natural variations from trends. Simulation of long-term effects of the system requires more process-oriented models, like the MAGIC model (Cosby et al. 1985). A more thorough discussion on modelling objectives and model complexities as concerns the effects of acid rain on runoff is given by Bergström and Lindström (1989).

Effects on nitrogen leaching

The yearly atmospheric nitrogen deposition today in southern Sweden is 15 - 25 kg/ha, in central Sweden 10 kg/ha and in northern Sweden less than 10 kg/ha (see for example Monitor 1981).

In Swedish forests nutrients are present in limited quantities. A forest can accumulate about 3 - 20 kg/ha per year in increased biomass. The finely branched system of roots is very effective when absorbing the nutrients. The low concentrations and transport of nitrogen from forests are illustrated by nitrate measurements in small forest basins in south central Sweden in Figures 3 and 4 in paper V. The same is valid for inorganic nitrogen. Increased nitrogen leaching from forest areas in south Sweden at Söderåsen in Scania county, directly or indirectly due to human effect, is discussed (Nihlgård 1990), but the homogeneous small basin used in the Ring-

sjön paper (VI) has still low concentrations. The yearly nitrogen losses are 3 - 6 kg/ha. The transport of nitrogen from forests is therefore normally low (Rosén 1982) and follows the runoff fluctuations with the largest transport during high flow.

Lakes act as sinks for nutrients by denitrification, biological uptake and sedimentation. This is reflected in low concentrations in summer and higher in winter (Monitor 1989), a significant factor for transport calculations.

EXAMPLES OF HUMAN IMPACT

Human impacts on the water balance

Clearcutting

Hydrological effects of clearcutting have been studied by many research projects in different countries. Reviews of studies are presented by Hibbert (1967), Bosch and Hewlett (1982), and Grip and Lundin (1987). In general the investigations indicate an increasing water yield and higher groundwater levels following the clearcutting of forested areas. This is commonly explained by reduced interception, and evapotranspiration from the trees (Federov and Marunich 1989). In addition, more intensive and earlier snowmelts are generally reported in open areas than in forests (for example, Brandt 1986).

In our study of clearcutting (paper I) we used data from central Sweden. Mean annual precipitation for the district is 650 mm, but the first years after treatment were wetter. Runoff (untreated conditions) for all the years studied varied from 200 to nearly 400 mm during the wet years. The empirical model coefficients of the hydrological model were set by calibration against observed runoff for the pre-treatment period for the basins. The effect of clearcutting could then be determined as the difference between simulated and recorded runoff when the model was run with the meteorological data measured after clearcutting. Between 70 and 100 % of the basins was harvested. The yearly increases of runoff were obvious and varied between 165 and 250 mm (the figure 200 mm is incorrectly given in paper I). This means an increase of 40 to 75 %. The greatest effect was observed in spring, when the increase was 50 to 120 mm during spring flood. During the summer and autumn periods, the increase varied between 75 and 95 mm. We have also tried to analyse data from the project at Himmelsberget by the same method (Brandt et al. 1987 a), but poor data homogeneity hindered quantification of the effects.

A new model calibration was done for the post-treatment period. The difference between pre-harvest and post-harvest conditions in the model simulation is shown in Figure 7. Our study shows that the effect of clearcutting in a humid climate region can be summarized as follows: snowmelt starts earlier and is more intensive, summer and autumn conditions are significantly wetter due to a smaller soil moisture deficit.
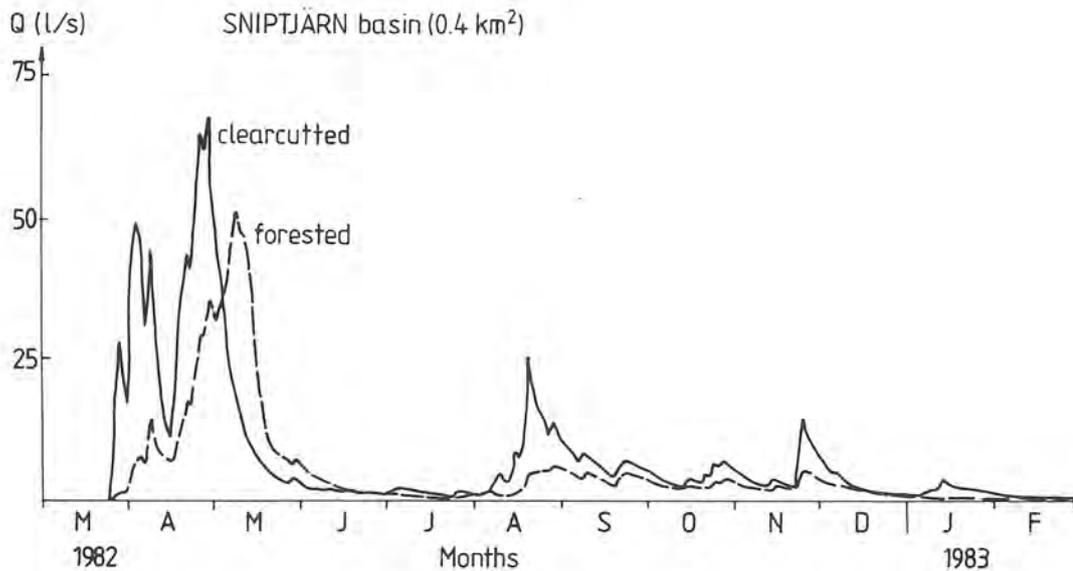
Figure 7. Runoff simulations for the Sniptjärn basin, illustrating the effect of clear-cutting as described by the model with two different sets of coefficients (from paper I).

Hibbert (1967), and Bosch and Hewlett (1982) have compiled a large number of studies, mostly from the USA, and have shown that a reduction of forest cover increased water yield. The magnitude of treatment response varied considerably from 34 mm to 450 mm per year after complete cutting. Figure 8 is taken from Bosch and Hewlett and shows the result from the pair-basin studies during the first five years following harvest, together with our results. The response to treatment is highly variable, and it is not predictable from this diagram.
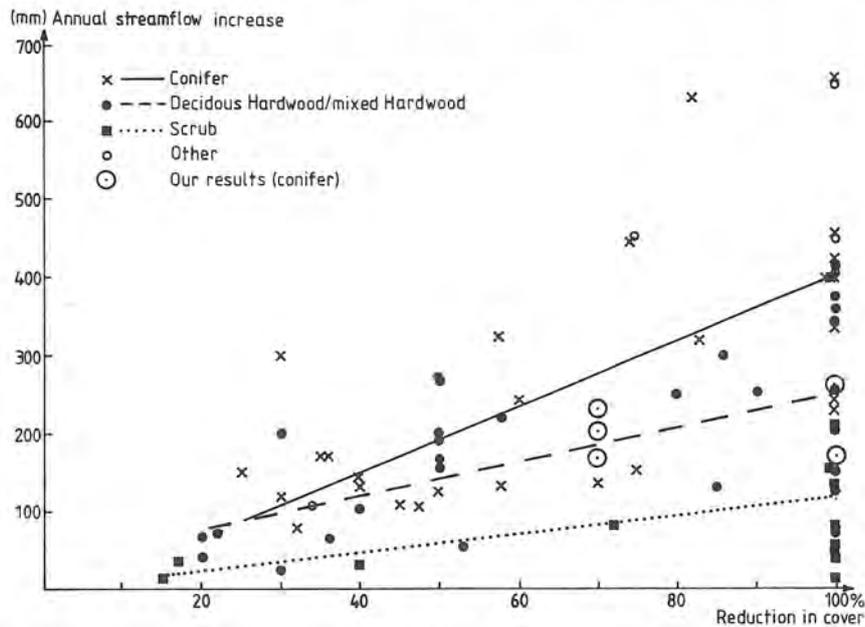
Figure 8. Water yield increases following changes in vegetation cover. The diagram is redrawn from Bosch and Hewlett (1982) and completed with results from the clearcutted basins Kullarna and Sniptjärn (circles).

A summary of basin experiment results is complicated because of variations in experimental conditions. Topography, climate, soils and basin size influence the result. Figure 9 shows a summary of water yield changes after clearcutting of conifers and scrubs as a function of annual precipitation taken from Bosch and Hewlett and with our results added to it.
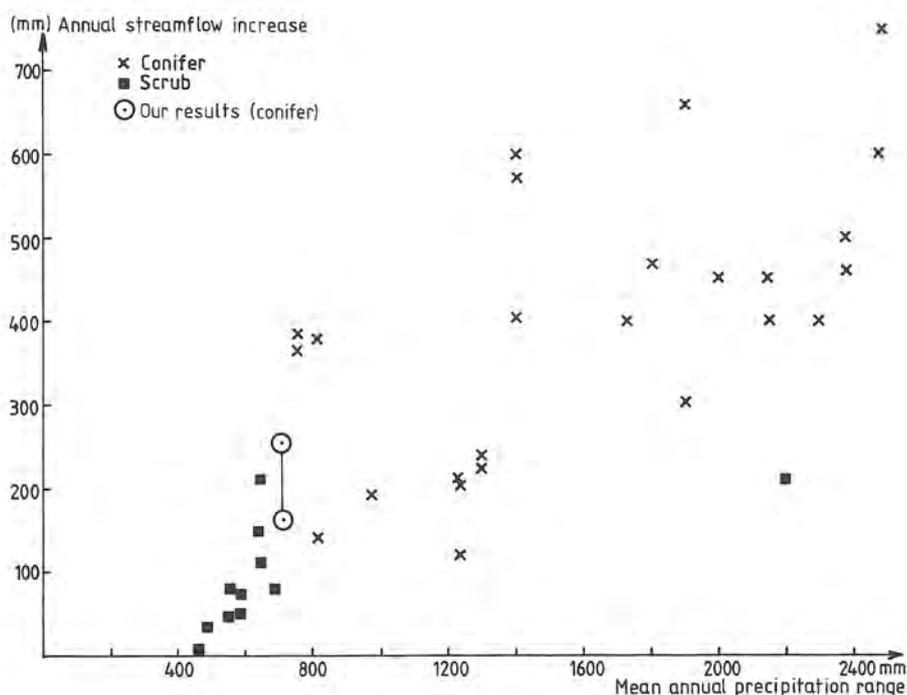

Figure 9. *The summary of water yield changes after clearcutting of conifer forest and scrubs as a function of mean annual precipitation from a review by Bosch and Hewlett (1982) and completed with results from the clearcutted basins Kullarna and Sniptjärn (circles).*

The Swedish data seem to fit in well, but it must be noticed that our study has been made in a different climate region from most of the others and has a much lower potential evapotranspiration. If the basin is too small, errors from the failure of the water divides can be substantial. If the basin is too large it can be difficult to control treatments. Bosch and Hewlett recommend a size of 50 - 100 ha.

Larger basins are seldom totally clearcutted. About 1 % of Sweden's forests is clearcutted every year. It takes several years before new trees grow up again and the situation starts to return to pre-treatment conditions. In paper I we hypothetically tested a 10 % partial clearcutting in a larger basin. The basin was divided into sub-basins, and the two coefficient sets were used to simulate the effect of hypothetical cuttings in different parts of the basin. The results indicate that the effect on runoff of a 10 % partial clearcutting (or 1 % clearcutting per year or less) is relatively small and, for the spring, related to the location of the cuttings in the basin. Cutting in the upper part of the basin will lead to a more intensive and

earlier snowmelt in this part of the basin. The study showed a peak flow increase of 9 % during spring floods due to 10 % of cut areas. Clearcutting near the outlet of the basin results in a more evenly distributed spring flood. According to the study, peak flows at spring flood with cutting near the outlet were unchanged or even lower. Peak flows during summer and autumn increased by up to 5 % wherever the cuttings were done.

This hypothetical study was partly confirmed by another study (Persson 1987). Two sub-basins of the Kassjöån drainage area (the same drainage area as used above) were partly harvested by up to 7 - 10 % over a rather short period. The PULSE model was calibrated for the sub-basins before the harvest and run with the meteorological data measured after the treatment. The study showed no obvious change in runoff, which also indicates that in larger river basins the effects of a clearcutting of 10 % are fairly small, but that they can have a strong local effect in smaller basins.

Forest growth

Forest biomass in Sweden has increased during the last 50 years according to biomass taxation (National Forestry Register) performed by the National Board of Forestry. This has been suggested as one possible cause of long-term variability of the water balances of Swedish river basins. An increased forest biomass will increase evapotranspiration losses (Major 1975), and this should effect runoff.

In the spring of 1989 a project started to analyse relatively long hydrological and climatological records jointly and to relate these to forest growth (Jutman et al. 1989). I am now responsible for the project which is still going on and a study from the Dalälven river (ca 29 000 km^2) has been completed so far. In that particular area the biomass has increased with some 25 % since 1945 (Jutman et al. 1989). Ten to eleven precipitation stations and five temperature stations were used to simulate runoff by the HBV model at eleven stream gauging stations in the river. The simulated runoff was compared with runoff records. In Figure 10 preliminary results from the study as accumulated runoff differences (computed minus recorded runoff) for some of the stations in the Dalälven river are shown. An increase in the accummulated difference means that there is a decrease in the observed runoff if the computed runoff is homogeneous. Figure 10 shows breaks in the trend lines, which indicates that something has happened to the water balance.

The project proceeds with studies in other rivers in both north and south Sweden, and a deeper analysis of forest biomass growth and age. A study of this type has to be interpreted with great care. There is an obvious risk that lack of homogeneity in the climatological records will be interpreted as inhomogeneity in the water balance, because hydrometeorological data are used in the reference model. The gradual introduction of wind shields for precipitation gauges makes older records less valuable. Even more difficult is the quantification of the effects of growing awareness of the need for sheltered sites for precipitation measurements.
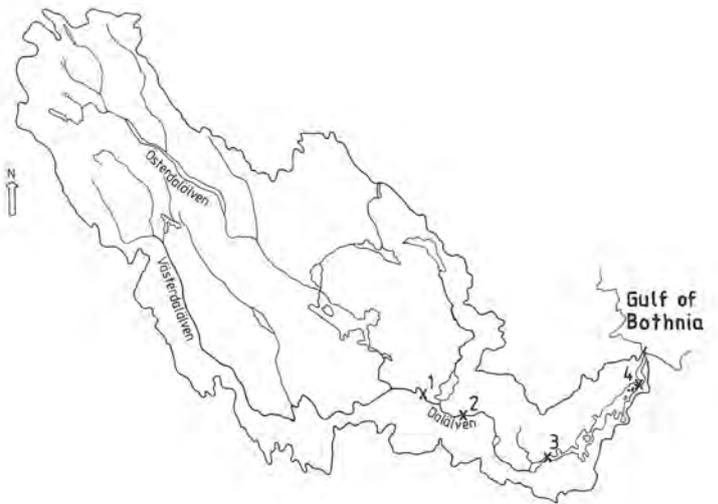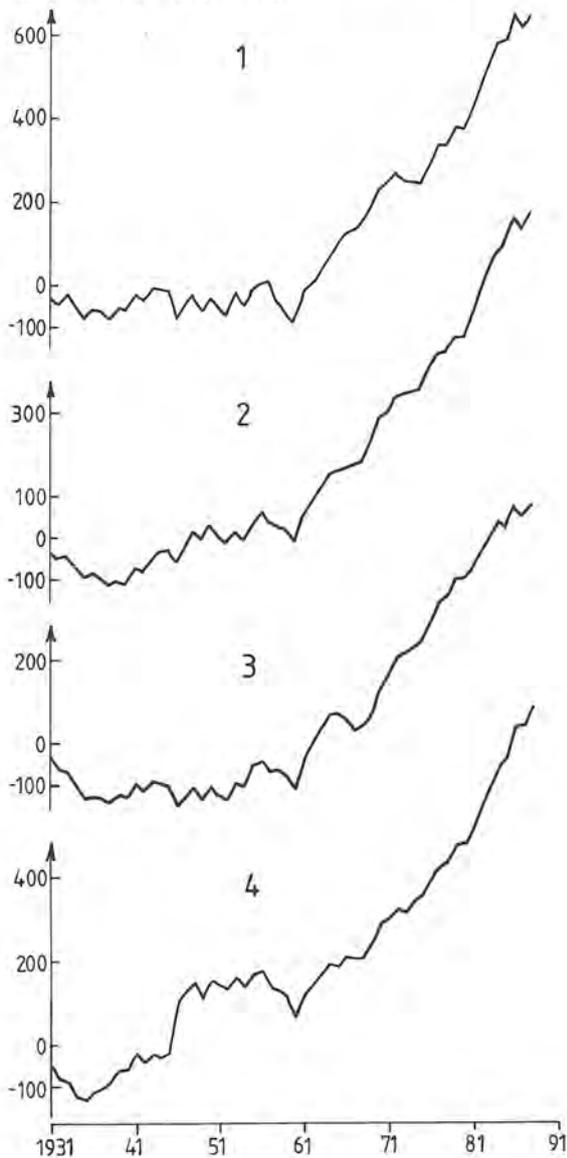
Figure 10. Accumulated differences in runoff (mm) calculated as computed (HBV model) minus observed runoff at some stations in the Dalälven river.

Drainage

Drainage of wetland and forest has increased considerably in recent years, mainly through increased drainage of clear-felled areas (Simonsson 1987). The area of peatland planned for peat production is relatively small. The purpose of draining is to lower the groundwater level so the forest will grow better. An unsaturated zone with a capacity for temporary storage of precipitation and meltwater is thus created. This means reduced flow peaks if the upper zone is dry. The total evaporation is reduced after draining a mire without vegetation, but evapotranspiration will increase gradually as the forest develops.

Results from different drainage studies are not consistent. Increasing high flows have been reported by for example Braekke (1970), Seuna (1974 and 1988), Mustonen (1975), and Bergquist et al. (1984), decreasing high flows by Multamäki (1962) and both increased and decreased peak flows by Lundin (1984) and Heikurainen et al. (1978). Low water flow rates generally increase after drainage (Braekke 1970, Heikurainen et al. 1978, Seuna 1988). Hyvärinen's and Vehviläinen's (1980) interpretation is that the differences in the draining effects of spring flood between southern and northern Finland to some extent depend on differences in type of wetlands and in forest growth.

I have tested runoff data from two Swedish pair-basin projects on effects of drainage with the model approach (Brandt 1987 c). The extent of drainage in the basins was 8 - 14 % of the total areas. The conclusion of the analysis was that there are only small changes in runoff caused by a 10 % draining of a basin. The precision of the model is not high enough to discern the changes.

Mining deposits

Mining can have a dramatic influence on the flow regime but normally only small areas are affected. In paper II b effects of mine tailings deposit have been studied. The deposits had no topsoil and a minimum of vegetation cover and consisted of materials from gravel to block sizes which means a very small soil moisture storage and a quick response to percolating water. The water balance of the deposit was modelled according to these conditions. As there was no possibility to measure runoff from the deposit directly, the model was indirectly controlled by estimations of the deposit's fraction of total runoff at an observation site further downstream, which were based on hydrochemical observations in the basin.

There was normally no runoff from the deposits when the area was snow-covered. The snow melted earlier on the deposit than in the forest involving a larger fraction of deposit-affected water when the spring flood started in the stream. The evapotranspiration losses from the deposit were smaller than those from the forest areas, and the percolation through the soil was quicker, which means that the fraction of deposit water in the stream could be fairly high in the summer.

Human impacts on transport of water-carried and dissolved substances

Effects on erosion and sedimentation

In paper III the effects of man-made changes of land use, such as bare fields, are discussed. The results are difficult to generalize as erosion rates and sediment yields from plots and small basins are not directly comparable with those in larger basins, mostly because of deposition losses (Brandt 1982, Walling 1988).

A study in southern Wisconsin, USA (Trimble 1981) has demonstrated the problem of predicting sediment transport and sedimentation after changes in agricultural management. During an initial period, 1853 - 1938, the management of land led to severe erosion. Conservation measures were introduced during the second period, 1938 - 1975, and the sheet and rill erosion were reduced by about 25 %. However, after conservation the sediment transport out of the basin remained the same

as before, owing to the fact that sediment stored in the valley area was remobilised.

Heede (1987) states from studies of sediment delivery caused by timber harvesting in the USA that the largest part of all sediment transport there resulted from poor road locations and channel damage by equipment and not from the harvested area.

I have not found any erosion and sediment transport model that is convenient for Swedish conditions. Such a model has to account for the thawing and snowmelting effects in winter and spring. Paper III is therefore a summing-up of results from Sweden. It is difficult to make a distinct conclusion and generalization concerning unmeasured basins.

A trend analysis of the transport of dissolved material was made on the basis of monthly mean values for the period 1967-87. I can not from this analysis confirm any increase of dissolved concentration indirectly due to man-made effects, but the transport of dissolved matter seems to have increased in some parts of Sweden owing to runoff increase during the last ten years. Ahl (1980) concluded that long-term variation in the concentration of chemical substances is due to a great extent to climatic factors. The Yearbook of Environmental Statistics (1987), on the other hand, shows trends for some substances in rivers in Sweden during the period 1971-85. In many rivers increasing trends are found for total nitrogen, for oxygen consuming substances, and for $KMnO_4$-consumption (organic matter). Conductivity shows both negative and positive trends in the observed rivers and sulphate decreasing trends in many cases. When analysing observations from central Sweden it is particularly important to consider the possible effects of extreme flooding of some rivers in September 1985.

Erosion and sediment load in rivers is normally not a serious problem in Sweden. The problem is rather the sediment-associated transport of nutrients and contaminants. Therefore there is a growing interest in physical and chemical properties of fine grained sediments and in deposition and remobilization of sediments.

Effects on alkalinity and pH

The difference in dynamics of the water balance due to mine deposits, discussed above, also affects the alkalinity and the pH of the water (II b). The runoff from the deposit is out of phase with the runoff from the nearby forest basin owing to the former's small soil moisture and waterstorage capacity, and early snowmelt. In winter with frozen conditions the water and hydrochemistry contributions to the stream from the deposit are insignificant, and the alkalinity and the pH of the stream are quite normal for the district. When spring comes the snow melts first on the open deposit resulting in large contributions from the deposit with low pH in the stream as a result. Also in summer the contributions from the deposit are large as there is no evapotranspiration. This results in a low pH in the stream.

One of the main environmental issues in Sweden during the 1980s has been long-term acidification of rivers and lakes (Swedish Ministry of Agriculture Environment '82 Committee 1982). As mentioned earlier this problem has to be studied by more advanced models than the PULSE model, which is restricted to short-term variabilities.

Effects on nitrogen leaching

The contributions of nitrogen leaching from forest areas are normally smaller than from arable land in Sweden (V) if not affected by fertilization or clearcutting. Non-point pollution, such as leaching from arable land, is suggested as one important source of eutrophication of lakes and of the sea, but atmospheric deposition due to increased contributions from industries, traffic and stored manure is also mentioned. Wastewater and water pollution from industries also contribute to the total leaching.

Paper VI demonstrates an attempt to calculate the contributions from different sources to the total nitrogen leaching. Figure 11 shows the yearly budgets of nitrate for the Ringsjön area and their sources. About 34 % of the drainage basin consists of arable land, but the contribution of nitrate transport from this area is as much as 85 % of the total nitrate load to the lake. The atmospheric deposition on the lakes (10 % of the total area) is significant too.
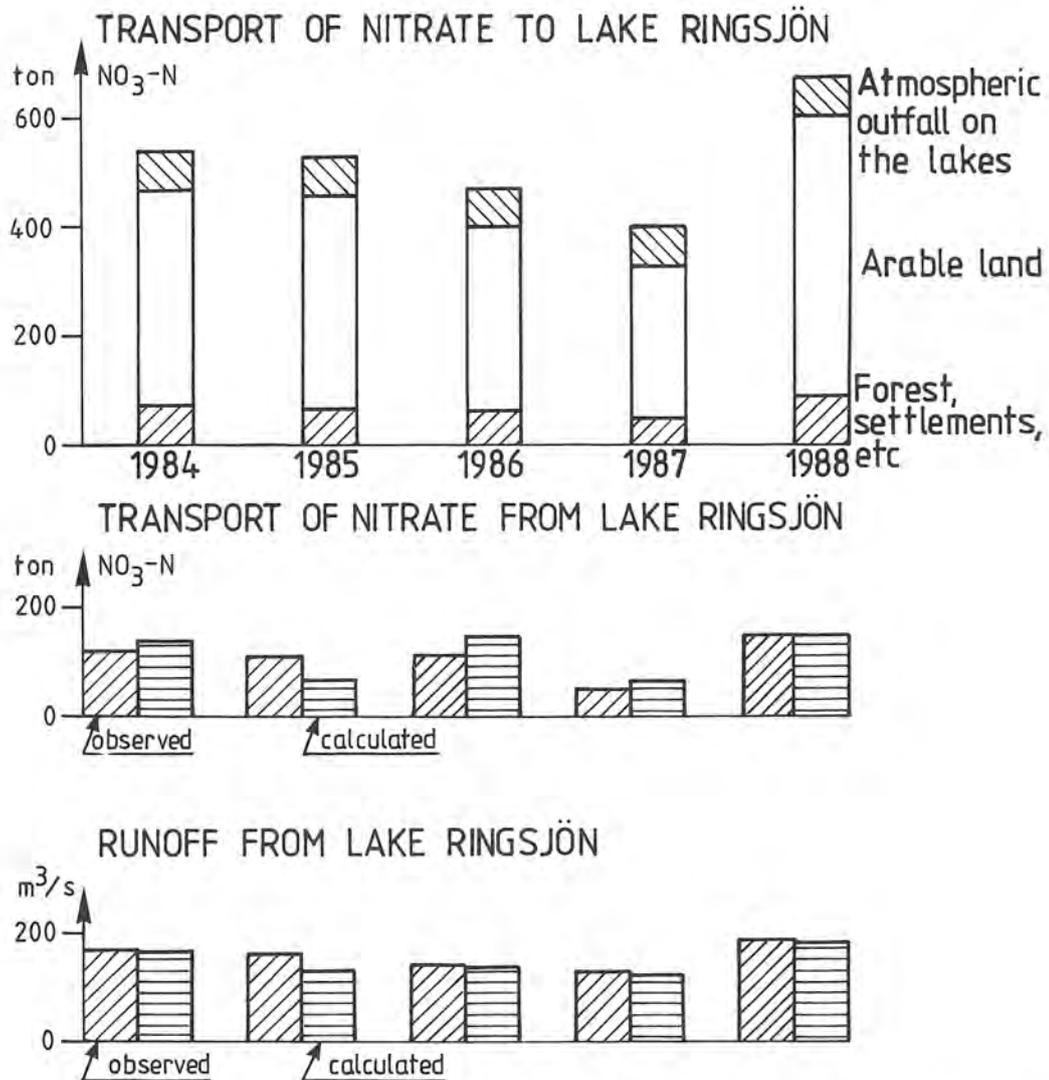
Figure 11. Yearly budgets of nitrate for the inlet and outlet of Lake Ringsjön together with observed and calculated runoff from the lake (from paper VI).

In the model studies (paper VI and unpublished results from Vemmenhögsån in Scania county) I have not seen any great effect of retention in the stream beds. Biological uptake and denitrification are low during the period from late autumn to early spring due to low temperature (Jansson et al. 1990). This is well illustrated by Christensen et al. (1990) in their study in a nutrient-rich Danish lowland stream. They show a pronounced seasonal variation in denitrification activity in the sediment with low activity from winter to early spring and more than ten times higher activity in summer under dark conditions. Under lighter conditions photosynthetic O_2 production increases the oxic zone and reduces denitrification activity during the whole year and by up to 85 % in spring.

Lakes serve as a sink of nitrogen transport in a river system. Therefore lakes in cultivated areas are often eutrophic. Retention depends on biological uptake, sedimentation, resuspension, and dentrification. Retention of nitrogen in lakes depends much on the turnover time of the water in the lakes, but also on temperature. Even the trophic level of the lake is an important factor.

I have found a retention value of 45 - 60 % of total nitrogen in Lake Ringsjön, which means a lake retention of 65 - 110 kg/ha per year. The variations of my results depend mostly on the water balance of each year.

The role of wetland in reducing nitrogen from the watersheds has been debated and restoration of wetlands is suggested as one measure against eutrophication of lakes and of the sea (Fleischer et al. 1989). It is important to consider the hydrological regime, particularly in south Sweden, when discussing nitrogen transport (Figure 12). In this part of the country most of the runoff occurs during the period late autumn until early spring, when the biological uptake and denitrification are low. Thus, to be effective, the wetlands must be large enough to be able to store parts of the floods from winter to spring, and to remain wet during summer, when biological activities and denitrification are at their highest.

A full scale experiment with macrophytes as nutrient removers in special ditches has started in the most polluted tributary (Snogerödsbäcken) to Lake Ringsjön. The highest nitrogen reduction occured in summer (68 %) mostly owing to denitrification. In winter with dominating runoff and transport the reduction was 16 %. Only 9 % of the nitrogen was found in the harvested plankton (Gumbricht 1990). This method can be used to treat sewage water (Aniansson 1990) or highly polluted streams, but it seems too expensive to be used in natural streams. Other measures discussed are change of crops, such as a catch crop in autumn, or a change to more ley. A crude test of the effects of a change to ley cultivation only in the Ringsjön basin with the model resulted in a nitrogen reduction of 10 - 20 % without any regard to indirect effects of reduced nitrogen load in the lake (VI). This test has, however, to be considered as a hypothetical model experiment. It is not a realistic pollution control action.
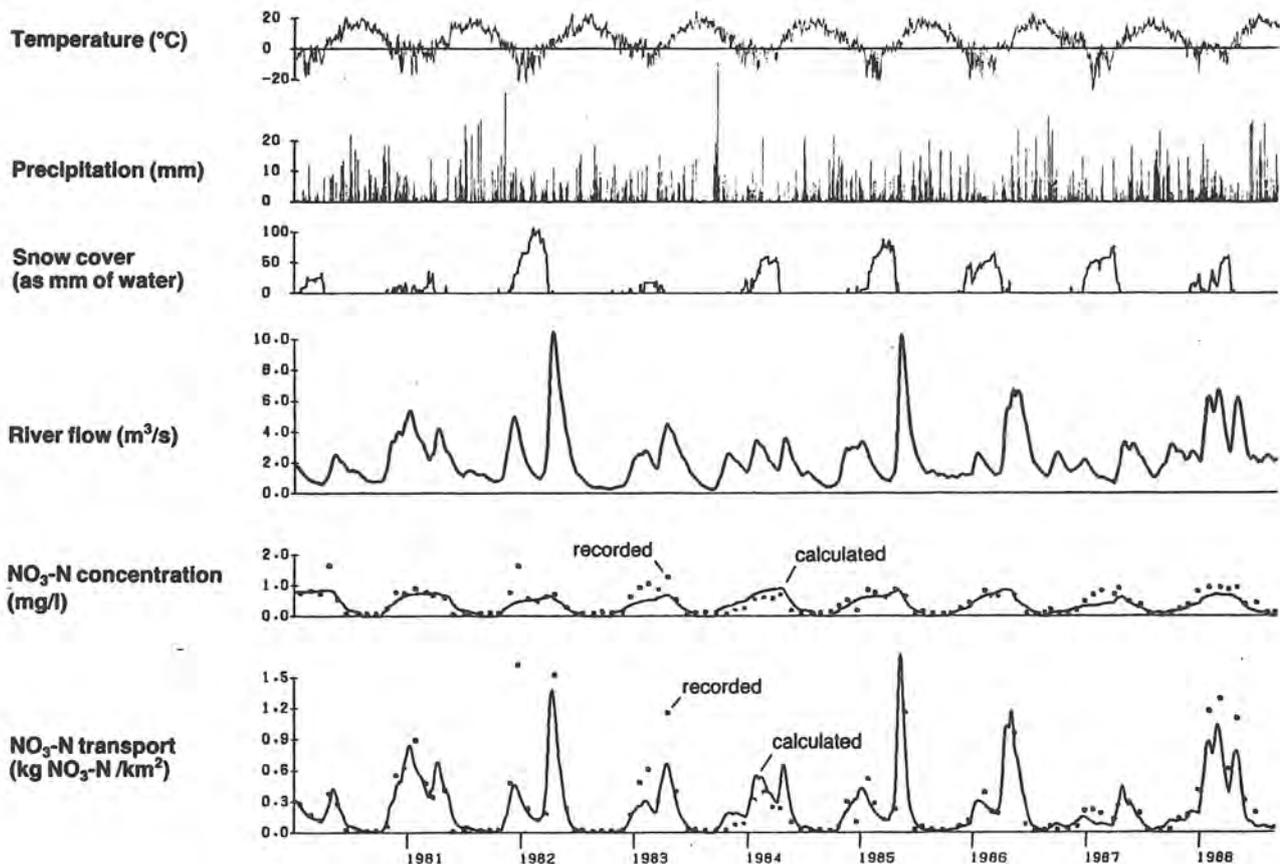
Figure 12. Weather-dependent changes in nitrate concentration and nitrate flux in river Hedenlundaån (Ekenäs research area) (from paper V, redrawn in Monitor 1989).

CONCLUSION

The study shows that water balance and water quality models can often be useful tools for distinguishing human effects from natural short-term weather-dependent influences.

When working with transport problems in rivers, both as to water-carried and dissolved substances, runoff data are essential as the runoff normally fluctuates much more than the water quality parameter of interest. The greatest part of the transport occurs during short periods with high flow, in particular from basins without lakes. As good runoff measurements are expensive, a water balance model of proper performance is a good alternative, and it can also be used retrospectively if precipitation and temperature data are available.

The models used in my studies are conceptual runoff models (HBV and PULSE) linked with simple hydrochemical subroutines. Models of this type have the advantages that they are simple and use input data from a normal data coverage of a basin. They can distinguish weather-dependent fluctuations from other effects. They assume stationary conditions, meaning that long-term changes due to changes in atmospheric deposition, land use or climate can not be handled directly. Therefore it is important to remember the assumptions behind the model when using it. The lack of processes describing, for example, long-term changes of the base saturation in the soil, long-term changes of the nitrogen pool in the soil and sedimentation processes of nitrogen in a lake, restricts the use of the models.

One problem is the number of coefficients to be calibrated and the interdependence between them. The combination of two semi-empirical models, one for water balance and one for alkalinity or nitrogen loss, makes it difficult to generalize the model coefficients, and the model can easily become overparameterized. Today we can not use the alkalinity and the first version of nitrogen routines in unmeasured basins. The risk of overparameterization makes it very important to complement the model application with sensitivity analyses of the empirical coefficients.

One lesson learned from our modelling of nitrogen transport is that a complex model developed for a small-scale problem can not be easily transferred to larger basins of mixed land use. The estimation of coefficients becomes a major problem and equally good performance can be obtained by much simpler model formulations.

The modelling process consists of four phases comprising preparation of input data, model calibration, validation against independent measurements, and sensitivity analysis. If the agreement between the model and the measurements is good and the model seems to describe the processes in a reasonable way, the model can be used as a monitoring tool together with measurements. It can help us to detect changes hidden, for example, behind effects of weather-dependent fluctuations, if these changes are large enough. A good example of this application is found in paper I, where the use of the model as a prediction tool of effects of partial clear-cutting is also shown.

Modellers are sometimes accused of hiding hydrological and environmental processes in various black-box approaches (Klemeš 1986 and 1988, Falkenmark and Chapman 1989). It is true that many scientists and technologists have a blind faith in the use of models. Our knowledge of all processes involved in the hydrological cycle and interactions between them together with links between water and the environment around, is far from sufficient. Even more physically based models have to work with mathematical simplifications and approximations and can not be used without calibration today. In spite of this I hope to have shown with this thesis that the use of simpler models can be one way to describe and estimate effects of human influences and to distinguish them from natural variabilities. Good measurements are, however, still the most important prerequisite for a scientifically sound analysis of the problem.

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I

Modelling the Effects of Clearcutting on Runoff Examples from Central Sweden

by

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Modelling the Effects of Clearcutting on Runoff—Examples from Central Sweden

By Maja Brandt, Sten Bergström and Marie Gardelin

In order to study the effects of forest management on hydrology in central Sweden a hydrological model (PULSE model) was used both as a reference and as a forecasting tool. Three small basins were monitored before and after clearcutting. Increased total runoff and more pronounced peak flows were observed. The model was recalibrated to post-harvest conditions, and the two sets of model parameters were used to simulate the hydrological effects of hypothetical partial cutting in a large basin. During spring, the location of the cuttings affected peak flows because of nonsynchronous snowmelt and time delays in the basin. In autumn, the location of cuttings had very little effect on the peak flow. The total effect on peak flow of a 10 percent clearcut in a large basin was considered small compared to the effects of extreme weather conditions.

Two years after clearcutting at Aspåsen, Central Sweden. Photo: M. Brandt.

INTRODUCTION

The extreme autumn floods that occurred in Sweden during 1985–1987 have focused interest on the hydrological effects of clearcutting. In particular the flood that occurred in September 1985, which caused several dam failures in central Sweden, roused interest among the hydroelectric power companies. Local authorities in Sweden have also expressed concern about the possible connection between forest-management practices and flooding problems. Forest cover is of particular interest when designing structures susceptible to peak flows, such as dams and spillways and for river-valley development.

Worldwide interest in the hydrological effects of forest management is illustrated by the vast number of research projects in different countries. The problems involved, for example, are discussed in the works of Křeček and Zelený (Czechoslovakia) (1), Gupta (India) (2), Miţă (Romania) (3), Pearce et al. (New Zealand) (4), Plamondon and Ouellet (Canada) (5), and Ponce (USA) (6). Peak flows are particularly interesting for dam builders and have recently been discussed by, among others, Liu (China) (7).

Reviews of studies on changes in runoff are presented by Hibbert (8) and Bosch and Hewlett (9). Most studies use reference areas and regression analysis to calculate the changes. By reanalyzing the data from previous studies on clearcutting Harr (10) pointed to the difficulties involved in using regression analysis. His analysis showed that peak flows during snowmelt increased. This finding contradicts the findings of previous analyses.

The first study in Sweden was initiated in the 1920s (11). Later studies by Grip (12) and Rosén (13) used a forested reference area to determine changes in runoff. Runoff from both the reference and the study area was recorded before and after clearcutting, and the effect was calculated using regression analysis.

In general, studies indicate an increasing water yield following clearcutting of forest areas. This is commonly explained by reduced transpiration and evaporation of the water and/or snow that is normally intercepted by the trees. In addition, more intense snowmelts are generally reported and this agrees well with what is known about the difference between energy balance in open areas and in forests. The problem lies in quantifying the effects. The regression method requires that the basins are of similar type and that the changes in one basin do not affect the untouched re-

ference basin. For example, clearcutting a forest in one area can affect snowdrift, and thus water balance, in an adjacent basin. Furthermore, regression analysis requires a relatively stable climate. A regression equation derived from a period of relatively dry years may produce uncertain results, if the change in one basin is followed by wet years with higher runoff values.

The above problems may be overcome, at least to some extent, if a hydrological model is used. Instead of a gauged reference basin the modelled runoff is then used as a reference. Hydrological-model analysis is more capable of coping with climatological fluctuations, but it is also more susceptible to inconsistencies in climatological records. The regression analysis approach relies on reference basin data, the hydrological model approach relies on climatological records.

METHODS

The Hydrological Model

In this study the hydrological PULSE model for runoff simulation was used to study the effects of clearcutting on runoff.

The PULSE model (14) is a modification of the HBV runoff model (15, 16), which is being used in several countries for hydrological forecasting. The HBV model has been compared to other models of this type by the World Meteorological Organization (WMO) (17) with encouraging results, in spite of its relatively simple structure and limited data demands. The basic structure of the PULSE model is shown in Figure 1. The model takes into account daily totals of precipitation and mean air temperature together with monthly standard values of potential evapotranspiration. Runoff simulation involves three steps:

- snow accumulation and ablation;
- soil moisture accounting;
- generation of runoff and transformation of the hydrograph.

Precipitation is accumulated as snow if the air temperature is lower than a threshold value. A snowfall correction factor accounts for winter evaporation, gauge representativeness, and aerodynamic losses at the precipitation gauge. The melt routine of the model is essentially a degree-day approach according to the following equation:

$$m = C (T - T_0)$$

where : m = snowmelt (mm/24 h),
  C = degree-day melt factor,
  T = mean daily air temperature (°C),
  T_0 = threshold temperature.

A 10 percent liquid water-holding capacity of the snow has to be exceeded before any meltwater can leave the snowpack.

Although very simple, the degree-day approach has proved very efficient in basinwide modelling. The well controlled WMO intercomparison of snowmelt models (17) failed to identify any significant improvements when using more complex models in intermediate and large size basins.

Figure 1. Basic structure of the PULSE model.
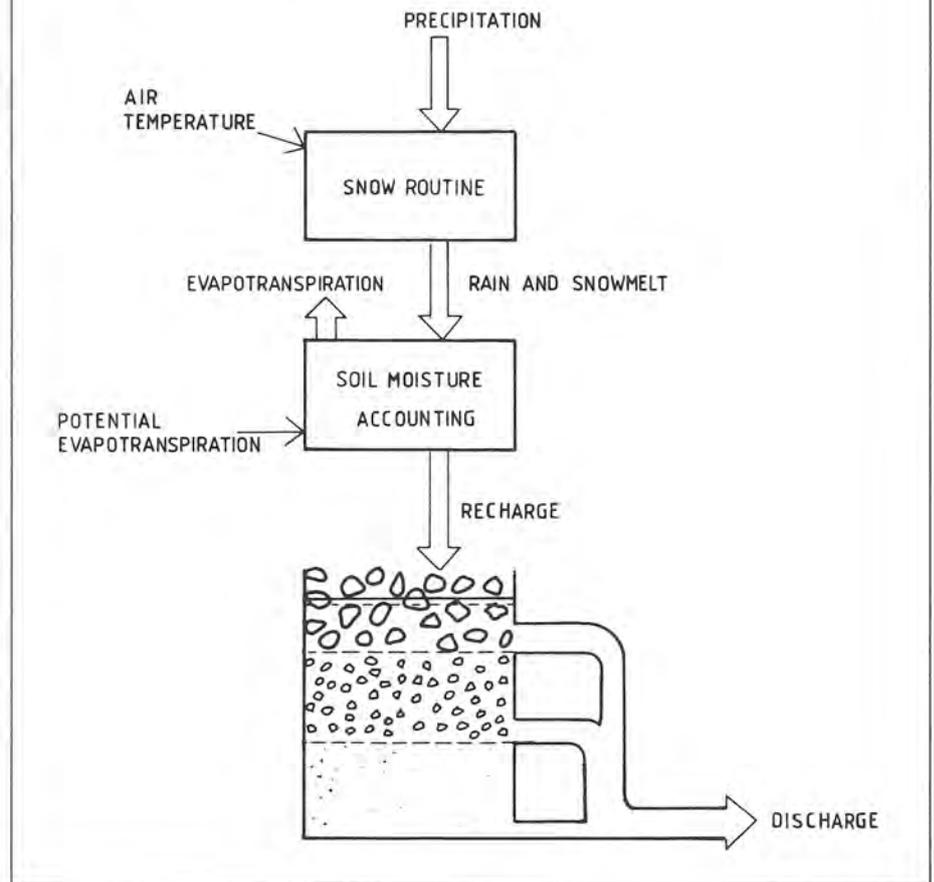
Figure 2. Principal components of the soil-moisture accounting procedure of the PULSE model.
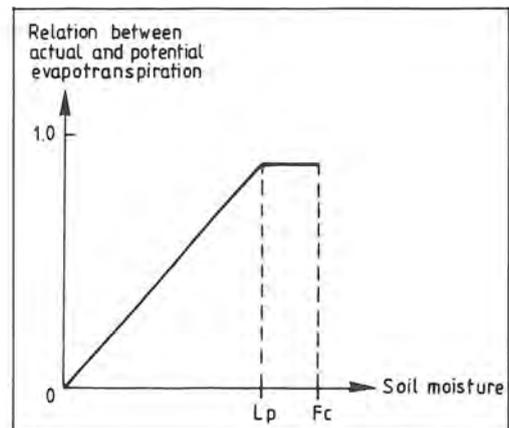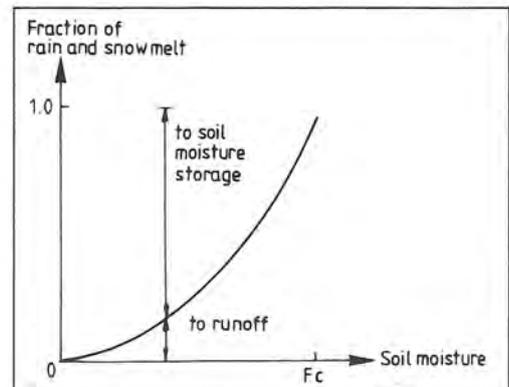
Figure 3. Substructure configuration of the distributed PULSE model.

Submodel II   III

Routing through lakes
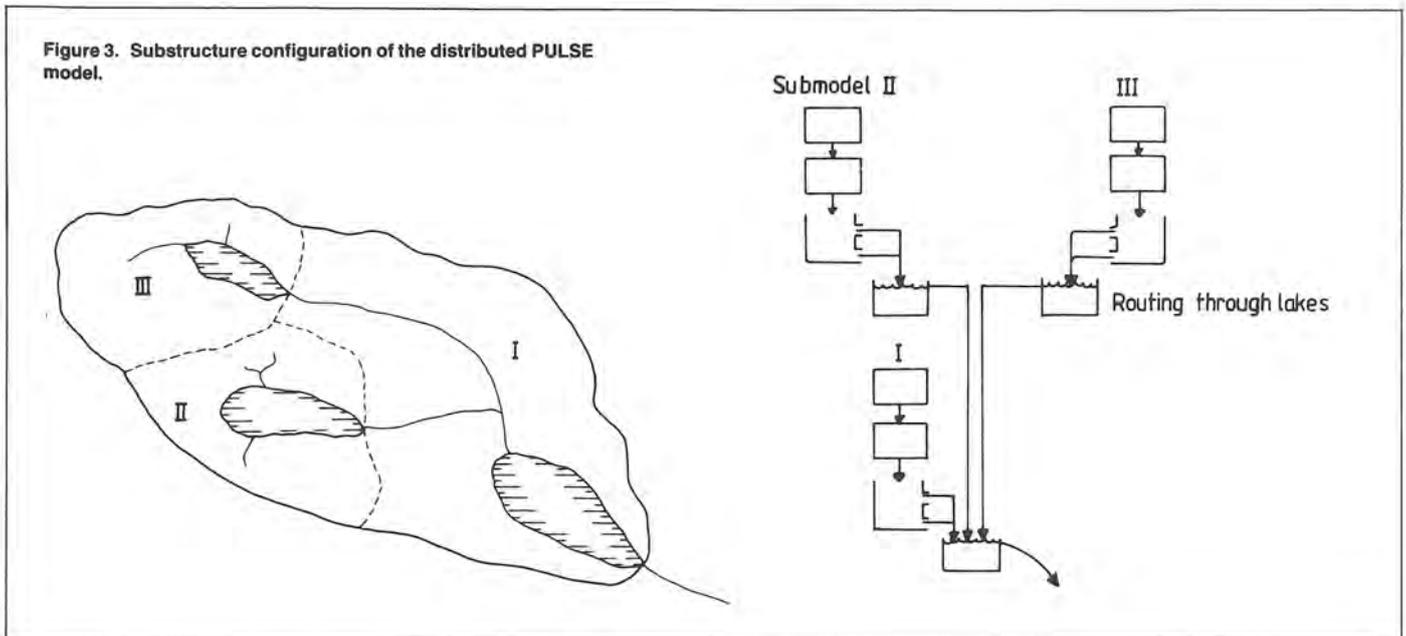
The soil-moisture accounting routine is summarized in Figure 2. This routine implies that the contribution to runoff from rain or snowmelt is small when the soil is dry and larger under wet conditions. Actual evaporation decreases as the soil dries out. A key parameter of the model is the Fc. The value of Fc represents the maximum amplitude of soil moisture storage in the model. If the model is run with low Fc-values, the actual evapotranspiration levels will drop more rapidly as the soil-moisture storage is emptied. Consequently, the overall soil-moisture deficit of the model will be less pronounced and simulated runoff will increase.

All excess water from the soil moisture zone is collected in the saturated zone. Water is drained at different levels with different recession coefficients, which account for rapid superficial runoff and deeper groundwater with slow drainage.

The routine implies that overland flow is not considered, unless the groundwater table is close to the surface. Among Swedish hydrologists this concept is generally accepted for till soils, see for example Rodhe (18).

When applying the PULSE model to larger basins a division into submodels is recommended. The variation of input variables, precipitation, and air temperature, with elevation above sea level, can thus be taken into account. The use of submodels is particularly important in basins with lakes.

If the model is structured into submodels, defined by the outlet points of the lake (Figure 3), the effects of the lakes on the shape of the hydrograph can be considered in a more physically correct way. This is not possible when a lumped structure is applied. This substructure of the PULSE model limits the demand on model calibration (19) and provides the model with a realistic time distribution of the flow contributions from different parts of the basin. The latter is a very important feature when analyzing the various consequences of forest-management practices.

DATA BASE

The location of the basins studied is shown in Figure 4. Runoff from the basins Kullarna (1.5 km²) and Sniptjärn (0.4 km²) has been gauged since 1977. In 1980, 70 percent (Kullarna) and 100 percent (Sniptjärn) of the areas were clearcut. The basins were drained in 1982 and 1983. At Aspåsen (0.16 km²), runoff measurements were initiated in 1979, and in the winter of 1982/83 85 percent of the area was clearcut.

Precipitation data for these three adjacent basins were collected from two

Figure 4. Geographical location of the studied basins.

Kassjöån

Kullarna, Sniptjärn
Aspåsen

STOCKHOLM

100 km

meteorological stations situated 30 kilometers south and 15 kilometers north of the basins. The data series were tested with double-mass analysis. No notable inconsistencies were observed, and the data were found fully acceptable for this study, although the stations were not situated close to the basins or at equivalent heights above sea level. Temperature data were collected from a station 30 kilometers south of the basins. In all cases monthly estimates of the potential evapotranspiration according to Eriksson (20) were applied.

Partial deforestation in large basins was studied using the representative basin Kassjöån (164 km²). Runoff data for the period 1975–1984, were chosen. Precipitation and temperature data were collected from a station 20 kilometers north of the basin.

RESULTS

Model Application to Small Basins

In the application of the model to the small basins Kullarna, Sniptjärn, and Aspåsen, the empirical model parameters were set by calibrating against observed runoff for the time period prior to clearcutting (three years). The effect of clearcutting could then be determined as the difference between simulated and recorded runoff when the model was run with the meteorological data measured after clearcutting.

Figure 5 illustrates the increase in runoff from the time of the clearcutting. The obvious effect is a quick drop in the accumulated difference curve (computed minus recorded runoff). The increase varied between 165 and 200 mm per year in the different basins. The greatest effects were observed in spring, when the increase was 50 to 120 mm during the spring flood. During the summer and autumn period, the increase varied between 75 and 95 mm.

When recalibrating the model to post-harvest conditions it was found to be

Figure 5. Runoff (Q) simulations for two small basins, illustrating the effect of clearcutting. Recorded hydrograph = thin line.

Computed hydrograph = blue line
Acc. diff. represents the cumulative difference between computations and observations.

KULLARNA (1.5 km²)

Q (l/s)

Acc diff (mm)

accumulated difference computed–recorded

70 % of the basin clearcut

SNIPTJÄRN (0.4 km²)

Q (l/s)

Acc diff (mm)

complete clearcutting of the basin

necessary to change the values of parameters for the snow routine and for the soil-moisture routine. The correction factor for snow accumulation was increased to account for increasing spring-flood runoff volumes. The degree-day factor was increased to account for a more intense snowmelt, and the threshold temperature was decreased to obtain an earlier start of the melting.

When trees are felled, transpiration and evaporation decrease. In the hydrological model this was accounted for by a decrease in the empirical parameter, Fc, in the soil-moisture routine. A summary of the most significant model parameters for conditions before and after harvest is given in Table 1.

The effects on the snowmelt parameters are in agreement with general experience from forested and open areas (21).

It is noticeable that the reduction in the amplitude of the soil-moisture storage, controlled by Fc, implies that the soil will become less important as a flood controlling factor after clearcutting.

The difference between pre-harvest and post-harvest conditions in the model simulations is shown in Figure 6. Runoff from

Table 1. Pre-harvest (I) and post-harvest (II) optimum-model parameters (calibrated).

	KULLARNA		SNIPTJÄRN		ASPÅSEN	
	I	II	I	II	I	II
Snowfall correction	0.85	1.15	0.8	1.10	1.00	1.30
Degree-day melt factor mm · (day · °C)$^{-1}$	2.25	3.50	2.25	3.50	3.00	5.50
Threshold air temperature (T_0) (°C)	0	−0.3	0.5	0	1.4	0.5
Soil-moisture amplitude (Fc) (mm)	200	50	150	40	200	50

Q (l/s)

SNIPTJÄRN basin (0.4 km²)

clearcut

forested

M   A   M   J   J   A   S   O   N   D   J   F
1982   Months   1983

Figure 6. Runoff (Q) simulations for the Sniptjärn basin (0.4 km²), illustrating the effect of clearcutting as described by the model with two different sets of parameters.

310

Figure 7. The Kassjöån drainage basin. Hypso-graphic curve and two alternative locations of a hypothetical clearcut covering 10 percent of the drainage basin are indicated.
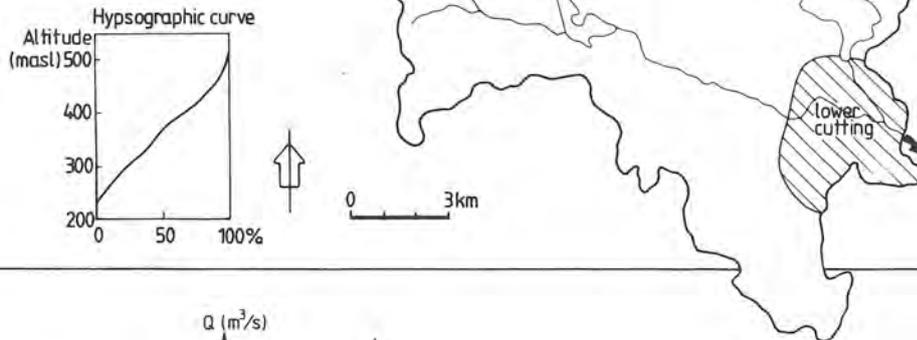

Figure 8. Examples of runoff (Q) simulations for the Kassjöån basin (164 km²), illustrating the effect of different locations of a clearcut covering 10 percent of the basin.
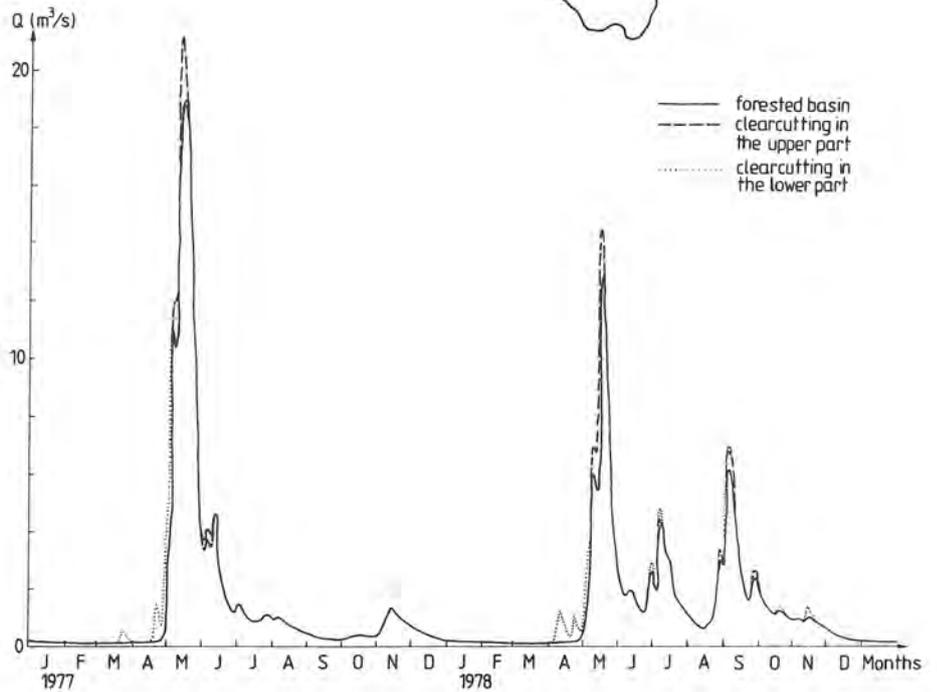
the clearcut basin is generally larger. Snowmelt starts earlier and is more intense, and summer and autumn conditions are significantly wetter due to a smaller soil-moisture deficit.

Model Applications to a Large Basin with Hypothetical Partial Clearcutting

The difference between the two parameter sets, one for forested and one for deforested conditions, formed the basis for the study on partial cuttings in large basins. For this study the drainage basin Kassjöån was used. The basin was divided into sub-basins making it possible to simulate the effect of hypothetical cuttings in different parts of the basin. Simulations were made for two different locations of a clearcut covering 10 percent of the basin area (Figure 7).

Examples of simulations of runoff with partial clearcuts are shown in Figure 8. The results indicate that the effect on runoff is relatively small and, for the spring, related to the location of the cuttings in the basin. The effect on peak flows during summer/autumn and spring is
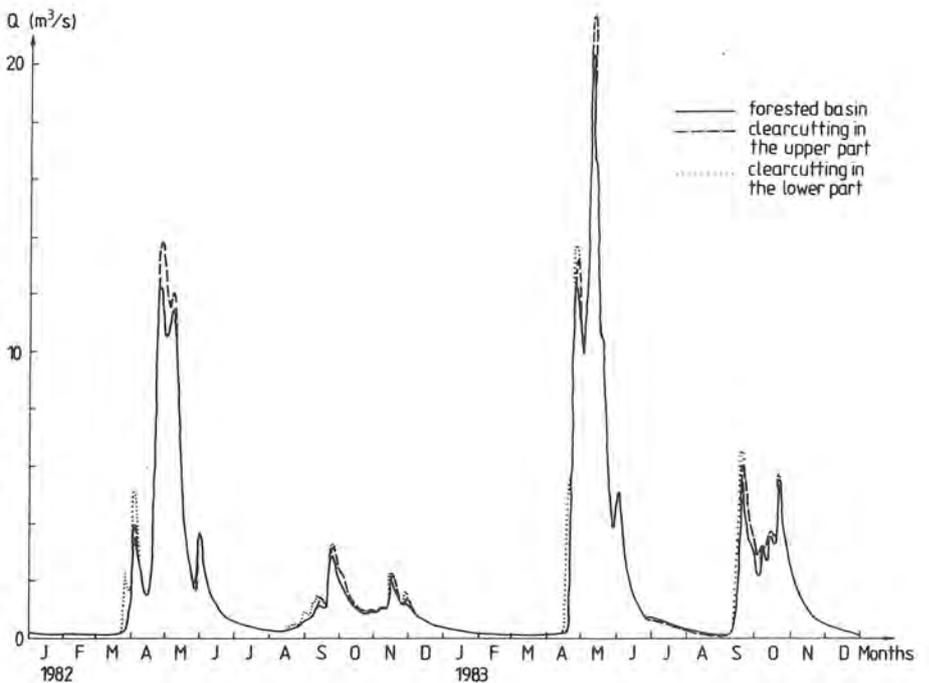
Figure 9. Simulated peak flows during summer and autumn in the Kassjöån basin (164 km²). Effects of a hypothetical clearcut covering 10 percent of the basin, located in the upper part (a) or in the lower part (b) of the basin (period: 1975–1984).
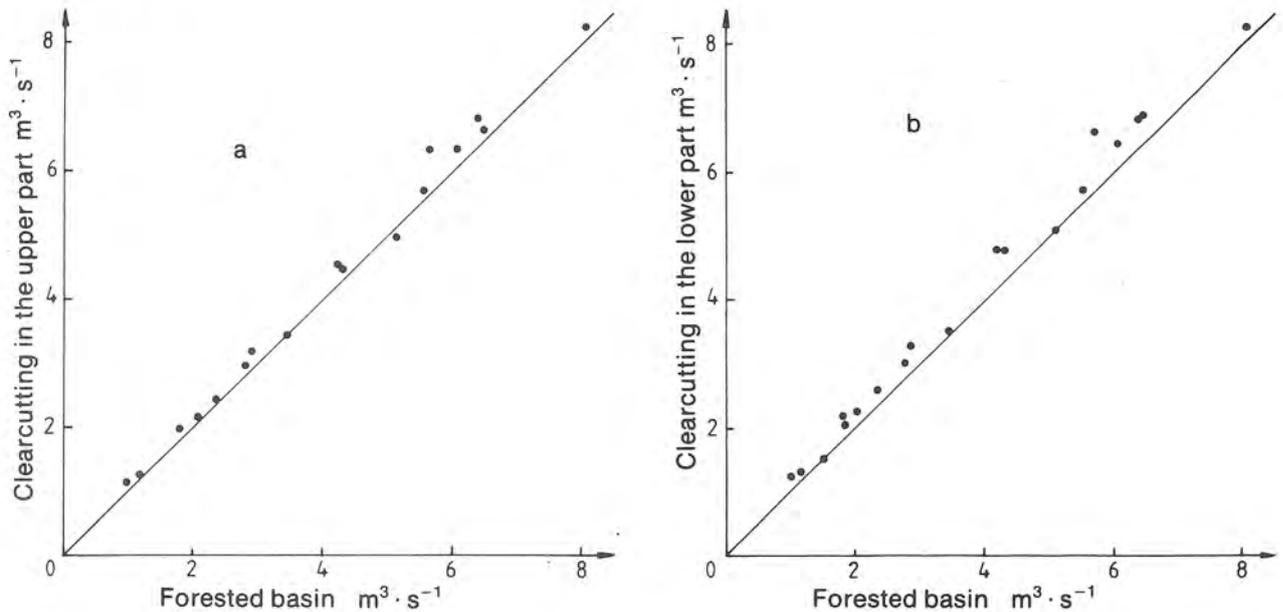
summarized in Figures 9 and 10, respectively.

Peak flows during summer and autumn increased by up to five percent as a result of clearcuts in both the studied locations. The importance of the location of the clearcut was considerable for spring flood. In the forested basin, spring flood normally starts in the lower parts. Clearcutting in the upper part of the basin will lead to a more intense snowmelt and an earlier spring flood in this part of the basin. Combined with melting in the lower part this resulted in a nine percent increase of peak flows during spring floods. On the other

hand, clearcutting near the outlet of the basin resulted in a more evenly distributed spring flood, due to nonsynchronous snowmelt and the time delays caused by routing of water through the lakes. Peak flows were unchanged or even lower.

DISCUSSION

The conclusion that complete clearcutting of a drainage basin will result in a considerable increase in runoff confirms a well-known fact. However, using a model as a reference base instead of a basin is new. The model avoids the problems of climatic

fluctuations that occur during the study period. Instead, the homogeneity of climatogical records becomes a very crucial factor, which has to be controlled. The model approach also constitutes a considerably less expensive method than maintaining a reference basin.

It is important to bear in mind that the study of the effects in the large basin is based on comparisons between simulations with hypothetical differences in forest cover. The manipulated small basins were only used to identify suitable model parameters for pre- and post-harvest conditions. In other words, the model is used


Extreme autumn floods in Central Sweden 1985 started a discussion on the hydrological effects of clearcutting. Photo: H. Sanner.

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Figure 10. Simulated peak flows during spring in the Kassjöån basin (164 km²). Effects of a hypothetical clearcut covering 10 percent of the basin, located in the upper part (a) or in the lower part (b) of the basin (period: 1975–1984).
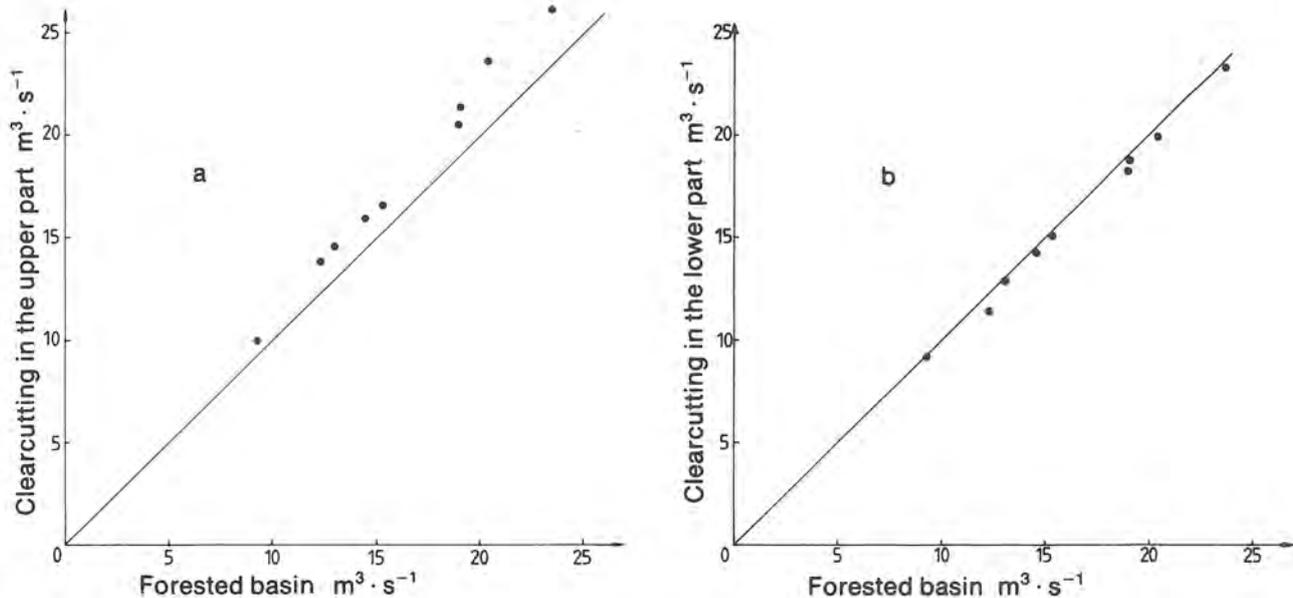
as an instrument to spatially and temporally extrapolate known results from a small research site to an integral part of a large basin. Of particular importance is the distribution of snowmelt and damping of contributions from sub-basins by the routing through lakes.

The relatively minor effects of forest management practices are mainly due to the fact that 10 percent is a relatively small, but probably realistic, fraction of a basin of this size. If the full heterogeneity of the system is taken into account, the results may be in agreement with intuition, but the model helps us to quantify the integral effects of different geographical locations of clearcut areas.

The study is based on a relatively crude distinction between forested and clearcut areas, which limits its generality. In any basin there is, of course, a range of tree sizes and a water demand relative to tree age. Forest density may also be changing due to more efficient forest management practices. Another factor, which can disturb the analysis, is that clearcutting is often followed by increased drainage to protect the area and increase production.

Nevertheless, we feel that the model approach to the analysis of hydrological effects of partial clearcutting provides useful data on the order of magnitude of these effects. The results show that forest management practices can have a strong local influence on flood risks, especially in small streams in areas where the percentage of deforestation is high. Considering the relatively small total fraction of clearcuttings in large river basins, it is not realistic to assume that forest management practices are the main cause of high floods in Sweden's main river systems during recent years. These floods can be satisfactorily explained by extreme climate and antecedent soil moisture conditions.

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II a

Environmental Impacts of an Old Mine Tailings Deposit Hydrochemical and Hydrological Background

by

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Environmental Impacts of an Old Mine Tailings Deposit – Hydrochemical and Hydrological Background

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The release and distribution of metals as well as acid from a mine tailings deposit into a stream have been studied, particularly the distribution and neutralization of acids, the distribution and chemical speciation of metals and metal adsorption and precipitation phenomena. The Bersbo mine, some 250 km SSW of Stockholm, was selected for the study. An over-view is given of the local geology, hydrology and hydrochemistry as well as the research program (field measurements, sampling, analytical procedures and modelling). The databases (chemical and hydrological) generated within the project are outlined. The paper serves as an introduction to a series of three papers based on field data from the same area.

Introduction

The spreading of hazardous elements in the ecosystem, both from natural and antropogenic sources, has become a major environmental problem (Förstner and Wittman 1980; Salomons and Förstner 1984).

Despite the extensive amount of published data related to the releases of metal contaminants it is not generally possible to quantitatively model the transport and redistribution of metals in the aqueous environment. Knowledge of chemical speciation, interactions with geologic phases, transportation and natural background levels is rather limited.

The general aim of the present project is to gain information on the release and redistribution of acid and metals (particularly Al, Fe, Mn, Cu, Zn, Cd and Pb)

from a point source represented by a mine tailings deposit.

The following areas are studied in detail:

1) Distribution and neutralization of acid; integrated hydrological and chemical modelling.
2) Distribution and chemical speciation of metals
3) Metal adsorption and precipitation phenomena.

The Bersbo mine in the municipality of Åtvidaberg, some 250 km SSW of Stockholm (16°3'E, 58°16'N) was selected as a suitable point-source (Törnebom 1885; Sundius 1921; Sandegren *et al.* 1924; Tegengren 1924; Söderbäck 1974; Hellström *et al.* 1983). The release of metals from the mine tailings as well as biological uptake have previously been studied in the area (Jacks 1976; Qvarfort 1977; Karlqvist and Qvarfort 1979; Noorlind *et al.* 1980; Börkén 1982; Adelsvärd *et al.* 1983; Lundholm and Andersson 1985). Some preliminary studies by the present project group have also been reported (Karlsson and Sandén 1984; Allard *et al.* 1985; Allard *et al.* 1986).

The Bersbo Area

Mining History

Mining of copper in the Åtvidaberg region started already in the 14th century, possibly even earlier, particularly in the Närstad and Bersbo areas (Törnebom 1885; Tegengren 1924; Sandegren *et al.* 1924) cf. Fig. 1. Rational mining was undertaken from ca. 1760 to 1902 with a peak production during 1855-1870. During this period the Åtvidaberg mining area was the largest copper producer in Sweden.

The mining at Bersbo has resulted in a continuous system of shafts and tunnels down to a depth of 465 m.

The mine is now filled with water. Visible remains of the mining is the large deposit of tailings, covering some 0.2 km^2 and containing ca. 300,000 m^3 of waste rock.

Geology

The oldest rocks in the Åtvidaberg area are of Precambrian age, and supra crustal origin consisting of rocks as amphibolites, leptites and mica schists (Tegengren 1934). Intrusions of younger granitic rocks (oligoclase rich gray gneissic granites as well as intermediate and alkaline red gneissic granites) are frequent, just as formations of metamorphic gabbro and diabases. The Bersbo mines are situated in a zone of leptitic rocks, however with a large intrusive amphibolite body containing hornblende, augite and calcium rich plagioclase as well as some younger pegmatite rich granitic intrusions. The ore veins, which contain pyrite, magnetic chalcopyrite and occasionally sphaelerite and galena are found in the leptite at some distance
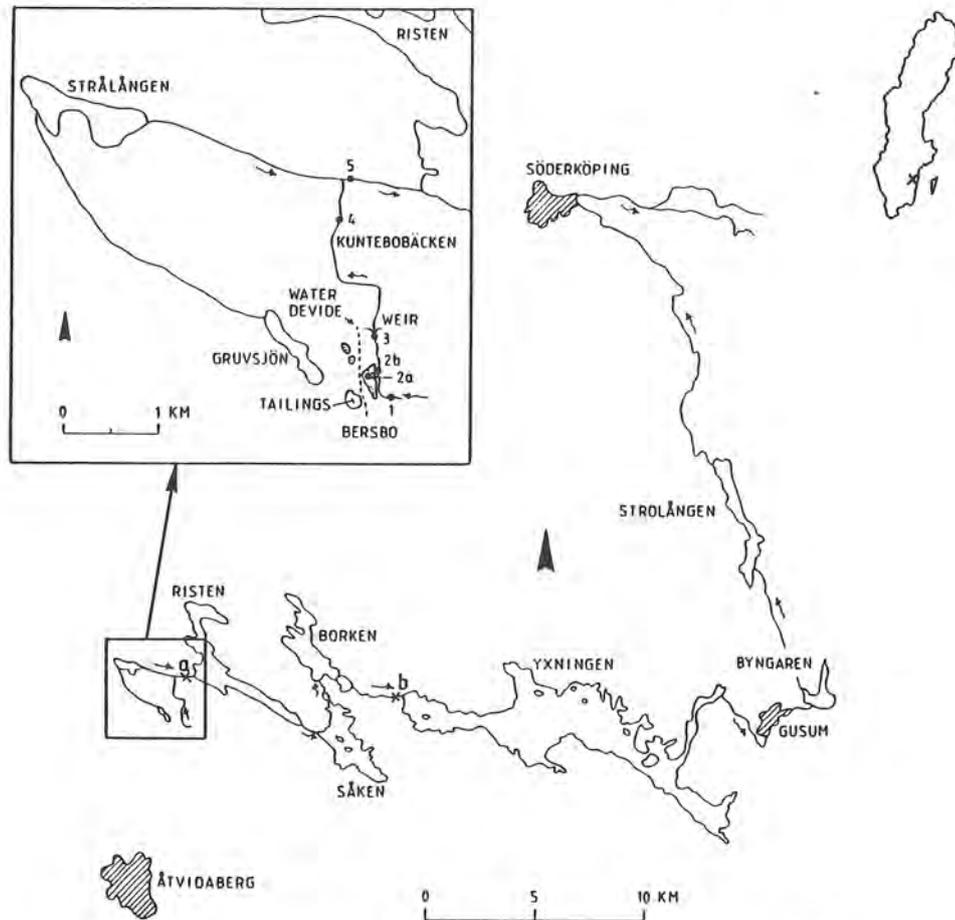
Fig. 1. The Bersbo area with sampling locations in the eastern stream.

from the amphibolite/leptite interphase, or associated with the amphibolite. Thus, they form zones in the leptite that are parallel to the large amphibolite intrusion. All these rock types are found among the tailings.

The concentrations of metals in the ore have been estimated to be 0.5-3% copper, 1-3% zinc, 1% lead and 20% iron (as well as 25% sulphur), based on the available data on the total production (Karlqvist and Qvarfort 1979).

The bedrock in the vicinities of Bersbo is covered with till and, in lower parts, clay. The clay zone has a thickness of 2-4 m underneath the tailings deposit. It is probable that this clay contains substantial amounts of calcareous shells.

Hydrology

A ridge through the deposit area will act as an effective water divide. Surface water from the precipitation and percolating water from the deposit will be drained

281

Fig. 2. Monthly precipitation, in mm, at the Bersbo area and water flow in the eastern stream.

essentially in westerly direction (to the Lake Gruvsjön, altitude 80.5 m above sea level) and in easterly direction (to the Kuntebobäcken, originating in a bog area at 97.1 m above sea level) Fig. 1. The westerly drainage will pass through Lake Strålången and meet the easterly drainage at Missmyra, Fig. 1, and further into Lake Risten (at 62.4 m above sea level) and into a series of lakes (Lake Såken, Lake Borken, Lake Yxningen).

The Bersbo area is largely covered with coniferous forests, with some minor bogs upstream of the deposit, and a few cultivated fields along the stream.

The hydrology of the unaffected part of the Bersbo area is typical for the southeast of Sweden. The average precipitation is estimated to 650 mm/year, the evapotranspiration is 450 mm/year, and 200 mm/year forms the runoff. The variability of the water balance components is high, in particular during winter conditions. Some winters are stable with a pronounced snowmelt flood in spring, while others are mild with frequent episodes of rain, snowfall and melt. The water balance of the study period is summarized in Fig. 2.

The till soil in the area can be assumed to be pervious enough to prevent any surface runoff, unless it reaches full saturation. This is probably also true when the soil is frozen, even if some extreme situations with surface runoff caused by icing on the ground may occur locally. The hydrological pathways in a dry and a wet situation are schematically illustrated in Fig. 3. The combination of superficial groundwater in the more acid upper horizon of the soil during a flood situation is responsible for the temporary drop in alkalinity and pH, while it recovers during dry periods when contributions from deeper groundwater dominate the flow.

The area is rich in lakes. Further down the river system these are dominating the hydrological and hydrochemical response. This is illustrated by Fig. 4, which shows a model simulation of the hydrograph at two points, one upstream lake Risten and one downstream lake Borken, by the PULSE-model (Bergström *et al.* 1985) with a subroutine for routing of the flow through lakes attached to it.

282
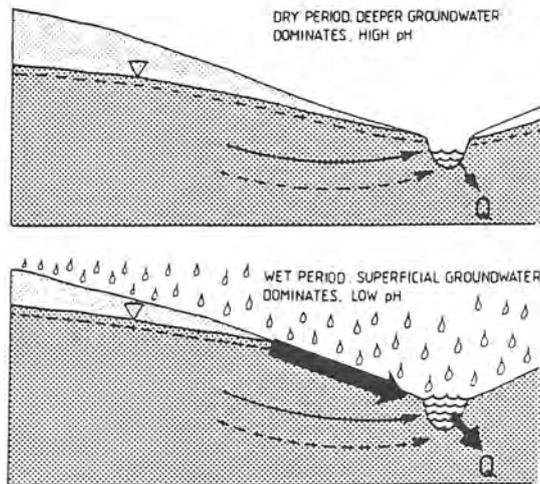
Fig. 3. Schematic picture of the hydrological pathways in the unaffected forested parts of the Bersbo area.

The water balance of the deposit differs from that of the surrounding area, mainly because of lack of vegetation but also due to the coarse size fraction of the waste material. Evaporation is low and transpiration is almost negligible. The hydrological response will therefore be relatively fast even after long dry periods. The deposit under study contains ca. 300,000 m^3 of waste rock and covers ca. 0.2 km^2. Precipitation readily percolates through it since coarse size fractions dominate. Hydraulic conductivity has been estimated to 10^{-3} m/s but the leachate is to some extent retained at the bottom of the deposit before it enters the stream. The drainage areas to the respective sampling sites are given in Fig. 5. Details of the water balance of the area are further discussed by Brandt *et al.* (1987).

Sampling and Chemical Analysis

Sampling in the eastern stream began in Sept. 1983 at the sites given in Fig. 1. Samples have been collected weekly and combined with automatic samplers (1-4 samples per day, sites 3 and 4) to allow sampling under rapid changes in flow
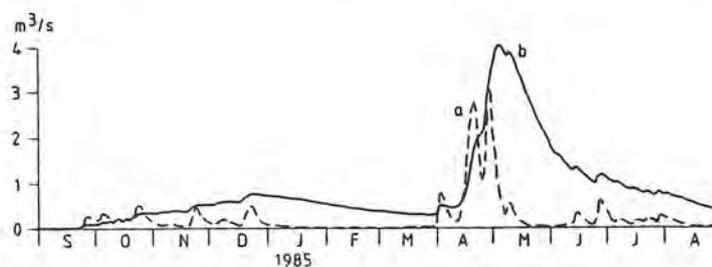
Fig. 4. Model simulated hydrographs at the inlet to the first large lake (a) in the water system downstream the Bersbo area and the outlet of a large lake further down in the system (b), cf Fig. 1.

283
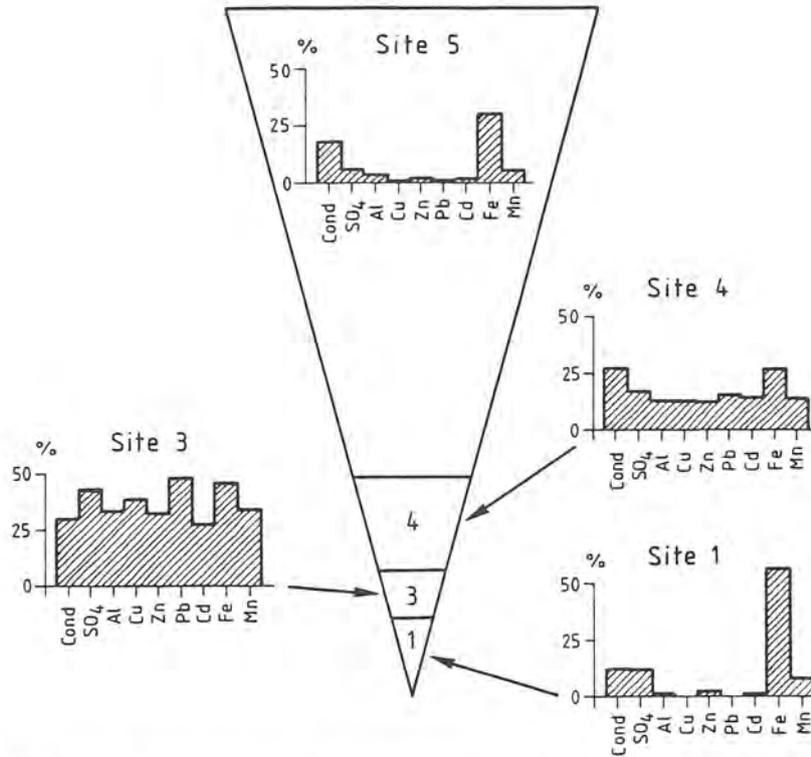
Fig. 5. Schematic presentation of the drainage areas to the respective sampling sites in the eastern drainage. Average values of chemical constitutents are given as percentage of those corresponding to the leachate. The triangular areas are proportional to the drainage areas to the five sites, respectively.

conditions. Water flow has been continuously recorded at the weir at site 3 and the daily precipitation at the gauge located close to the water divide (Fig. 1). The sampling site 1 represented »background« conditions of surface water in the area. The leachate was characterized by sampling in the open water table at the clay interface (2a) and just as it entered into the stream (2b). Sampling sites 3 and 4, at 400 m and 1400 m from the outlet, were chosen to represent various input of leachates and subsequent mixing with groundwater (cf. Table 1). The most remote sampling site (5) represented the junction of the two major streams, ca 2,100 m from the deposit with regard to the eastern stream. Samples were also collected from 6 groundwater wells installed perpendicular to the eastern stream adjacent to the deposit. These samples represented water in contact with or just below the illitic clay zone, at 4-6 m below the soil surface.

Sampling and sample preparations were made in acid washed (ca 2 M HNO$_3$ + 1 M HCl) plastic (polyethylene or polypropylene) vessels. Samples for determinations of dissolved oxygen were collected in a way as to minimize contact with the

284

Table 1 – Composition of the Water in the Easterly Stream (Kuntebobäcken), Sept. 1983 – Dec. 1986; weekly samples (Locations according to Fig. 1).

Constituent[a]	Site 2b (n=65)				Site 3 (n=21)				Site 4 (n=86)				Site 5 (n=90)			
	Mean	S.D.	Min	Max	Mean	S.D.	Min	Max	Mean	S.D.	Min	Max	Mean	S.D.	Min	Max
Na	249	68	128	374	281	60	96	450	400	126	118	827	337	67	169	501
K	104	24	48	166	84	44	31	227	98	53	35	291	115	29	66	257
Mg	1270	496	115	2068	857	1342	32	6244	680	950	40	4800	240	56	118	392
Ca	437	160	137	885	356	184	44	1324	520	240	67	1204	493	142	149	1089
Al	1760	735	7	3543	481	567	0.7	2539	175	222	2	1510	52	35	7	220
Fe	90	68	17	445	50	42	13	305	23	14	1	83	28	16	11	103
Mn	92	33	23	153	29	29	3	125	13	11	3	87	4	4	0.7	19
Cu	151	49	15	254	49	54	3	250	19	25	0.5	173	1	0.9	0.2	5
Zn	992	438	2	2031	283	361	18	2431	140	152	3	936	16	12	0.5	67
Cd	1.742	0.426	0.507	2767	0.394	0.364	0.018	1.432	0.210	0.165	0.009	0.889	0.021	0.022	0.001	0.106
Pb	0.170	0.057	0.063	355	0.047	0.052	0.001	0.265	0.009	0.015	0.001	0.126	0.002	0.003	0.001	15
Cl^{-}	62	16	25	113	231	70	68	415	378	182	99	1117	327	100	132	646
SO_4^{2-}	6220	2010	1700	10300	2130	2000	300	7940	1257	980	243	5752	336	181	160	1160
CO_{3tot}^{2-}	0	–	–	–	0.11	0.06	0	0.22	0.42	0.44	0.01	2.15	0.94	0.40	0.24	2.45
O_2[b]	10.2	2.2	7.1	15.8	7.6	2.3	0.6	10.8	10.4	1.9	2.0	14.3	6.6	2.6	0.7	11.1
pH	3.40	0.21	2.93	3.91	4.42	0.60	3.41	5.88	5.82	0.73	4.20	7.3	6.75	0.24	6.22	7.28

a Concentrations in μ mol/l

b Concentrations in mg/l

Table 2 – Analytical methods

Parameter	Analytical method/equipment
pH	Radiometer PHM84, glass electrode (G2040C) and calomel reference electrode (K4040)
Alkalinity	Titration with 0.02 M HCl to pH 5.4 using the electrodes above under continous flux of carbonate free N_2
Electric conductivity	Radiometer CDM83 instrument with electrode CDC104 automatic temperature compensation to 25°C
Chloride	Precipitation titration in KNO_3(0.50M) + HNO_3(0.0105M) with Ag^+(0.01M) and indicating the inflexion point with the $Ag/AgCl/HgSO_4$ electrode pair
Sulphate	Indirectly by ionexchangeable cations until Dec. 84 and thereafter directly by the Ba-methylthymole blue method (Flow injection technique)
Dissolved oxygen	Winkler method
Metal constituents	Atomic absorption spectrophotometry (Perkin-Elmer 5000/Zeeman instrument). Direct calibration of the instrument (flame or furnace) after controll of non-specific absorbance at the selected wavelengths.
Phase separation	Pressure filtration using 0.47 mm($\emptyset$) polycarbonate filters (Nuclepor. Corp.), pore size 0.40 μm.

atmosphere. Water samples were always collected from the center of the stream to avoid resuspending stationary sediment phases.

The analytical program included major constitutents of the waters (Na, K, Mg, Ca, Cl^-, SO_4^{2-}, O_2(aq) and pH) as well as trace metals (Al, Fe, Mn, Cu, Zn, Cd and Pb). Separation of solids and the operationally defined soluble phases was made by pressure filtration through a polycarbonate filter. Analytical procedures are summarized in Table 2.

Phase separation and chemical analysis of non-metal constituents were usually completed within 8 hours after the samples had been collected. Metal analysis was performed on acidified samples as well as on acidified filtrates and were completed within 48 h. The use of automatic samplers did not allow for accurate determinations of the solid and soluble phases of trace metals since precipitation obviously occurred already at moderate pH during the storing in the field. Only major constituents were determined in such samples.

Table 3 – Composition of the water in the Bersbo area.

Constituent[a]	Groundwater (n=18)		Leachate (site 2a) (n=80)		Surface water[b] (site 1) (n=120)	
	Mean	S.D.	Mean	S.D.	Mean	S.D.
Na	601	117	290	90	254	62
K	165	30	118	34	45	32
Mg	2115	1600	1881	1424	355	711
Ca	1900	240	512	288	207	149
Al	16	36	1875	946	37	21
Fe	36	50	160	240	57	73
Mn	23	5	111	44	7	16
Cu	0.3	0.5	132	67	0.6	1
Zn	3.9	5	1066	550	8	17
Cd	c	c	1.790	0.824	0.028	0.051
Pb	c	c	0.130	0.066	0.003	0.003
Cl^-	145	157	62	23	204	83
SO_4^{2-}	3	2	7701	3134	367	354
CO_{3tot}^{2-d}	1.58	0.28	0	–	0.23	0.36
O_2^e	–	–	8.4	2.00	4.9	2.9
pH	7.18	0.18	3.42	0.20	5.33	0.41

a Concentrations in µmol/l
b Cf. Figs. 1 and 2
c Less than 3 nmol/l
d Concentrations in mequ/l (not determined when pH < 5.4)
e Concentrations in mg/l

General Hydrochemistry

The shallow groundwater in the area is of Ca^{2+}-SO_4^{2-}-CO_3^{2-}-type (Table 3) with an alkalinity of 2.5-3 meq/l and pH 7.1-7.5 (max. 7.8). The total salinity is fairly high (< 500 mg/l) but without any obvious marine origin (low Na^+-Cl^--concentrations). The alkalinity/pH-properties reflect the presence of calcium carbonate (as shells) in the ground. There is no significant contamination of the groundwater with effluents from the deposit at the location of the groundwater sampling, some 20 m east of the deposit at 4-6 m below the soil surface. Evidently, the clay layer acts as an effective barrier that prevents direct exchanges between leachates and groundwater.

The surface waters close to the deposit typically have total salinities of 70-75 mg/l, Tables 1 and 3, with Ca^{2+}-Na^+ as major cations, sulphate as dominating anion and pH around 4.9-5.7. Water coming from bog areas has large contents of organics (humic and fulvic acids) which can be as high as 50 mg/l.

Oxygenated water percolating through the tailings will lead to a severe weathering of the sulphide minerals (Stumm and Morgan 1981). The final products would eventually be metal ions, sulphate and hydrogen ions (a yield of totally 2-4 moles of

H^+ and 1-2 moles of SO_4^{2-} per mole of Fe, Cu, etc.). The released hydrogen ions could partly be consumed in the weathering of amphiboles, micas and feldspars of the leptite-amphibolite host rock. The released metals may form sparingly soluble compounds, e.g. hydroxides, hydroxysulphates, etc.

Estimated yearly maximum amounts of metals that are leached from the deposit in the western and eastern directions, respectively, are ca. 3,000 kg aluminium, 6,000 kg zink, 700 kg copper, 25 kg cadmium and 2 kg lead, based on data from 1983-86 (in the Kuntebobäcken and in the outlet from Lake Gruvsjön). The corresponding produced amount of sulphuric acid is of the order 100,000 kg. There is however large variations in the transported amounts between different years, related to hydrological factors. Annually transported metal loadings 1983-86 for the easterly drainage are 1,300 kg aluminum, 1,600 kg zink, 340 kg copper, 6 kg cadmium and 1,5 kg lead.

The leachates from the tailings are mixed with precipitation and alkaline groundwater in the system of streams that are draining the deposit. Thus, several different processes will lead to a decreacing acidity and depletion of metals from the aqueous phase in the streams:

- Dilution with surface water and groundwater
- Neutralization with alkaline groundwater
- Precipitation of sparingly soluble metal hydroxides, notably of iron and aluminium
- Coprecipitation or adsorption of dissolved hydrolysable trace metals
- Adsorption of trace components on any exposed surfaces
- Sedimentation of metal rich particle fractions.

The result, in terms of gradual changes in surface water chemistry, is illustrated in Table 1 (cf. Table 3). The variations caused by the hydrological conditions and the subsequent chemical responses are further discussed in Sandén *et al.* (1987).

Perspectives on Metal Distribution in the Area

The present database consists of time series with field measurements of hydrological parameters (e.g. water flow at site 3 and precipitation) and the chemical composition of the stream water with its suspended matter at the respective locations. During the three years of sampling the variations in flow conditions as well as chemical conditions have been large (cf. Tables 1 and 3). Changes in chemical characteristics are, however, not confined to any specific location but rather related to seasonal variations as well as more rapid responses to single precipitation events (cf. Fig. 2). Thus, the area can serve as an excellent model system. Effects of pH on metal speciation and distribution, solid phase formation, sorption, sedimentation etc. are demonstrated. The remarkable buffering capacity of the system is

288

briefly discussed since it causes a rapid increase in pH with increasing distance from the deposit. Data from the field site are presently used for modelling of water flow in the stream and the impacts of hydrological conditions on pH and alkalinity (Brandt *et al.* 1987). These parameters are likely to influence the chemical state of metals in the stream, especially those originating from the leachates (Sandén *et al.* 1987). The partitioning of metals between suspended solid phases as a result of changes in chemical conditions (e.g. pH, saturation state and distribution of dissolved species) is discussed in Karlsson *et al.* (1987).

The present series of papers consider phenomena related to the eastern drainage of the deposit. Measurements have recently started to include the more slowly reacting western drainage as well as the lake system downstream the area.

Acknowledgements

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II b

Environmental Impacts of an Old Mine Tailings Deposit Modelling of Water Balance, Alkalinity and pH

by

Brandt, M., Bergström, S., and Sandén, P.

Environmental Impacts of an Old Mine Tailings Deposit – Modelling of Water Balance, Alkalinity and pH

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A diversity of hydrological and hydrochemical data is the foundation for an integrated modelling study of a leaching tailing deposit at Bersbo in southern Sweden. A distributed conceptual water balance model system combined with empirical hydrochemical subroutines are used to describe the variation pattern in runoff, alkalinity and pH of the outlet of the drainage basin. Examples of the sensitivity of the model output to perturbations of some of the model parameters are shown.

Introduction

The hydrological and hydrochemical observations at Bersbo, as described in a companion paper by Allard *et al.* (1987), give opportunity to develop and test integrated conceptual water balance and hydrochemical models. Problems associated with the interaction between water balance components of the model and hydrochemical processes can partly be overcome by careful use of the diversity of data from various locations in the area.

The total size of the investigated basin is 0.9 km², of which the deposit covers approximately 0.08 km². For the modelling exercise, the area is divided into four subbasins, as illustrated in Fig. 1. Three of these basins consist of unaffected forested areas, while one is the tailing deposit itself. Thus, each one of the sub-basins can be modelled separately and verified against data, representing each individual sub-basin.
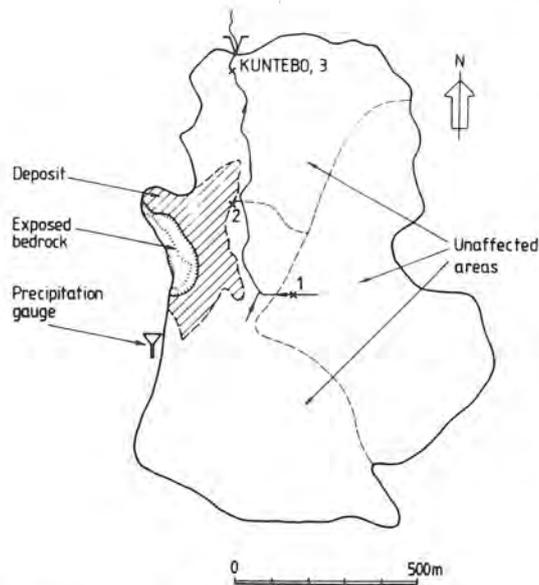
Fig. 1.
Subdivision of the study area for the
model application. 1, 2 and Kuntebo,
3, represent sampling points.

Steps in the Modelling Exercise

The aim of the modelling exercise is to describe the variation pattern in runoff,
alkalinity and pH at the Kuntebo runoff station. This requires the following four
submodels:

- A model for the water balance of the deposit.
- A model for the water balance of surrounding forested areas.
- A model for the quality of leaching water from the deposit.
- A model for the quality of water from surrounding forested areas.

Once these submodels are at hand, the integrated output at Kuntebo can be calcu-
lated by a simple mixing procedure. The four submodels are not established or
calibrated entirely independently. For example, the modelling of the water balance
of the deposit is made simultaneously with the calibration of its hydrochemical
counterpart.

Data Base

The data base to our disposal is described in detail by Allard *et al.* (1987). It can be
summarized as follows:

- Daily totals of runoff and hydrochemistry of variable frequency at Kuntebo.
- Hydrochemistry of variable sampling frequency from the area upstream of the
  waste dump (site 1 in Fig. 1).

292

- Hydrochemistry of variable sampling frequency of the small streams, which leave the waste dump (site 2 in Fig. 1).
- Daily totals of precipitation on top of the waste dump.
- Daily mean air temperature from Malmslätt 35 km northwest of the area.
- Estimates of seasonal monthly means of potential evapotranspiration according to Penman's formula (Eriksson 1981).

The chemical observations consist mainly of weekly analyses of pH, conductivity, oxygen, sodium, potassium, calcium, magnesium, chloride, sulfate, carbonate, aluminium, iron, manganese, copper, zink, cadmium and lead. Periodically automatic sampling has been used with a frequency of 2-3 samples per day. Unfortunately, the total length of the hydrological and hydrochemical data base is too short to permit a split sample model application. This means that there is an obvious risk for overfit of the model. However, previous experience from model applications of this type (see for example Bergström *et al.* 1985) leaves us with some confidence, in particular as the number of model parameters is small and the model is calibrated on both hydrological and hydrochemical time series simultaneously. The total information to our disposal for model calibration is therefore substantial although rather variable for the different sub-basins.
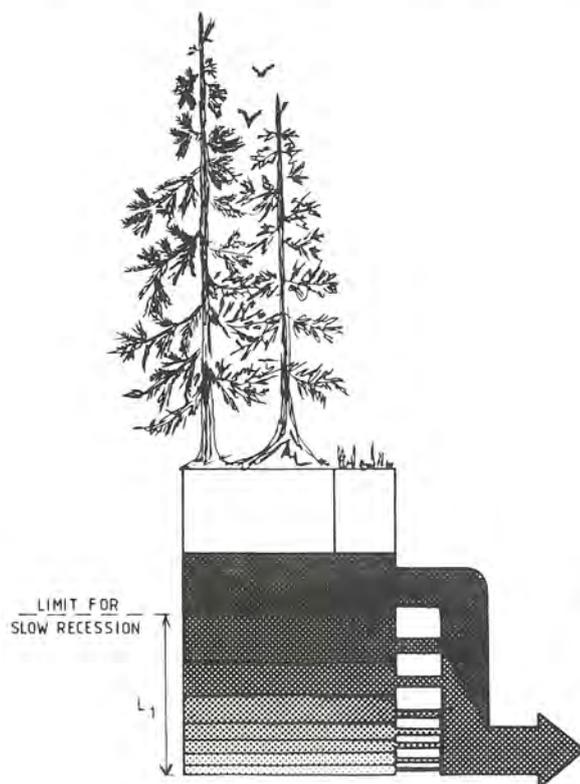
Fig. 2.
Principal sketch of the drainage of an aquifer in the PULSE model.

293
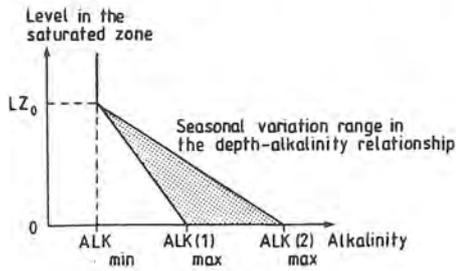
Fig. 3.
Principal relationship between the depth
and alkalinity/acidity in the saturated zone
of the PULSE model.

Basic Model System

The foundation of this modelling study is the PULSE model system described by
Bergström *et al.* (1985), which is a development of a conceptual runoff model
(Bergström and Forsman 1973, and Bergström 1975). The PULSE model integra-
tes hydrology and hydrochemistry in a simple manner. The basic idea is that the
hydrological pathways are governing short term variations in alkalinity and pH in
running water. The fundamental concept is that the alkalinity of the water is
determined by the groundwater level in the aquifer which drains into the river. This
relation is shown in Fig. 2, where also a seasonal variation in the depth/alkalinity
relationship is indicated with higher alkalinity in summer than in winter for a given
groundwater level. The hydrochemical subroutine is normally calibrated after
calibration of the water balance of the model and requires another four model
parameters.

The PULSE model can be distributed into submodels, which makes it feasible
for application to the Bersbo area with its different land use.

One relatively unique feature of the PULSE model is that each pulse of snow-
melt or rainfall is traced separately through the model without assumption of
complete mixing, as illustrated in Fig. 3. This makes it particularly applicable to
problems, where the level in an aquifer is a significant factor. The procedures
illustrated in Figs. 2 and 3 give us a continous model output of the water balance
and the climate-induced variability of the alkalinity of the water. The alkalinity is
finally transformed to pH with a relationship according to Fig. 4.
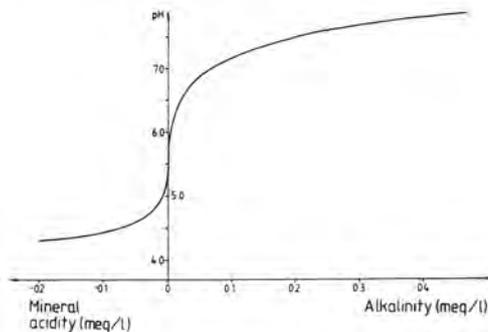
Fig. 4.
Relationship between alkalinity/acidity
and pH in the PULSE model.
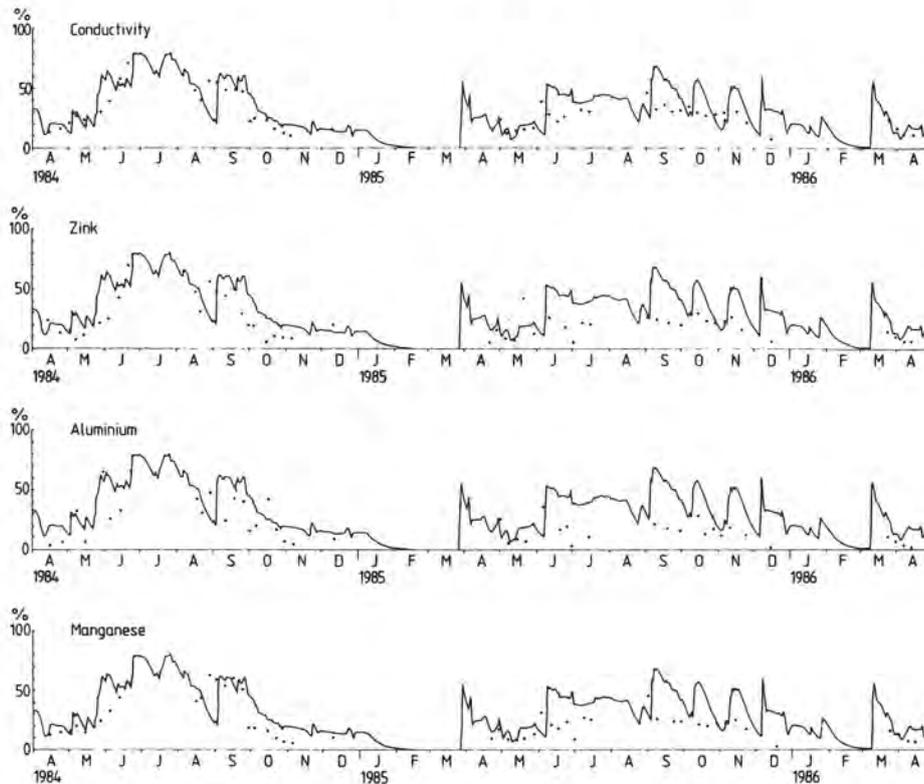
Fig. 5. Indirect comparison between the modelled fraction of runoff from the deposit and corresponding estimates based on concentration analyses at three points.

Water Balance of the Deposit

The water balance of the tailing deposit was modelled by one submodel. The effect of the soil moisture accounting procedure was drastically reduced, as there is no soil and only a minimal vegetation cover. There were no possibilities to measure runoff from the deposit directly. The model was therefore indirectly controlled by comparisons between the modelled fraction of total runoff from the area and corresponding estimates from the hydrochemical observations at sites 1, 2 and 3, Kuntebo. In Fig. 5 this comparison is shown for conductivity, zink, aluminium and manganese.

The concentrations of ions at the three points were thus used as internal sources of information and helped us determine the storage and drainage coefficients of this submodel. The results in Fig. 5 show relative large deviations during some time periods. This is due to the fact that the hydrochemical model (described below) was calibrated simultaneously, which forced us to a certain amount of compromizing. It was not possible to get a best fit of the hydrochemical model and the water balance model of the deposit with identical model parameters.

295

Water Balance of Forested Areas

The water balance of the forested areas was modelled by the water balance component of the PULSE model. The model was calibrated against runoff data for Kuntebo, which means that runoff from the deposit is included. This calibration was therefore integrated into the modelling of the water balance of the deposit. The results are presented together with the water quality simulations in Fig. 8.

Hydrochemistry of the Deposit

The original PULSE model has a seasonal depth/alkalinity relationship for the aquifer. For the quality of water leaving the waste dump, the temperature was considered a more significant parameter, since it effects the weathering processes in the waste. The drainage water from the deposit is thus assumed to become increasingly acidic with depth and with higher temperatures. This is modelled by a 60 days running air temperature relationship according to

$$\text{ALK}_d = \text{ALK}_0 - C_1 \, d \, T_{60}$$

where

  ALK_d – alkalinity of water contributing from level d of the deposit aquifer,
  ALK_0 – alkalinity of water at the surface of the deposit aquifer,
  d – depth in the deposit aquifer,
  T_{60} – running 60 days air mean temperature,
  C_1 – empirical coefficient.

In Fig. 6 the results of the calibration of the model versus the variations in pH in the water that leaves the waste deposit are shown. The result is, as described above, to some degree a compromize, as the optimum parameters did not match the water balance of the deposit completely.

Hydrochemistry of the Forested Areas

The pH and alkalinity of the forested areas were modelled by a conventional calibration of the hydrochemical subroutine to observations from the area upstream the deposit (site 1). Support from flow data was lacking, but the water balance model parameters were rather well under control by the previous calibration against runoff data for the total basin of Kuntebo and agreed well with experiences from the PULSE model in other applications. Results from the calibration of the hydrochemical model parameters for the unaffected forested area are shown in Fig. 7. The optimum chemical model parameters obtained for this part of the basin were then assumed to be valid for the total unaffected forested part of the basin.
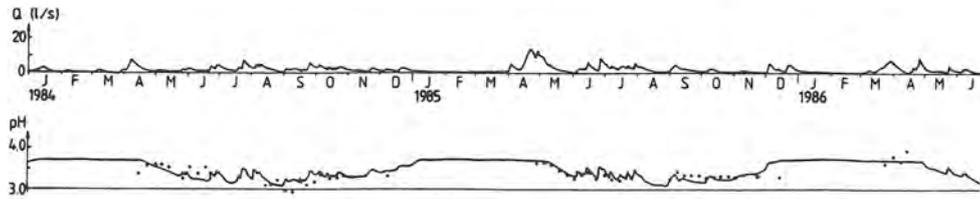
Fig. 6. Simulted runoff and pH from the deposit (site 2 in Fig. 1). Dots represent measurements.
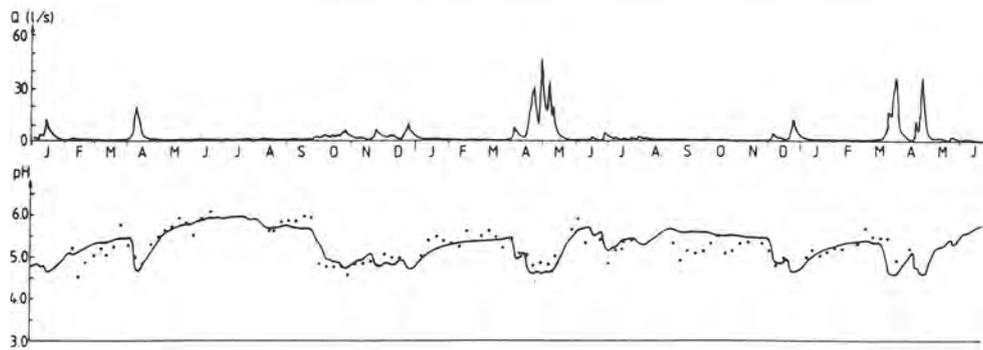
Fig. 7. Simulated runoff and pH at station 1, upstream the deposit. Dots represent measurements.
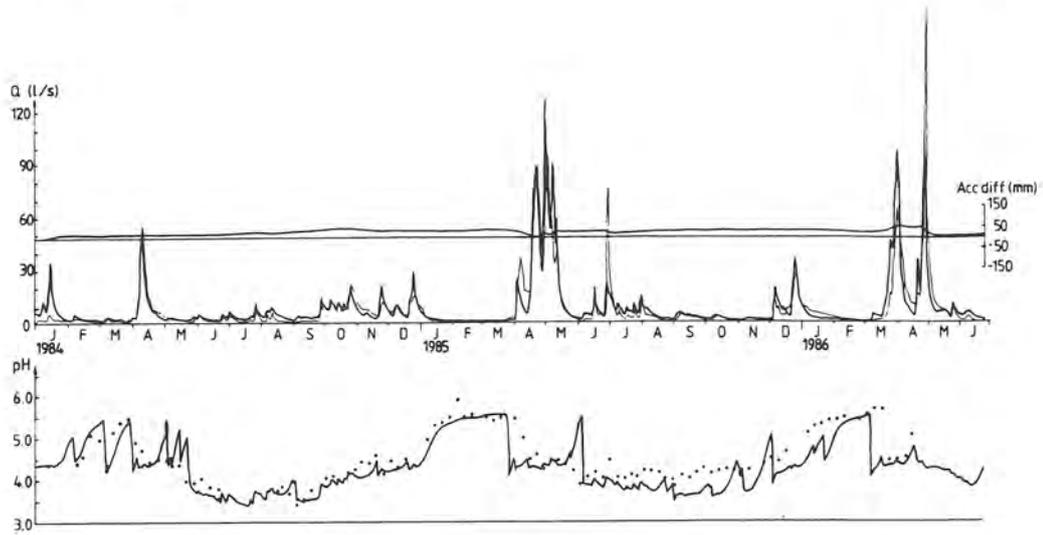
Fig. 8. Integrated simulation of runoff and pH at Kuntebo. Thin line and dots are measurements.
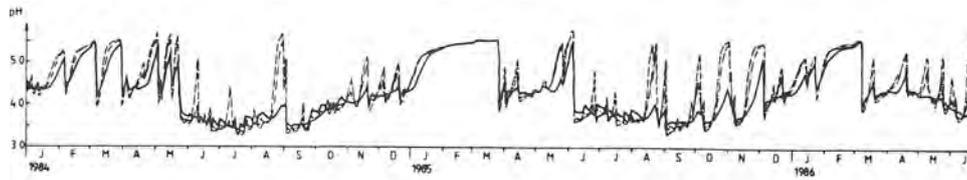
Fig. 9. Sensitivity of the model to a change in the upper recessions coefficient for the deposit. Values are 0.1, 0.2 and 0.3 (day^{-1}) respectively.
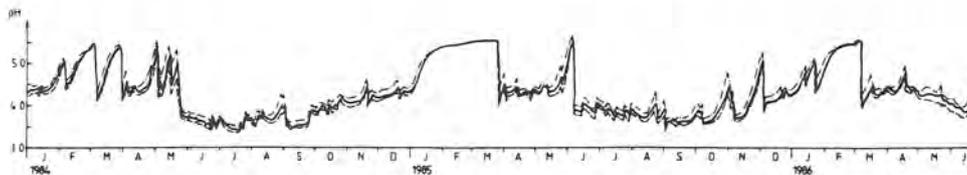
Fig. 10. Sensitivity of the model, if the areal extent of the deposit is increased or decreased by 30% respectively.

Integrated Modelling of Alkalinity and pH at the Outlet

With all previous model calibrations at hand, a simulation of runoff, alkalinity and pH was finally performed for the outlet of the basin, station Kuntebo. The results, which are shown in Fig. 8, are thus in a way an independent test as far as the hydrochemical variables are concerned, while the water balance has been subject to calibration in an earlier step. The water balance matches reasonably well, and some of the disagreement can be attributed to the long distance to the air temperature station (35 km) and the simple degree-day melt function. The systematic underestimation of pH-levels by the model could possibly be corrected by a recalibration of the chemical components of the submodels. Here, again, we have to face a compromize, because the recalibration would improve the simulation at the outlet but would undoubtedly affect the performance of the submodels in an adverse way.

Not surprisingly the hydrochemistry responds directly to the acidic waters from the deposit, which makes the correct modelling of the contribution of water from each subarea extremely important. This result is supported by the study of Sandén *et al.* (1987) in a companion paper. The effect is further illustrated by the sensitivity analyses in Fig. 9, which shows the effect on the pH simulation of a change in the upper recession coefficient of the submodel for the deposit.

Fig. 10 illustrates a sensitivity analysis of the model to the areal extent of the waste disposal site. The fact that this sensitivity is small compared to that of the recession coefficient, indicates that it is more important to determine storage characteristics of the waste than it is to determine its exact areal coverage.

298

Conclusions

The application of the PULSE model to the Bersbo area showed that the variation pattern of the hydrochemistry to a high degree can be explained by climatological and hydrological conditions.

A diversity of data from different subareas can be used for model calibration and support in the assessment of the various model parameters. A certain amount of compromizing was necessary when trying to model the water balance and water quality of waters from the deposit. Oversimplifications in the structure of this submodel are probably the main causes of this problem. There is also a slight conflict between best fit of the submodels and the complete model. One reason for the latter could be uncertainties in the control of drainage water from the deposit, but also the generalization of hydrochemical model parameters from the area upstream the deposit to the total unaffected forested area.

The importance of a correct water balance simulation of each subbasin is emphasized in the integrated basin simulation, in particular if, like in this case, one part of the basin has a completely different and dominating hydrochemistry compared to surrounding areas.

Acknowledgements

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III

Generation, Transport and Deposition of Suspended and Dissolved Material
Examples from Swedish Rivers

by

Maja Brandt

GENERATION, TRANSPORT AND DEPOSITION OF SUSPENDED AND DISSOLVED MATERIAL - EXAMPLES FROM SWEDISH RIVERS

by

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ABSTRACT. Soil erosion, sediment transport and deposition in river systems in Sweden are discussed. The database consists of observations from a research project and from the Swedish network for the measurement of sediment transport. Examples are given from measurements in small plots, and from river basins of different sizes and characteristics. Effects of hydrological regime, of deposition in lakes, and of geology and human impact are illustrated. It was found that observations of erosion losses in index plots cannot easily be extrapolated to large areas, and that trends of transport most likely reflect trends in runoff.

Introduction

Erosion and sediment loads in rivers have not been considered a serious environmental problem in Sweden, except locally on agricultural land and if vegetation has been removed due to construction work, mining etc. The low erosion rates have generally been attributed to low rainfall and snowmelt intensities, and to permeable soils, limited surface runoff and dense vegetation cover.

Sediment derived from eroded land can, however, be a major pollutant and a carrier of polluting chemicals, such as nutrients (especially phosphorus), pesticides, and contaminants such as heavy metals and complex organic components. Radionuclides may also be carried by soil particles. Pollutants can of course also be transported dissolved or adsorbed to complex colloids. Transport mechanisms and related environmental problems are thoroughly discussed in the literature by, e.g. Ongley (1982), Novoty *et al.* (1986), Broberg and Persson (1988), Ahl (1988), Sandén *et al.* (1987), Svensson (1987), Wolman (1977), Sundborg and Rapp (1986).

The aim of this paper is to give examples of erosion studies made in Sweden on different scales in order to bring into focus the role of sediment transport in Swedish rivers and to form a background for future studies of pollution. Two main sources of data have been used in the analyses - a study in small plots and basins in the Verkaå research basin, about 35 km north of Stockholm, and the nationwide network for the measurement of transport of sediment and dissolved material. The small plot investigations contribute to information about the processes and the studies in larger basins give more information about integrated large-scale effects.

The stream or river transport of sediment and dissolved material at the outlet of a basin is a final result of a number of mechanisms: diffuse discharges from drainage areas, point and nonpoint source emissions, fluvial erosion, sedimentation, uptake, enrichment, and other interactions between the running water, sediment, biota, atmosphere etc.

The major nonpoint sources of pollutants are associated with the fine material and organic fractions of sediments, and they are often enriched during transport (Avnimelech and McHenry 1984). The paper therefore concentrates on transport of suspended material, not considering bed load.

Data and methods

Verkaå data

Water erosion in plots (5 m²) and in smaller basins (1.5 - 8 km²) were studied during the International Hydrological Decade (IHD) during the period 1969-72 in the representative area, Verkaå, by the author (Edqvist 1972). The Verkaå area is 116 km² and is situated 35 km north of Stockholm (Figure 1). It consists of 44 % forest and 52 % open land and its land use is mixed. The Lake Fysingen (4 % of the area) is situated near the outlet. During the IHD-period the runoff and water quality parameters were studied at more than twenty small sub-basins (Nilsson 1969, Nilsson and Armolik 1980). Table 1 gives the land use and sizes of the sub-basins used in this study. Basins 3, 4, 7 and 8 have the highest proportion of arable land (40 - 45 %). Basin 16 consists mostly of forest; its 4 % arable land is situated close to the measurement station.

Table 1. Land use in the sub-basins of the Verkaå research basin.

Basin	1	4	2	3	7	8	16	13 + 16
Area (km²)	115.8	77.0	2.5	3.5	5	1.5	3.4	8.3
Forest (%)	44	46	65	48	26	42	86	73
Arable land (%)		42	30	42	45	40	4	14
Pasture (%)	52	9	4	8	12	8	2	6
Farms, villages, roads etc. (%)		3	1	2	17	10	8	7
Lake (%)	4	0	0	0	0	0	0	1
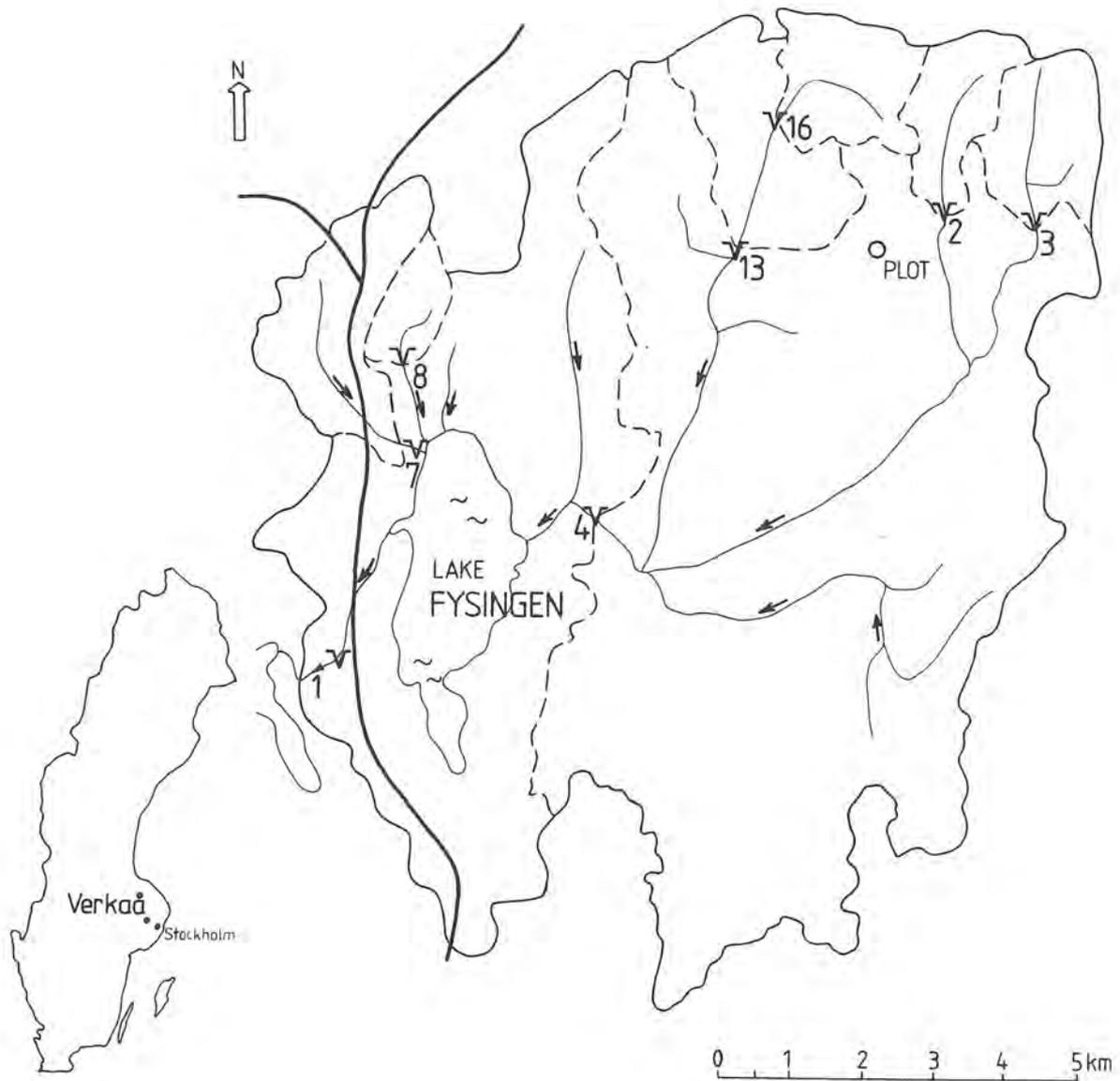
Figure 1. Location of runoff, sampling for sediment transport and plots in the Verkaån basin. The numbers indicating the sub-basins are the same as in Table 1.

Samples were taken every week, or even more frequently during heavy rain or snowmelt, in six sub-basins. Sheet erosion was measured in two plots (5 m²) (Figure 2). The plots were situated in medium clayey soil with a slope of about 5°. One plot lacked vegetation and the other was covered by grass. In the same area sediment transport was measured in two larger sub-basins (Hydrological Data - Norden 1973).

Figure 2. Photo of the plot construction (the plot without vegetation). The plate and the small catching container were normally covered by a plastic roof. The mean slope of the plot is about 5° on a medium clayey soil.

National network data

The transport of suspended and dissolved material in some Swedish rivers has been measured since 1967 in a nationwide network of permanent stations (Figure 3) (Nilsson 1971, Brandt 1982 a). The network is made up of 36 stations. Eleven of these stations have been measuring since 1966/67 (more than twenty years) and fourteen others have been measuring more than ten years. The sizes of the drainage areas vary from 4 to 48 000 km². The stations of the network were chosen to constitute a representative choice of Swedish rivers. Most of the stations are situated near the mouths of large rivers, as one aim was also to calculate the transport to the seas around Sweden. Many of the rivers are regulated.

5

Figure 3. Location of sediment transport stations in Sweden (nationwide network). The station names mentioned in the paper are underlined.

The following physical water quality parameters are measured: concentration of suspended inorganic and organic matter, concentration of dissolved matter, turbidity, conductivity, colour, pH and water temperature.

The measurements have shown that it is important to take frequent water samples during periods of rising runoff in order to get satisfactory transport calculations and to notice rapid changes in the water quality parameters. The sampling frequencies for the rivers vary from every day to twice a month, depending on the variations of runoff.

The amount of suspended matter is calculated by filtering the water sample through a membrane filter of 0.05μm. The total amount of suspended material is weighed after drying at 105 °C, and the amount of inorganic matter is defined as the residue after ignition at 550 °C for two hours.

The total dissolved solids in the water are measured as the residue after evaporation at 105 °C (filtered water). No allowance for the changes of carbonates has been made.

Suspended sediment discharge rating curves are constructed by regression analysis (Miller 1951, Piest 1963, Temple and Sundborg 1972), and the transport for days without measurements is estimated with the aid of these curves.

The transport of dissolved solids is estimated by linear interpolation of concentrations between the sampling dates and multiplication by daily runoff at the corresponding sites. These runoff values are based on the official network of the Swedish Meteorological and Hydrological Institute (SMHI).

Results from the Verkaå study

Table 2 shows the transport of suspended matter measured during different seasons and includes sheet wash in the 5 m² plots and integrated water erosion in three of the small basins investigated in the Verkaå area.

The study showed that sheet erosion on medium clayey soil occurred during snowmelt on both the uncovered and grass-covered plots (Table 2). A rapid thaw with rain in the latter part of March 1971 caused much of the topsoil in the bare plot to become mobile, as rain and meltwater flowed over the surface. Some intensive rain in the summer of 1971, especially 11 - 12 August, caused comparably high sheet erosion from the uncovered plot.

In the beginning of the summer of 1971 there was no sheet erosion but was still some runoff in the basins, mostly as base flow after the spring flow. The rain in July to September caused sheet erosion in the two plots but nearly no runoff in the basins. The rainwater was very quickly infiltrated in the soil and only raised the soil moisture.

Alström and Bergman (1988) studied sheet and rill erosion in the south of Sweden, and their study also showed a wide range of losses during two winter/spring seasons. During the period January to April 1988 the loss of sediment was only 1/500

Table 2. Erosion on plots compared with sediment transport in small sub-basins in the Verkaån area.

Period	5 m² plots		Small basins (2.5 - 3.5 km²)		
	Uncovered in mg/m² (=	Grass kg/km²)	Basin 3	2 in kg/km²	16
Autumn 1970 11.9.-18.11	0	20	27	6	100
19.-26.11.1970 Snowmelt and prec.	675	80	322	597	247
21.1.-14.2.1971 Snowmelt	22	227	199	349	231
20.3.-1.4.1971 Snowmelt	3 113	364	508	556	45
Early summer 1.6.-18.7.1971	0	0	15	3	7
Summer 1971 prec. (only >20 mm/day considered) 19.7.	17 458	48	0	0	0
11.-12.8.	82 570	167	0	0	0
15.8.	227	18	0	0	0
3.9.	2 074	52	0	0	0
Autumn 1971 4.9.-31.10.	64	44	5	0	11
Winter 1971 1.11.-16.12.	40	43	98	200	189

of the loss from the same period in 1987, though the precipitation was twice as high. This was the result of the mild winter of 1988, when there was not any snow cover, frozen soil, or snow melt. Sheet and rill erosion is more dependent on intensive rain and snowmelt than on the total amount of precipitation during the year. In particular the snowmelt is the most critical factor for erosion losses.

We could expect the sediment removal to be larger the more arable land there is, but the differences in sediment transport between the sub-basins in Verkaå area are normally small (Tables 1 and 2). The sediment derived from the sheet and rill erosion that sometimes could be noticed on arable land in the area was often deposited on the nearest flat land and did not reach the streams.

The land use characteristics of the basins, which vary between 40 to 90 % forest and 4 to 40 % arable land, is not sufficient to explain differences in sediment transport. The transport of suspended material is more dependent on the location of the sediment sources, and on the channel network. Only small portions of the total drainage basin and the stream bed contribute to the transport. The measurements showed that the transport was highest when the banks had been frozen and the soil structure was decomposed when thawed. This is illustrated in Figure 4, which shows the unit sediment yields for some of the sub-basins of the Verkaå area for April 1971 during a snowmelt period. The proportions of arable land and forest in the sub-basins are found in Table 1.
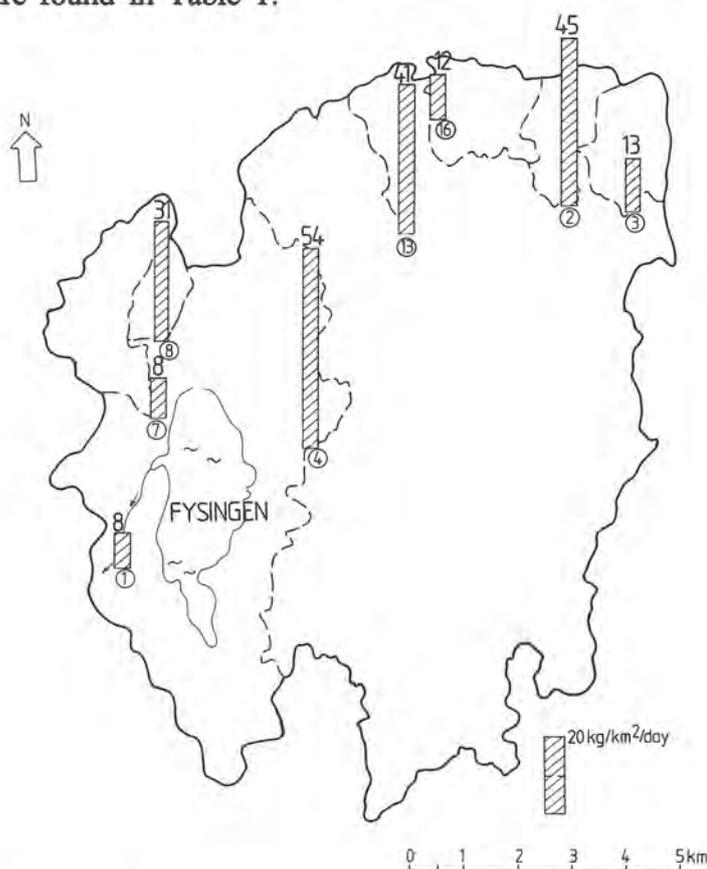
Figure 4. *Unit sediment yields for some of the sub-basins of the Verkaå area in April 1971 during a snowmelt period. They illustrate that yields are not directly related to the proportions of arable land in the sub-basins. Basins 3 and 7, with the highest proportions of arable land (about 45 %) have about the same yields as the forested Basin 16 (4 % arable land). At the outlet of Lake Fysingen (Station 1) lower sediment yield is measured than at the inlets to the lake (Stations 4, 7 and 8, which correspond to 75 % of the total drainage area of Lake Fysingen) due to deposition of the larger particles.*

In the eastern part of USA a Universal Soil Loss Equation (USLE), based on more than 10 000 plot years, is used to predict areal erosion (Wischmeier and Smith 1965, Amoldus 1977). One of the limitations of the equation is that it fails to reflect the erosive potential of runoff from thaws and snowmelt runoff (Jansson 1982). Thus we can not use this type of equation in Sweden.

The soil losses show great spatial variation. In this study, using multi-regression analysis, I have not found any clear statistical relationship between possible factors such as land use or dominating soil type. In drainage basins in the Scania region of southern Sweden Ryding (1984) found a relationship between the amount of suspended material, the percentage of phosphorus in the suspended material and the degree of arable land, but in the most cultivated areas there was a high degree of scatter. Alström and Bergman (1988) found in the same district that slope, drainage area and percentage of arable land had the greatest impact on soil loss by rill erosion.

Results from larger basins

Results from the nationwide sediment transport network have been documented by Nilsson (1971) and Brandt (1982a, b and 1986). Some of the most interesting conclusions concern the effect of hydrological regime, of deposition in lakes, and the effects of geology and human impact and this will be presented below. The stations discussed, in total 12, are underlined in Figure 3.

Large-scale effects of the hydrological regime on sediment transport

Seasonal and interannual variations in sediment transport in larger basins in Sweden are mainly determined by the climate. This is exemplified in Figure 5 by the monthly sediment yields for 1986 (which was a quite normal climate year) for four representative unregulated basins. The rivers are chosen to illustrate the most important differences in annual runoff regimes in Sweden (Melin 1970). In the north of Sweden the winter normally lasts several months, followed by snowmelt first in the lowlands and later in the mountains. Because of this there is low runoff and transport in the winter and high runoff and transport in the spring, summer, and autumn in rivers from the mountains, as for example the river Piteälven (Bölebyn in Figure 5).

Figure 5.   *Monthly sediment yields in 1986 (a normal climate year) at four stations*
(next   *in unregulated rivers in Sweden. (Notice that the scale of sediment yields*
page)   *is logarithmic.) The curves are representative for runoff regimes and*
             yields from a large unregulated mountain and forested drainage area
             also in the north (Böleby), a smaller forested area also in the north
             (Torrböle), two mostly forested drainage areas in the southwestern (Ås-
             bro) and southeastern (Emsfors) parts of Sweden. The runoff regime
             gradually changes from the north to the south of Sweden. Monthly runoff
             and sediment yields from both dammed and undammed rivers in the
             north and the central Sweden are further illustrated in Figures 6 and 7.
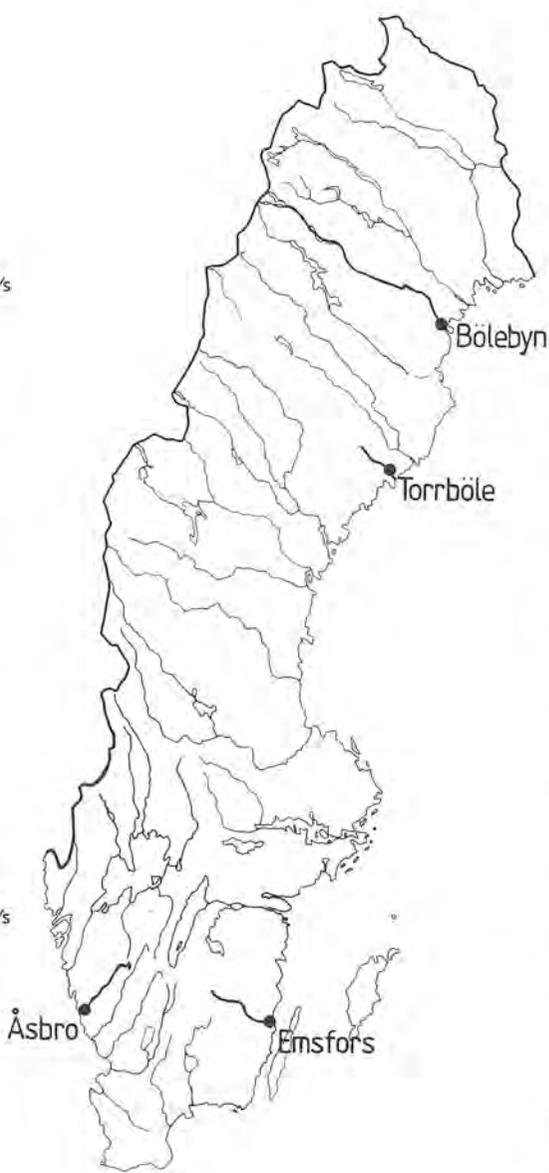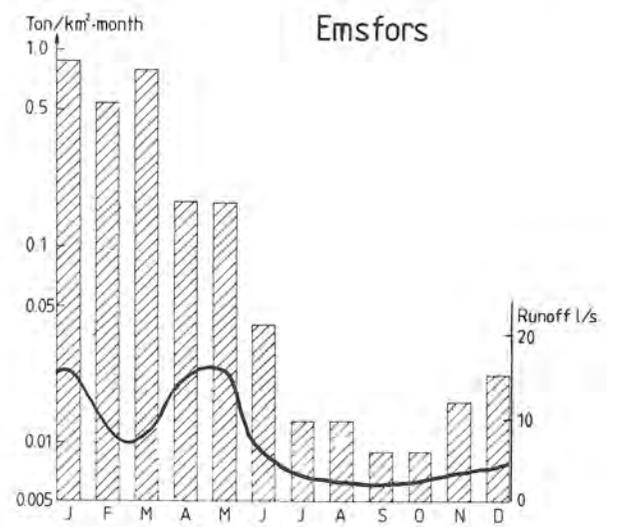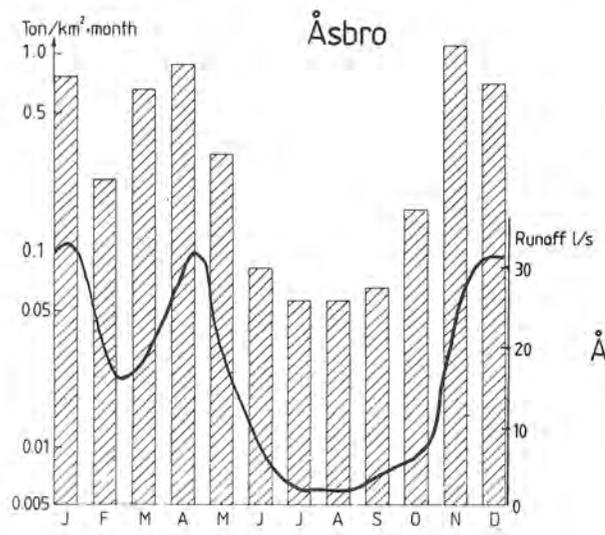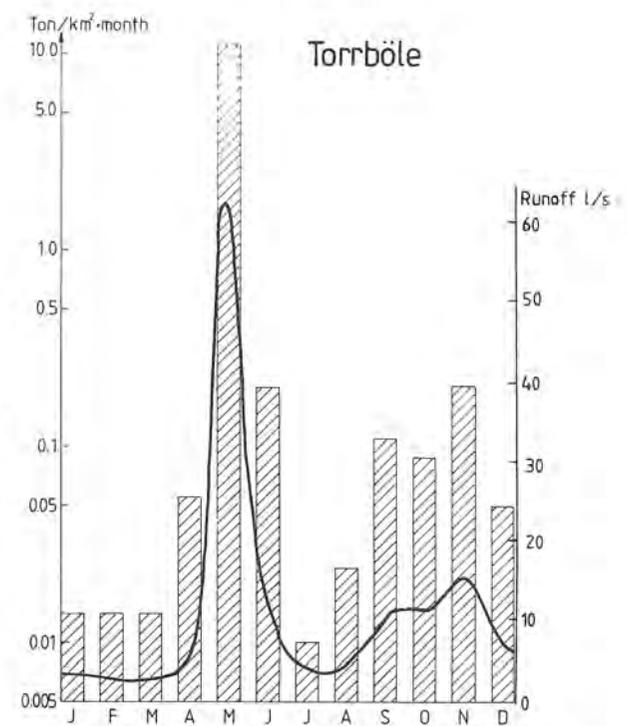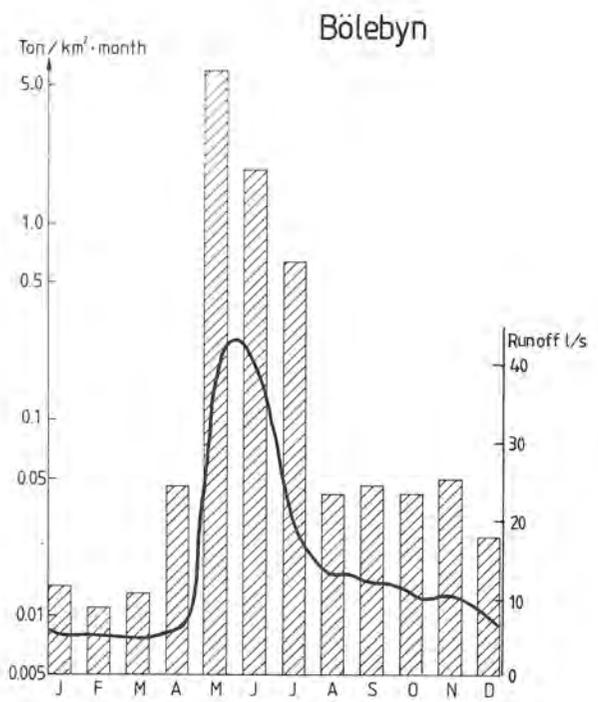
Torrböle station on the river Öre älv illustrates a forest river system, with high flow during both snowmelt in May and June and in the autumn due to precipitation. In summer the runoff is low due to evapotranspiration. The annual runoff regime gradually changes from the north to the south of Sweden. In the south of Sweden the winter is much shorter and is interrupted now and then by snowmelting. Thus, there is a quite different runoff regime, with the highest flows from November to May, but low runoff and transport in the summer during the growing season. Southwestern Sweden is much wetter than the southeastern area, due to weather systems coming mostly from the west.

Effects of lakes

The concentration of suspended material is normally lower in larger rivers than in small ones. In the smaller basins with no lakes, such as in Bergshamra, Stormyra, Ryttarbacken and the sub-basins of Verkaå, the concentrations are more than twice the mean value of those of nearby larger basins. In larger basins the suspended material settles in lakes and reservoirs, on flood plains and in the river channel. Results from plots, from small basins and from large basins can therefore not be directly compared.

There are no direct measurements of the effects of lakes in the network but there is one river (Dalälven) with two stations separated by a lake system. Älvkarleby station is situated near the mouth of the river, downstream from a large lake system. For the years 1976-88 this station had less than 60 % of the annual unit sediment yield observed at Wikbyn station, located above the lake system. The effect of lakes on sedimentation is greatest at high flows when the concentration is high. The yield during May at Älvkarleby station was generally less than 30 % of the yield at Wikbyn. Figure 6 shows monthly sediment yields during 1987 for both stations.
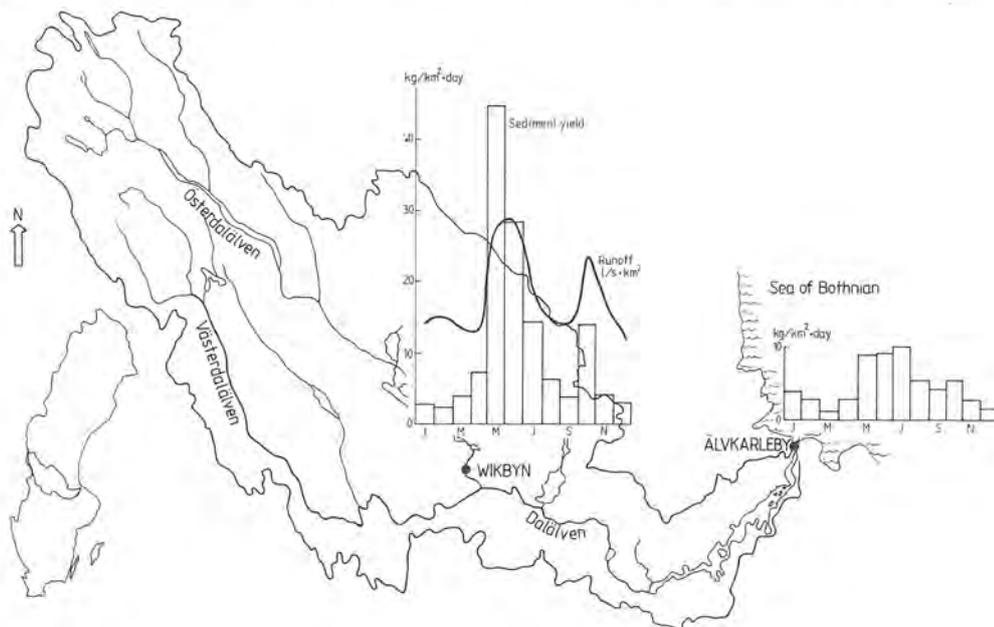
Figure 6. Monthly sediment yields in 1987 in the river Dalälven illustrating effects of deposition in a lake system between Wikbyn and Älvkarleby.

Other studies of suspended material and turbidity along rivers in Sweden confirm the importance of lakes working as sediment traps (Hjorth 1972, Brandt 1982 b).

Effects of geology and land use

Other factors that influence sediment transport in a large river basin are geology, soil type, land use, morphology and man's impact along the river. There is often interaction between geology, soil and land use. For example, Swedish forests often grow on till soil but arable land is mostly found on finer sediment or clay soil.

Some of the stations, as for example Maltbränna on the river Vindelälven, have very low concentrations of suspended inorganic matter even during spring flow (in average only 3 mg/l). This river flows through large narrow lakes near the mountain border in the northwest. Farther down, the drainage area consists mostly of flat forest and wetland areas of till soils. The effect of the geology can be illustrated by the measurements at Maltbränna compared with the measurements at Torrböle on the river Öre älv (Figure 7). The relief of the Öre älv basin is much steeper, with the river running in a deep valley. In the valley there are glaciofluvial materials, glacial and postglacial sediments etc., which are easily eroded. Much higher suspended inorganic concentrations are found here at high flows (mean 40 mg/l, maximum 230 mg/l).
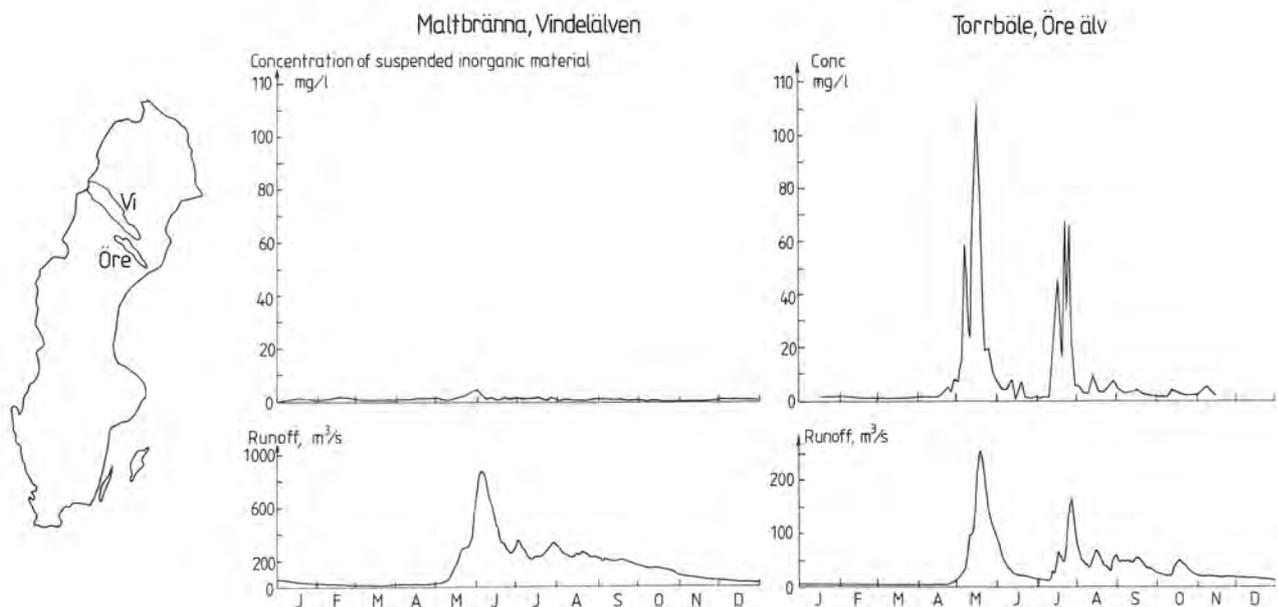
Figure 7. *Runoff and concentrations of inorganic suspended material at Maltbränna station on the river Vindelälven (= Vi) and the Torrböle station on the river Öre älv (= Öre) 1979. The river Vindelälven runs through flat forest and wetland areas with low erosion. Station Torrböle is situated in a valley consisting of sediments that are easily eroded.*

Examples of the impact of man's activity on sediment transport

Most rivers in the north and central part of Sweden are regulated for hydro power production. This affects the sediment transport. Fluvial erosion depends to some extent on water velocity, which normally varies in an unregulated stream. Short term fluctuations with high water velocities are more pronounced in a regulated river. Silting in reservoirs is a severe problem in some parts of the world, but in Sweden this is normally not a major problem. Shoreline erosion due to high amplitudes of dammed lakes occurs but most of the eroded material settles in the large artificial reservoirs (Brandt 1980).

Figure 8 shows runoff and monthly sediment yields during 1975 in a regulated river (Skellefteälven) and an unregulated river (Vindelälven) in the north of Sweden. In both cases the drainage areas are between 9 500 and 10 000 km^2 and the areas have about the same morphology, geology, and land use. The stations are situated inland far from the mountains and both streams have low concentrations of suspended matter. During the May and June spring flows in the period 1967-78 the yields in the unregulated Vindelälven were 50 - 75 % of the total annual sediment yield and in the regulated Skellefteälven only 15 - 35 %. During January and February the yields were only 5 % of the annual yields in Vindelälven and 15 % in Skellefteälven. This effect is mostly due to the change of runoff regime and not to silting in the reservoirs.
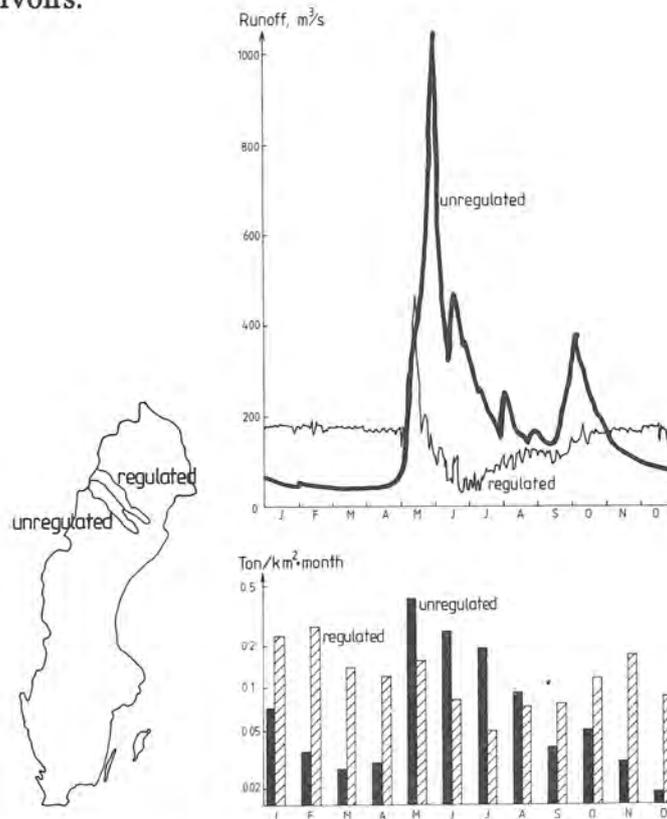
Figure 8. Runoff and monthly sediment transport 1975 in an unregulated river, Maltbränna station on the river Vindelälven (thick line in the upper and black bars in the lower diagram), and in a regulated river, Skellefteälven (Renströmsgruvan) (thin line in the upper and shaded bars on the lower diagram).

Nilsson (1976) has estimated the transport reductions to be 50 - 60 % after water regulation in three large rivers - Ume älv, Ångermanälven and Indalsälven - in Sweden. The development of hydro-electric power in Swedish rivers was most intensive during the period 1950-60, and there are few measurements before that time for comparison.

Trend-analyses of transport of dissolved material

The transport of dissolved material has been analyzed with respect to trends, as an increased transport of organic matter (measured as $KMnO_4$-consumption) has been reported (Forsberg and Löfgren, 1988) for the river Dalälven during the last 20 years. This increase is considered to be caused by an increase of dissolved humus in the water. Forsberg (1988) suggests that the main reason for a greater amount of increased humus is an increase in the atmospheric deposition of nitrogen, which is thought to be a result of forest growth, deposition of litter, and microbiological activities. Better forest management could also influence humus production. Humus is a carrier of metals, especially mercury and iron as complex organic compounds, and this could change fluvial transport of metals to the sea.

In order to find out if there is any trend in the transport of dissolved matters as measured by the sediment transport network, a trend analysis was made based on monthly mean values for the period 1967-87.

A nonparametric test of significance trend, the Seasonal Kendall test, has been used (Hirsch et al. 1982, Hirsch and Slack 1984). This test is insensitive to seasonality, robust against non-gaussian distribution and serial dependence. All tests are performed as two-sided tests since both upward and downward trends are of interest. Estimates of the slopes of the trend curves have been calculated by means of the seasonal Kendall slope estimator, which can be described as the median annual change adjusted for seasonal variation.

In Table 3 the results of the statistical analysis are presented for three stations in different parts of Sweden. 95 % confidence intervals have been computed and statistically significant trends at 5 % level have been marked. The relative Kendall slopes give the annual increase or decrease of the trend.

A trend in transport can be caused by changes in discharge or in concentrations or in a combination of the two. There is a significant positive trend for the discharge in the river Dalälven (at Wikbyn) for the period 1967 - 1987; this causes a trend for the transport of dissolved matter but not for concentration. There is also a significant positive trend for dissolved transport in the river Klarälven at Edforsen. This is probably due to positive tendencies of both discharge and concentration. In the river Viskan at Åsbro in southern Sweden there is no significant trend. From this analysis we can not confirm any increase in dissolved concentration, of which humus is a part. However, the transport of dissolved matter seems to be increasing probably because of an increase in runoff for the last ten years, due to a tendency to higher precipitation (Alexandersson and Eriksson 1989).

Table 3. *Result of the statistical trend analysis of runoff, concentration and transport of dissolved material 1967 - 1987.*

| Station | Type | p | Kendall slope | |
			absolute	relative %
Wikbyn	discharge	0.009*	+ 6.83	+ 2.13
Dalälven	dissolved conc.	0.189	+ 0.11	+ 0.29
	dissolved transp.	0.016*	+ 6.71	+ 2.12
Edsforsen	discharge	0.102	+ 1.00	+ 0.85
Klarälven	dissolved conc.	0.067	+ 0.20	+ 0.69
	dissolved transp.	0.023*	+ 136.00	+ 1.46
Åsbro	discharge	0.194	+ 0.25	+ 0.75
Viskan	dissolved conc.	0.205	+ 0.30	+ 0.32
	dissolved transp.	0.058	+ 85.00	+ 1.14

* = Statistically significant at the 5 % level.

Conclusions

To understand nutrient losses and the spread of other pollutants with sediment in the water it is important to know the erosion processes and the pathways of runoff, sediment transport and deposition in a basin.

In quite small basins ($< 10 - 100$ km^2) the sediments near the streams are most likely to produce the major contribution to sediment transport. The soils and the land use around the streams are important. Here conservation measures such as contour farming, shortening of the slope length, conservation tillage and sediment traps can reduce sediment losses but probably not dissolved nutrient losses. A reduction of the loss of dissolved nutrients to lakes and the sea around Sweden is intensively debated today, due to eutrophication problems. Methods to prevent this type of loss, such as reducing nitrogen combustion from traffic, changing land use (wetland restoration), changing cultivation management by more effective use of fertilizers and manure and advocating catch crops cultivated in the autumn after the seed harvest, are discussed.

This investigation does not show any clear relationship between erosion and sediment transport and various elements such as climate, relief, soil, vegetation and land use, either in small or large basins. The snowmelt is the most critical factor for erosion losses and, therefore, it can be very difficult to use any type of universal equation.

The concentrations of suspended material in the larger rivers are normally lower than in the waterways in the smaller basins. The plot studies give an estimate of the effect of splash, sheet and rill erosion. Sediment yield from small basins implies also fluvial erosion and some deposition at the end of the slopes and in depressions. In larger basins there is also deposition in lakes and reservoirs, on flood plains and in the river channel. Erosion rates and sediment yields from plots, from small basins and from large basins can therefore not be compared directly.

Acknowledgements

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IV

Simulation of Runoff and Nitrogen Leaching from Two Fields in Southern Sweden

by

Bergström, S., Brandt, M., and Gustafson, A.

Simulation of runoff and nitrogen leaching from two fields in southern Sweden

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ABSTRACT   A conceptual model for the simultaneous computation of runoff and inorganic nitrogen leaching is described. The model is semi-empirical with coefficients that are calibrated against observed data. Two applications of the model, using well-controlled data from small fields of arable land in southern Sweden, are demonstrated. These show that a major part of the variations of concentrations of inorganic nitrogen in runoff water can be explained by this model approach. Finally, the sensitivity and limitations of this type of model are discussed.

Simulation de l'écoulement et de la pollution par l'azote
provenant de deux champs au sud de la Suède
RESUME   L'article en question décrit un modèle destiné à montrer la simulation de l'écoulement et de la pollution par l'azote inorganique. Le modèle est semi-empirique, et fournit des coefficients ajustés à partir des données observées. Deux applications du modèle sont montrées à l'aide de données bien contrôlées, provenant de petits terrains cultivés, au sud de la Suède. Ces applications montrent que la plupart des variations de l'azote inorganique dans l'écoulement, peuvent être expliquées à l'aide de ce modèle. Finalement sont discutées la sensibilité et les limites d'un modèle de ce genre.

INTRODUCTION

The contribution of nitrogen to Swedish rivers, lakes, coastal waters and groundwater has increased due to human activity with risks for eutrophication problems. One of the sources of nitrogen is leaching from arable land, of which 90% consists of nitrate. The leaching depends on many factors, of which the water balance, amount of fertilizer, type of crop and soil are the most important.
    In 1983 a cooperative project between the Swedish University of Agricultural Sciences (SLU) and the Swedish Meteorological and

Open for discussion until 1 December 1987.

Hydrological Institute (SMHI) was initiated in order to model the hydrology and leaching of nitrogen from typical agricultural fields in Sweden. The work is based on an extensive data base collected by the Division of Water Management at SLU (Brink *et al.*, 1979) and, as an ultimate goal, is intended to be used for control of impact of agricultural activities on surface and ground waters.

The aim is to develop a model structure which includes the most important factors influencing nitrogen turnover and leaching but with a complexity that does not exclude application to areas with normal data coverage. This means that many of the very complex processes of the nitrogen cycle have to be simplified, and that empirical expressions with coefficients found by optimization methods often have to be used.

The starting point of the work is the simulation experiences gained with the HBV and PULSE conceptual models for runoff, pH and alkalinity in flowing waters (Bergström & Forsman, 1973; Bergström, 1975; and Bergström *et al.*, 1985) and studies of nitrogen leaching at SLU (Brink *et al.*, 1979). The routines for nitrogen turnover and leaching in the model have been influenced by the CREAMS model (Knisel, 1980), the SOIL-N-model (Johnsson *et al.*, 1987) and the NITCROS model (Hansen & Aslyng, 1984).

WATER BALANCE MODEL

A schematic presentation of the water balance model is shown in Fig.1. The model is run with daily time steps with precipitation and mean air temperature as input together with monthly standard values of potential evapotranspiration.

The simulation of runoff is made in three steps:
- snow accumulation and ablation,
- soil moisture accounting,
- generation of runoff and transformation of the hydrograph.
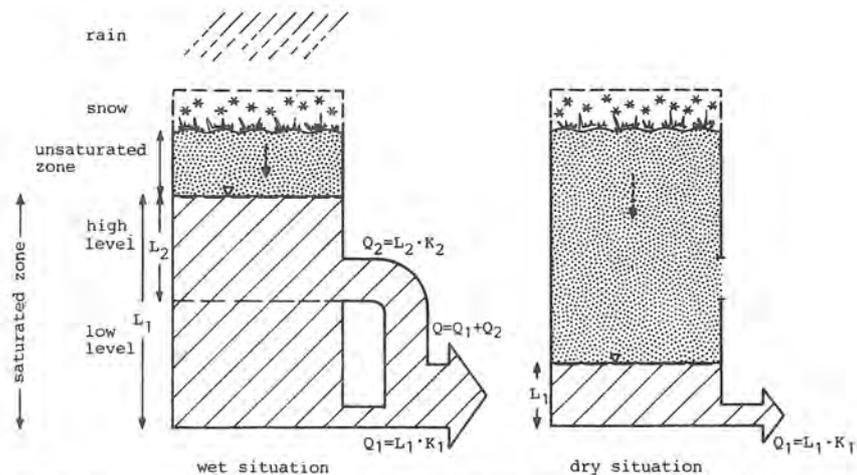
Fig. 1   Schematic presentation of the water balance model during wet and dry situations.

Snow accumulation starts as soon as the temperature is lower than a threshold value. The melt routine of the model is essentially a degree-day approach with a liquid waterholding capacity of 10%.

The soil moisture accounting routine is summarized in Fig.2 and has the effect that the contribution to runoff from an equal amount of rain or snowmelt is small when the soil is dry and larger under wet conditions. The actual evapotranspiration decreases as the soil dries out. On top of the soil moisture routine there is a simple interception storage, with a capacity of 1 mm, which is emptied by evapotranspiration at potential value.

All excess water from the soil moisture zone is collected in the saturated zone and drained at two levels with two recession coefficients which account for quick superficial runoff and deeper groundwater with slow drainage. A simple routine for capillary rise to the unsaturated zone allows runoff to drop to zero during long dry periods.

There is a noticeable spatial variability in soil water characteristics even in small fields. Hansen *et al.* (1986) have found the range of spatial dependence to be in the order of 20-40 m. Even if the soil can be considered homogeneous, the moisture conditions can vary considerably from higher areas down to lower ones. To account for spatial variability, the model is divided into six submodels with possibilities for different parameter settings.

NITROGEN MODEL

The loss of nitrogen from the soil is an integrated result of a number of processes, all more or less controlled by physical environmental factors such as soil humidity, soil temperature and water movement, but also of management factors such as fertilization, crop rotation and cultivation. Due to this complexity and to variable weather conditions, the magnitude of leaching and its temporal distribution usually vary greatly both within and between years.

The strategy of the model approach used here has been to involve
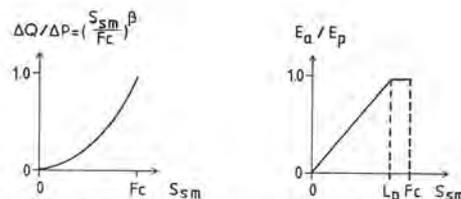
Fig. 2   Schematic presentation of the soil moisture accounting subroutine in the hydrological model.
S_{sm} = computed soil moisture storage,
ΔP = contribution from rainfall or snowmelt,
ΔQ = contribution to the response function/runoff,
Fc = maximum soil moisture storage,
β = empirical coefficient,
E_p = potential evapotranspiration,
E_a = computed actual evapotranspiration,
L_p = limit for potential evapotranspiration.

only the most important factors controlling nitrogen cycling in order to keep the model as simple as possible. Thus the main nitrogen sources considered are fertilization, mineralization and atmospheric fallout; and the main sinks are uptake by plants, leaching and denitrification (Fig.3). As indicated in Fig.4, there are four main compartments in the model where inorganic nitrogen is stored.

Nitrogen fertilizer

In the model, the commercial nitrogen fertilizer and farmyard manure are treated as a separate storage, N_f, in the surface layer of the topsoil. The fertilizer is assumed to dissolve in and follow the infiltration water, thus adding to the main inorganic nitrogen storage of the soil. This transport is assumed to be proportional to precipitation plus meltwater and the actual storage of the fertilizer in the surface layer according to the following expression:

$$\Delta N_i = C_i . N_f . P \tag{1}$$

where
ΔN_i = loss of fertilizer nitrogen through infiltration,
N_f = stored nitrogen in fertilizer in the surface layer,
P = rainfall and snowmelt (minus interception),
C_i = empirical coefficient.

Denitrification

Some of the nitrogen in the surface layer of the model is lost through denitrification. This occurs when the mean air temperature for five days is above 5°C and the soil moisture is at least 50% of available maximum capacity, as shown in equation (2). Diffuse loss
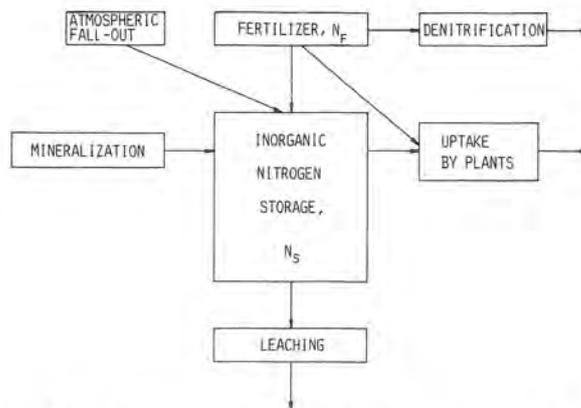
Fig. 3   Schematic presentation of the nitrogen turnover and leaching routines.

through denitrification from the main storage deeper in the soil is neglected.

$$\Delta N_d = C_d \cdot N_f \cdot \frac{S_{sm} - 0.5 \text{ Fc}}{0.5 \text{ Fc}}$$ (2)

$$\Delta N_d = 0 \text{ if } T_5 < 5°C \text{ or } S_{sm} < 0.5 \cdot \text{Fc}$$

where
ΔN_d = loss of fertilizer nitrogen through denitrification,
N_f = stored nitrogen in fertilizer in the surface layer,
S_{sm} = soil moisture storage in the model,
Fc = maximum soil moisture storage in the model,
T_5 = mean air temperature for 5 consecutive days,
C_d = empirical coefficient.

Atmospheric fallout

Grennfelt & Hultberg (1985) estimated the atmospheric fallout of nitrogen to be about 20 kg ha^{-1}year^{-1} in southern Sweden. In the model this source is assumed to be 0.04 kg ha^{-1}day^{-1} during winter and 0.07 kg ha^{-1}day^{-1} during summer, when there is more dry deposition.

Mineralization

Nitrogen mineralization is defined as the transformation of nitrogen from organic into inorganic forms. In the model only the soil moisture in the upper 200-300 mm of the soil and the soil temperature govern mineralization. The mean air temperature for five consecutive days is assumed to represent soil temperature. No mineralization will occur if this temperature is below 5°C.

A separate water storage in the upper soil profile was introduced for the computation of mineralization. This storage is filled by rainfall or snowmelt and emptied by potential evaporation. Its capacity is in the order of 20-50 mm water. This separate storage is only used in the mineralization routine of the model and will not affect its total water balance.

The procedure for mineralization is thus summarized in equation (3):

$$\Delta N_m = C_m \cdot T_5 \cdot \frac{S_u}{S_{u \text{ max}}}$$ (3)

$$\Delta N_m = 0 \text{ if } T_5 < 5°C$$

where
ΔN_m = contribution to the soil storage of nitrogen by mineralization,
S_u = soil moisture storage in the upper soil profile,
$S_{u \text{ max}}$ = capacity of the upper soil moisture storage,
T_5 = mean air temperature for 5 consecutive days,
C_m = empirical coefficient.
So far mineralization has been modelled without consideration of the amount of organic nitrogen in the soil. To account for the abundance of fresh organic matter from crop residues, the

mineralization rate after harvest of winter rape and ploughing of
ley was doubled.

Uptake by plants

The plants are considered to satisfy their nitrogen demand both from
the fertilizer stored in the surface layer (N_f) and from the
nitrogen stored in the soil (N_s) with priority given to the
fertilizer nitrogen. The date of sowing and the harvest date are
input to the model. The potential uptake for a given crop is
assumed to increase gradually during the first 20 days after
sowing. It remains at full potential during a period of rapid
growing for that crop and is then reduced after ripening. The
actual uptake is proportional to the potential uptake, to the sum
of the two inorganic nitrogen storages (both as fertilizer in the
surface layer and as main nitrogen storage in the soil) and to the
computed actual evapotranspiration, as shown in equation (4):

$$\Delta N_u = C_u \cdot E_a \cdot (N_f + N_s) \cdot U_p \tag{4}$$

where
ΔN_u = plant uptake of nitrogen,
E_a = actual computed evapotranspiration,
N_f = stored nitrogen in fertilizer in the surface layer,
N_s = stored nitrogen in the soil,
U_p = potential uptake,
C_u = empirical coefficient.

Leaching

Two mechanisms are used to control the leaching of nitrogen in the
model: transport by percolating water through the unsaturated zone,
and washout below the groundwater surface (Fig.4). The equation
for total loss by leaching is modelled according to equation (5):

$$\Delta N_\ell = (C_p \cdot Q_p + C_w \cdot L_2) N_s \tag{5}$$
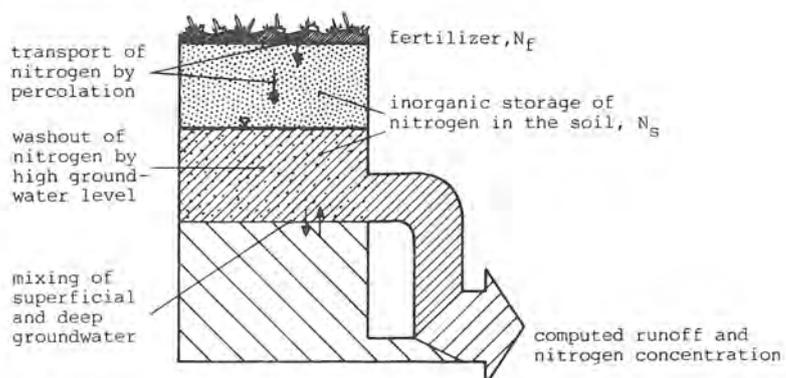
Fig. 4   Schematic presentation of the leaching mechanism in the nitrogen model.

where

ΔN_{ℓ} = leaching of nitrogen from the soil storage,

Q_p = percolating water,

L_2 = groundwater storage in the upper level of the model (see Fig.1),

N_s = computed storage of inorganic nitrogen in the soil,

C_p = empirical coefficient for transport of nitrogen,

C_w = empirical coefficient for washout of nitrogen.

The two leaching mechanisms were justified by experience when applying the model to real data. In particular, it was found essential to account for the washout effect of high groundwater levels in autumn in order to explain the sudden increases in concentrations.

To level out jumps in concentrations caused by sudden changes of the groundwater table in the model, the concentrations in the high and low levels of the saturated zone (Fig.1) are allowed to mix by an exchange coefficient, C_e. The final step in the computation of nitrogen concentrations in runoff is mixing of the water from the high and low levels of the saturated zone, and mixing of all contributions from submodels if a distributed approach is used.

DATA BASE

The model has been tested for two experimental fields in southern Sweden (Fig.5). Skottorp is situated close to the west coast in an area with a precipitation in the order of 750 mm year^{-1}. Sand is the dominating soil type of the field. The area is 0.15 km^2. The other field, Näsby gård, has an annual precipitation of 600 mm year^{-1}. The field consists mainly of loamy till and has an area of 0.36 km^2.

The discharge from the covered drainage tiles has been measured continuously for many years, and the drainage water has been analysed for nitrogen about every second week. Daily precipitation and temperature values were available from nearby meteorological stations. For Skottorp the precipitation station is 5 km away on a ridge of about 170 m higher elevation, which influences its representativeness for the site. Monthly standard values (30-year means) of Penman estimates of the potential evapotranspiration were taken from Eriksson (1981).

Fig. 5 Geographical locations of the chosen basins.

The crop rotation system in Skottorp was fairly complex during the last nine years, which complicated the modelling work. During the first couple of years the field was divided into two parts of equal size, one with cereal and the other with ley. At Näsby gård there was a crop rotation mainly consisting of spring wheat, barley and sugar-beet. There was also one year with winter rape and one with winter wheat. Commercial fertilizers were used at both sites and also liquid manure at Skottorp.

SIMULATION OF RUNOFF AND NITROGEN CONCENTRATION

The parameters of the water balance model were first calibrated for the two fields. Thereafter the nitrogen part of the model was calibrated over six years for Skottorp and over nine years for Näsby gård by a trial and error technique until acceptable agreement between observed and simulated concentrations was achieved. The last three years were left as an independent test period.

The nitrogen routines of the model have essentially seven empirical coefficients that have to be calibrated:

C_i for infiltration of nitrogen fertilizer (equation (1)),
C_d for denitrification (equation (2)),
C_m for mineralization (equation (3)),
C_u for nitrogen uptake by plants (equation (4)),
C_p for loss by percolation (equation (5)),
C_w for loss by washout (equation (5)),
C_e for exchange between the high and low levels of the model.

In addition to these seven parameters a potential nitrogen uptake pattern has to be established for each crop.

When calibrating the nitrogen routines, the option for spatial variabilities was used by the introduction of six submodels with variable depths of threshold of the high level of the saturated zone (Fig.1). This had no significant effect on computed runoff but gave nitrogen a smoother and more realistic variation pattern.

Skottorp

Figure 6 shows an example of a complete output for a one year simulation of the water balance and nitrogen turnover for Skottorp.

The results of the simulations of runoff and of the concentration of nitrogen in the drainage water for the calibrated period (1976-1982) at Skottorp are presented in Fig.7, and those for the independent period (1982-1985) are shown in Fig.8. In 1983, potatoes were grown for the first time. Thus the potential uptake by potatoes had to be adjusted during the independent period.

The concentrations show a pattern with relatively small temporal variations. In the autumn of 1984 the concentrations were quite high due to spreading of manure before sowing of winter wheat. Most likely only small amounts of this nitrogen were taken up by the crop, and the storage in the soil increased accompanied by heavy leaching.

Näsby gård

Figure 9 shows the runoff and nitrogen concentrations for Näsby gård
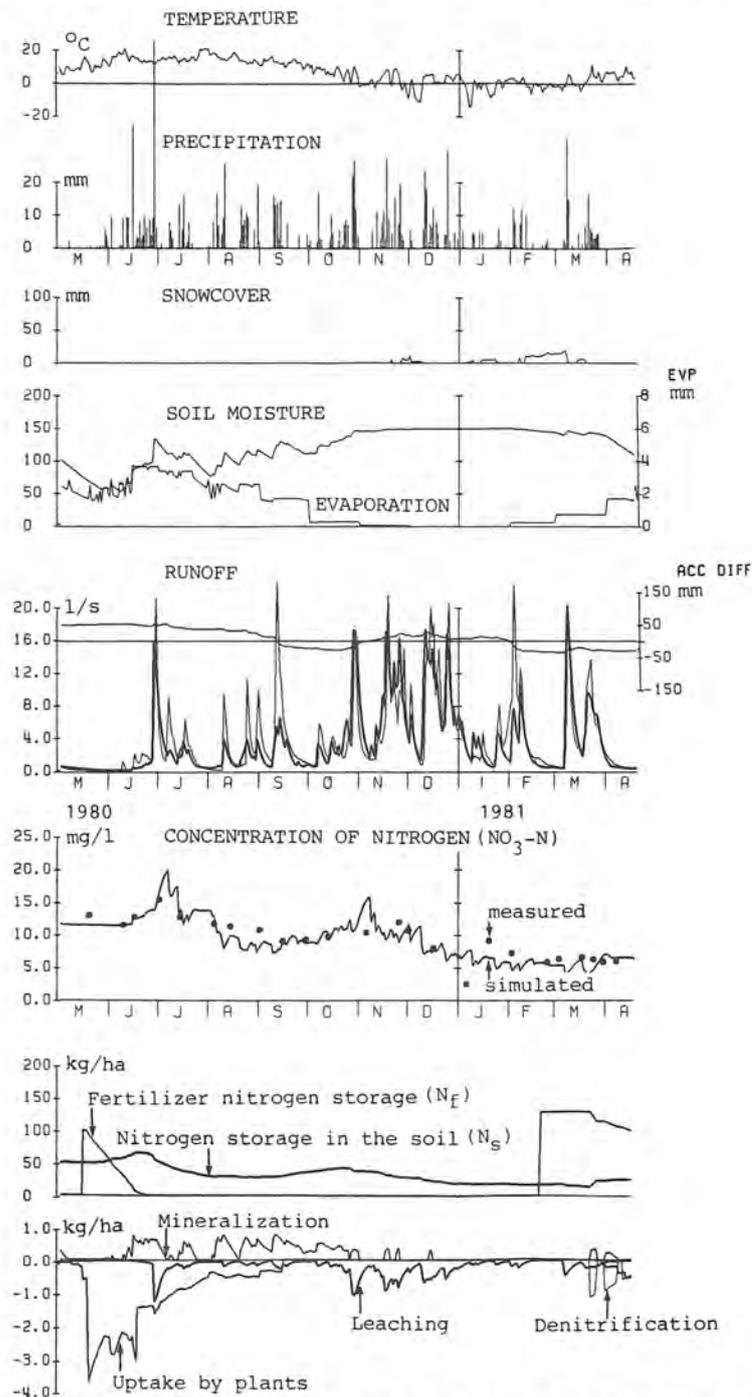
Fig. 6   Results of the water balance and nitrogen modelling for one year at Skottorp. Temperature and precipitation are measured. Snow cover, soil moisture and evaporation are simulated. The thin line is measured runoff and the thick line simulated runoff. The dots show the measured concentration of nitrate in the drainage water and the line the simulated concentration.
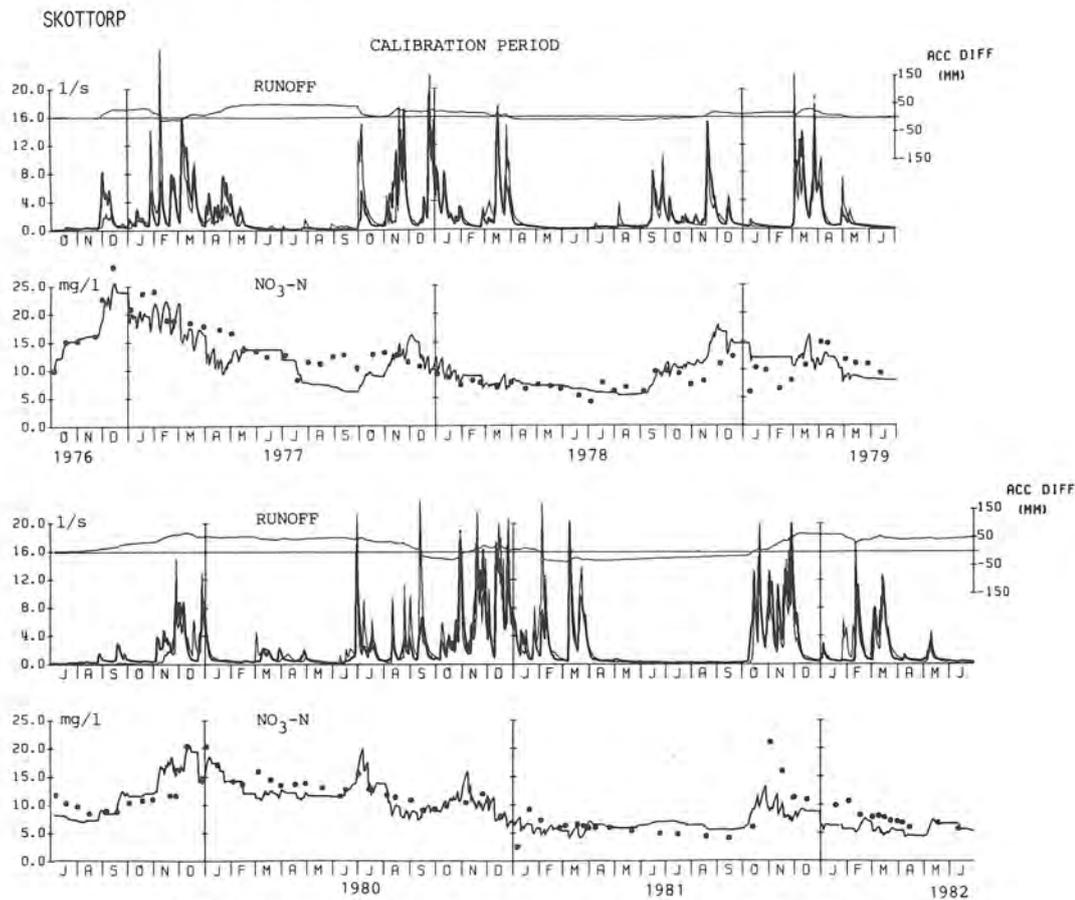
SKOTTORP

Fig. 7 Simulations of runoff and concentration of nitrogen for the calibration period at Skottorp. (Thick line = simulated runoff; thin line = measured runoff; dots = observed concentrations.)
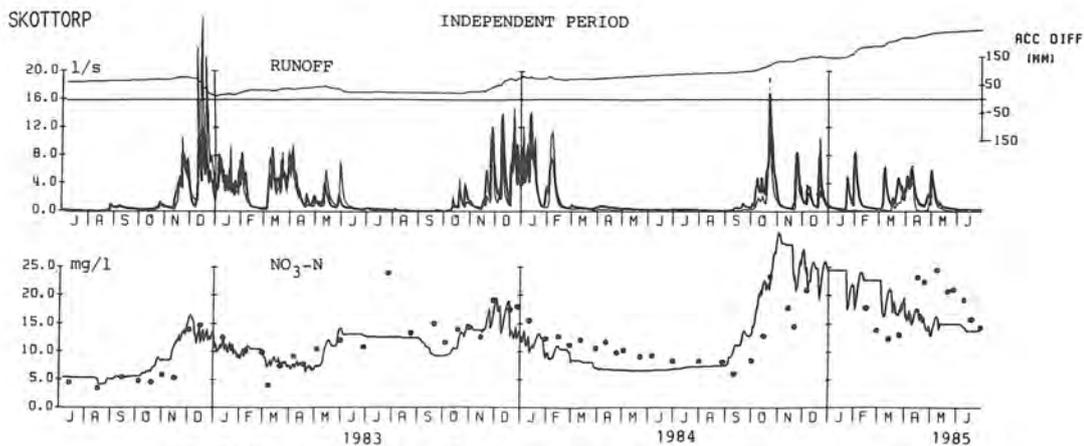
Fig. 8 Simulations of runoff and concentration of nitrogen for the independent period at Skottorp. (Thick line = simulated runoff; thin line = measured runoff; dots = observed concentrations.)

during the calibration period and Fig.10 for the independent
verification period. The drop of the simulated concentrations to
zero indicates that the modelled runoff is zero.

NÄSBY GARD
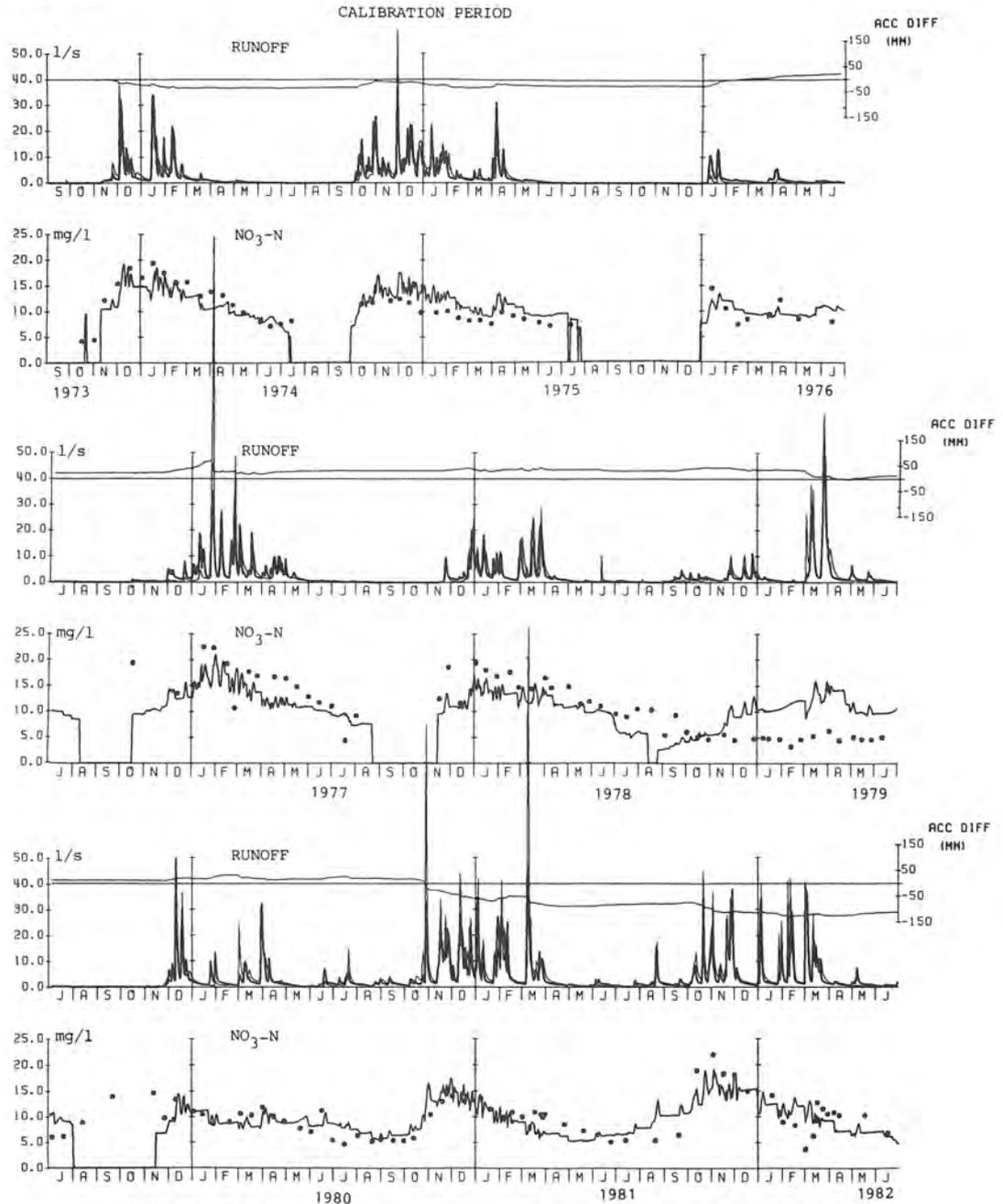
Fig. 9 Simulation of runoff and concentration of nitrogen at Näsby gård (calibration period). (Thick line = simulated runoff; thin line = measured runoff; dots = observed concentrations.)
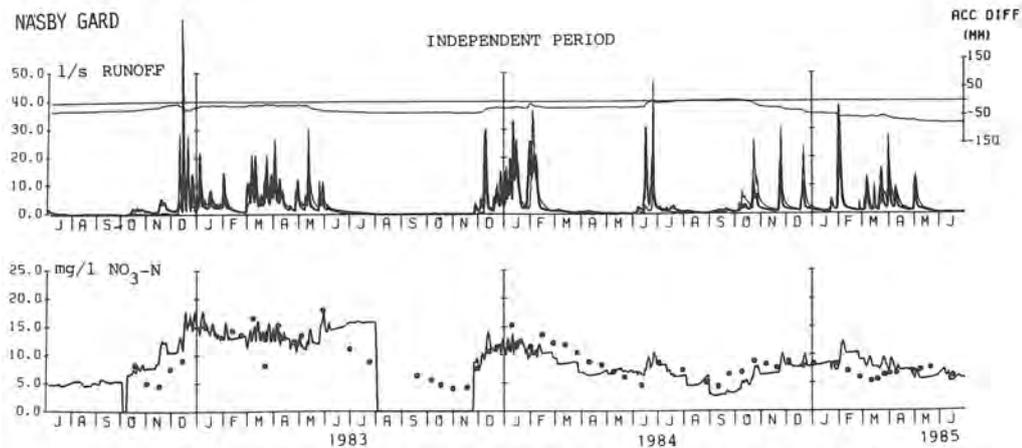
Fig. 10   Simulations of runoff and concentration of nitrogen at Näsby gård (independent period). (Thick line = simulated runoff; thin line = measured runoff; dots = observed concentrations.)

The simulated and measured concentrations of nitrogen agreed fairly well most years with some exception for parts of 1977 and 1979. There usually were increasing concentrations in combination with increasing waterflows during the late autumn/winter periods, followed by decreasing concentrations during drier periods of the year. The increasing concentrations can be explained by increased mineralization after harvest followed by intensive percolation and washout when the groundwater level rose.

SENSITIVITY OF THE MODEL

There is a considerable amount of feedback in the nitrogen modelling routines which moderates the effect of perturbations of its parameters. Figure 11 shows the effect of the simulated nitrogen concentration if the parameters C_m, C_u and C_w are increased or decreased by 30% respectively for the Skottorp site. C_u is the most sensitive one of the three parameters.

In order to explore the possible interaction between the water balance model and the subroutines for nitrogen turnover, a sensitivity analysis based on perturbations of parameters of the water balance routines was carried out. In Fig.12 the effect is shown of a 30% increase and decrease of the threshold, L, and of a substantial variation in the upper recession coefficient, K_2 (L and K_2 are explained in Fig.1).

The sensitivity analysis shows that the interaction between the water balance model and the routines for nitrogen turnover is not alarming. This is, of course, due to the fact that the uptake by plants is as dominating as the sink over leaching, but probably also due to feedback in the uptake equation (4) between stored nitrogen and uptake.

Concentration of nitrogen in the runoff
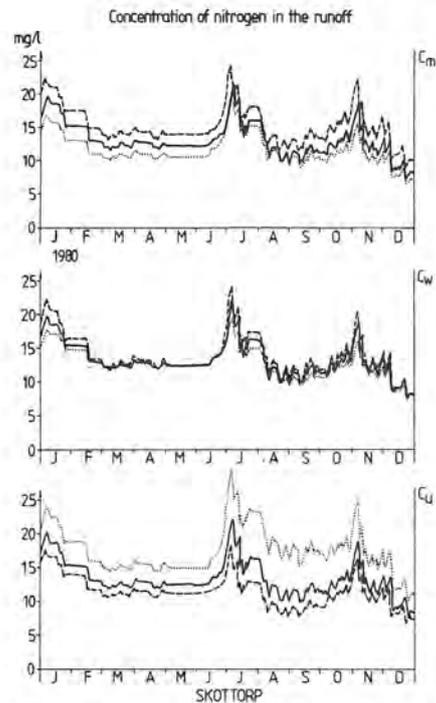
Fig. 11   Sensitivity tests of the nitrogen turnover routines for the Skottorp site. The parameters C_m (that effects mineralization), C_w (washout) and C_u (uptake by plants) are decreased (······) and increased (-----) by 30%.

Concentration of nitrogen in the runoff
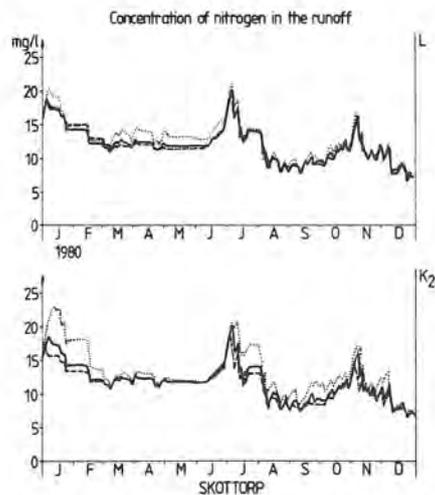
Fig. 12   Sensitivity test of response of the nitrogen turnover to the water balance model for the Skottorp site. The threshold parameter L is increased 30% (-----) and decreased 30% (······). Values of the upper recession coefficient K_2 are 0.1 (······), 0.2 (——) and 0.3 (-----) respectively. (L and K_2 are further explained in Fig. 1.)

DISCUSSION OF THE RESULTS

As shown in Figs 7, 8, 9 and 10, the variability in runoff is much

greater than the variability in nitrogen concentrations. This implies that runoff data or a good water balance model is crucial for estimates of the losses of nitrogen from arable land into rivers, lakes and coastal waters.

This modelling approach, with two measures for the calibration of the empirical coefficients, namely runoff and nitrogen concentration of runoff, leaves a lot of freedom for variation of the internal storages and sources and sinks of nitrogen. Therefore, these have to be considered as internal variables only, and their interpretation must not be abused. As a matter of fact, there is only limited control of rates of mineralization, denitrification and plant uptake. Infrequent analyses of the inorganic storage of nitrogen in the soils in the two experimental fields have, however, shown that the modelled storage has a realistic order of magnitude.

The model occasionally showed large deviations from measured nitrogen concentrations. It is, however, encouraging to note that this normally happened during low flow periods, which only are of minor importance for the magnitude of leaching from the fields. It can therefore be concluded that the model is quite accurate for calculation of transport peaks of nitrogen during flood situations.

The combination of two semi-empirical models, one for water balance and one for nitrogen loss, makes it difficult to generalize the model parameters. If the water balance model is recalibrated and its optimum parameters are adjusted, we may very well need to recalibrate the nitrogen model as well and may end up with a different set of model parameters.

This is a general problem for integrated hydrological and hydrochemical models and has been recognized by, for example, Christophersen *et al.* (1982) when applying the Birkenes model to hydrological and hydrochemical data in Norway. Sensitivity analyses of the response of the nitrogen turnover to the water balance model show, however, that this problem is relatively small in this model approach.

After this study it is now felt that one main uncertainty is the assessment of nitrogen uptake by plants. Plant uptake is highly variable between years even for a single crop. Another main uncertainty is the rate of mineralization, which is one of the dominating sources in our nitrogen balance. In the present model we have, for example, no limiting effect controlled by the available organic matter in the soil.

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Simulation of Runoff and Nitrate Transport from Mixed Basins in Sweden

by

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Simulation of Runoff and Nitrate Transport from Mixed Basins in Sweden

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A conceptual model for simultaneous computation of runoff and inorganic nitrogen leaching is described. The model is semi-empirical with coefficients that are calibrated against observed data. It has been developed and tested for mixed basins consisting of forest, arable land and lakes. In two of the basins, lakes have a major impact on the transport of nitrate. The results show that a major part of the variations of inorganic nitrogen in runoff can be explained by the model, but details in the modelling of nitrogen turnover can be masked by biological and physical processes in the lakes.

Introduction

Non-point source pollution is in focus in many countries today and counter-action is sought to fight increasing eutrophication problems in our lakes and coastal waters. The problem is in many respects inter-disciplinary. The total transport of nutrients in our river systems is a joint effect of land use, farming and forest management, climatological and hydrological conditions and the configuration of the river system itself. The complexity of the system may look overwhelming at first sight, but a few steps towards modelling can be taken, if only the most significant factors effecting transport are considered. Simplification is necessary, and the problem is to identify which processes we are allowed to simplify and which ones we can omit. This decision is, of course, depending on the aim of our study.

A number of nitrogen models have been developed (see for example Giorgini and Zingales 1986; Haith 1982; Frissel and van Veen 1981; and de Willigen *et al.*

1989) but few have been quantitatively tested against measurements. Many of them are process-orientated and need a lot of indata or are specific for a basin or an arable field. There still exist uncertainties with respect to several important properties governing nitrogen flows such as mineralization, denitrification, and uptake by crops. Other models are made more available for practical use but usually only for agricultural land.

Our study started in 1983 as a cooperative project between the Swedish University of Agricultural Sciences (SLU) and the Swedish Meteorological and Hydrological Institute (SMHI).

In 1986 a model for small agricultural fields was developed and tested against long-term field measurements collected by the Division of Water Management at SLU (Bergström *et al.* 1987). The starting point was the HBV and PULSE conceptual models for runoff (Bergström and Forsman 1973; Bergström 1975; and Bergström *et al.* 1985a). The routines for nitrogen turnover and leaching in the model for arable land was influenced by the CREAMS-model (Knisel 1980), the SOILN-model (Johnsson *et al.* 1987) and the NITCROS-model (Hansen and Aslyng 1984). CREAMS is an empirical model developed for field-size agricultural areas to be used by practitioners for managing non-point pollution. The Swedish SOILN-model and the Danish NITCROS-model are more process-orientated models.

The present study has the objective to find an operational model for the calculation of nitrate transport from mid-size or larger river systems of mixed land use.

The idea is to make use of earlier model studies in smaller basins as well as data from existing monitoring programmes. This means that nitrogen turnover for each specific subarea has to be treated in a rather crude manner to keep the model complexity under control. The use of the model will thus mainly be restricted to simulation under stationary conditions, but the effect on water quality of land use changes in parts of a basin can be introduced if its local effect is known. The model will thus be able to answer a question like: "What would the nitrate transport look like if the forested area was larger?", but possibly not a question like: "What would happen to the transport if different farming practice was introduced in a specific area?"

Special attention is given to the scaling problem when detailed information from small research basins is generalized and applied to the larger river systems. The problem addresses the question of justification of complexity level of a model, considering available data in the basin.

The results show that a model can be used to transfer information from small homogeneous research basins to larger basins of mixed land use. Lakes play a dominating role in the dynamics of nitrate transport. A river system with significant lakes can therefore be modelled by simple standard loss functions for typified areas instead of by a more complete model for nitrate turnover. A simple exponential decay function seems to be appropriate for the simulations of nitrate reduction due to physical and biological processes in the lake itself.

14

Water Balance Model

The water balance part of the model is almost identical with that of the HBV and PULSE-models. This means that the model is run with daily time steps with precipitation and mean air temperature as input together with monthly standard values of potential evapotranspiration.

The simulation of runoff is made in three steps.

- Snow accumulation and ablation,
- Soil moisture accounting,
- Generation of runoff.

Snow accumulation starts as soon as the temperature is lower than a threshold value. The melt routine of the model is essentially a degree-day approach with a liquid waterholding capacity of 10 %.

The soil moisture accounting routine has the effect that the contribution to runoff from an equal amount of rain or snowmelt is small when soil is dry, and larger under wet conditions. The actual evapotranspiration decreases as the soil dries out.

All excess water from the unsaturated zone of the model is collected in the saturated zone and drained at four levels with recession coefficients which account for quick superficial runoff and deeper groundwater with slow drainage. A simple routine for capillary rise to the unsaturated zone allows runoff to drop to zero during long dry periods.

The model can further be distributed into submodels, which makes it feasible for application to an area with different land use.

The runoff routing of the inflow hydrograph through the lakes is explicitly described with discharge rating curves for the outlet of each significant lake. The basin is then divided into submodels defined by the outlet of each one of these lakes. The rating curve is described as

$$Q = K \, W^{E} \tag{1}$$

where

Q – computed runoff,
W – computed water level in the lake, above a given threshold,
K and E – empirical coefficients.

The coefficients K and E are known for a limited number of lakes in Sweden – if not, standard values can be used (Bergström *et al.* 1985b).

Due to the nonlinearities, the routing through the lakes must be calculated with shorter time-steps than one day.

The explicit description of lake routing in the model is a step towards a more physically correct representation of the river system and has proved to reduce the efforts needed to calibrate the recession coefficients of the model.

15

Nitrate Loss Model

The loss of nitrogen from the soil is an integrated result of a number of processes, all controlled by physical environmental factors, such as soil humidity, soil temperature and water movement, but also of mangement factors for arable land, such as fertilization, crop rotation and cultivation. Due to this complexity and to variable weather conditions, the magnitude of leaching and its temporal distribution usually vary greatly both within and between years.

There is an exchange between different kinds of nitrogen, for example mineralization of organic into inorganic nitrogen. Presently, however, the model only handles nitrate with exception for the mineralization source from organic nitrogen. The nitrogen leaching from arable land in the studied area consists of about 80 % nitrate, but from forest areas the corresponding figure is only about 15 %.

As the strategy of the model approach is to involve only the most important factors controlling nitrate turnover, the main nitrate sources considered are atmospheric fallout, mineralization and fertilization (as recommended ratios); and the main sinks are uptake by plants, leaching and uptake in lakes (Fig. 1). All biological and physical processes in the lakes, including uptake by microorganisms, sedimentation and resuspension, exchange between organic and inorganic nitrogen and denitrification, are summarized in a simple exponential loss function.

As indicated in Fig. 1, there are three main compartments in the landphase model, where inorganic nitrogen is stored. The fourth compartment is storage in the lakes.

Atmospheric Fallout

The atmospheric fallout of nitrogen as nitrate is assumed to be about 8 kg/ha year. It is calculated from measurements of wet fallout (Monitor 1983) and an estimation of dry fallout. In the model this source is assumed to be relatively constant in time with little more fallout in summer, when there is more dry deposition. In the forest, some of the nitrate is directly absorbed by the neddles and leaves, and the fallout from the atmosphere to the ground is therefore reduced by 15 %

Nitrogen Fertilizers

As the exact volumes of commercial fertilizers and farmyard manures are not known for all fields in the basins, recommended volumes and fixed dispersal dates for ley and cereals respectively have been used as input.

In the model, fertilizers and atmospheric fallout are entered into a separate model storage, N_f, in the surface layer of the topsoil. The nitrate in the topsoil layer is then assumed be dissolved by precipitation and available for the plants.

It can also follow the infiltration water, thus adding to the main nitrate storage of the soil (N_s). As in the CREAMS-model the transport is assumed to be proportional to precipitation plus meltwater and to the actual storage of the nitrogen in the surface layer according to the following expression

16
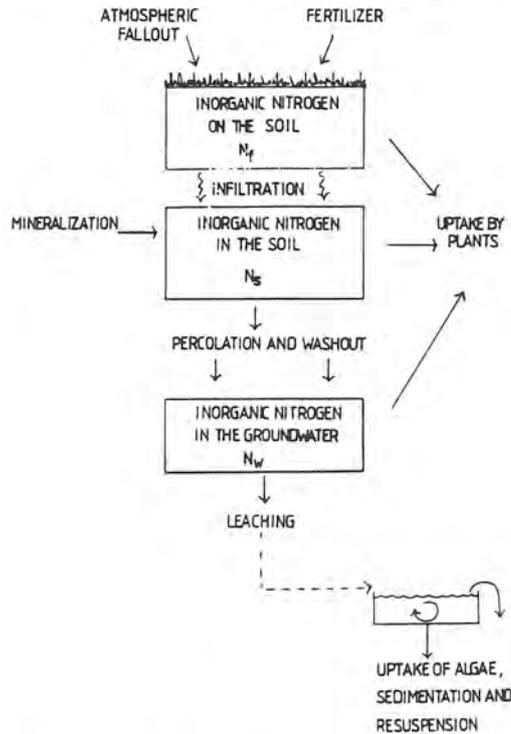
Fig 1. The nitrogen model structure.

$$\Delta N_f = C_i \, N_f \, P \qquad\qquad (2)$$

where

N_f – stored nitrate in the surface layer,
ΔN_f – vertical transport of nitrate from N_f,
P – rainfall and snowmelt,
C_i – empirical coefficient.

The denitrification losses from fertilizers and manures are ignored in this version of the model, because of the already rough estimate of used fertilizer and manure volumes.

Mineralization

Nitrogen mineralization is defined as the transformation of nitrogen from organic into inorganic forms. A comprehensive model of mineralization should take into account both different pools of organic nitrogen and environmental effects as temperature and moisture condition in the soil. The organic nitrogen pools are large and only a part of the organic nitrogen is readily converted into inorganic nitrogen. In the present model it is assumed that there are no limitations in the organic pools.

17

The NITCROS-model uses a slightly curved relation between soil temperature and mineralization (in the interval 0 to 20°C which is relevant under Swedish conditions) and a linear relation between soil moisture content and mineralization up to optium moisture content (field capacity) to describe the environmental effects. A similar idea is used in our nitrate model.

For arable land thus only the soil moisture in the upper 200-300 mm of the soil, that is considered the root zone, and the soil temperature govern mineralization. The mean air temperature for five consecutive days is assumed to represent soil temperature. No mineralization will occur, if this temperature is below a threshold value, which is normally in the order of 3-5°C.

A separate water storage in the upper soil profile was introduced solely for computation of mineralization, as the mineralization at greater depth is small. This storage is filled by rainfall and snowmelt and emptied by potential evaporation. Its capacity is in the order of 20-50 mm water. This storage is only used in the mineralization routine of the model and will not affect its total water balance.

The procedure of mineralization in arable land is thus summarized in Eq. (3)

$$\Delta N_m = C_m (T_5 - T_m) \frac{S_u}{S_{umax}}$$

$$\Delta N_m = 0 \quad \text{if} \quad T_5 < T_m \tag{3}$$

where

ΔN_m – contribution to the soil storage of nitrate by mineralization,
S_u – soil moisture storage in the upper soil profile,
S_{umax} – capacity of the upper soil moisture storage,
T_5 – mean air temperature of 5 consecutive days,
T_m – threshold temperature for mineralization,
C_m – empirical coefficient

In forested areas the mineralization is assumed to be dependent on temperature only. This simplification is justified by the low level of concentration and low variability. The mineralization equation for forest is thus described as

$$\Delta N_m = C_m (T_5 - T_m)$$

$$\Delta N_m = 0 \quad \text{if} \quad T_5 < T_m \tag{4}$$

The coefficient C_m for forest is much smaller than that for arable land.

Leaching

Leaching of nitrate in a model is normally calculated as proportional to the water flow and the nitrate storage in the soil (N_s). When we developed the model for small arable fields in southern Sweden, we found this process insufficient to describe high nitrate concentration in autumn and winter (Bergström *et al.* 1987).

Therefore we introduced an additional leaching mechanism, only active when groundwater levels are high. This was explained as a washout effect. This extra leaching is seldom active in the present study, but it is still a part of the model. The equation for total loss by leaching is thus modelled according to

$$\Delta N_s = C_p \, Q_p \, N_s \quad \text{if} \quad L < L_L$$

$$\Delta N_s = (C_p \, Q_p + C_w [\frac{L-L_L}{L_L}]) N_s \quad \text{if} \quad L \geq L_L \tag{5}$$

where

N_s – computed storage of nitrate in the soil,
ΔN_s – leaching of nitrate from the soil storage,
Q_p – percolating water,
L – groundwater level,
L_L – threshold groundwater level,
C_p – empirical coefficient for transport of nitrate,
C_w – empirical coefficient for washout of nitrate.

The leached nitrate is finally collected in the groundwater as the third storage, N_w (Fig. 1).

Uptake by Plants

The plants on arable land are considered to satisfy their nitrate demand from the fertilizer stored in the surface layer (N_f), from the nitrate stored in the soil (N_s), and sometimes from dissolved nitrate in the groundwater (N_w). The potential uptakes for cereal (taken as one group) or ley (as another group) are assumed to increase after sowing. The uptake remains at its maximum during rapid growing until August and is then reduced for cereals at ripening. The potential uptake for the group ley is not reduced in the autumn.

The actual uptake is assumed proportional to potential uptake, to the mean air temperature for five consecutive days, and to the computed actual evapotranspiration, as shown in Eq. (6)

$$\Delta N_u = C_u \, E_a \, (T_5 - T_u) \, U_p$$

$$\Delta N_u = 0 \quad \text{if} \quad T_5 < T_u \tag{6}$$

where

ΔN_u – plant uptake of nitrate,
E_a – computed actual evapotranspiration,
U_p – potential uptake,
C_u – empirical coefficient,
T_5 – mean air temperature for five consecutive days,
T_u – threshold temperature for uptake by crops.

Bergström *et al.* (1987) describe two' applications of nitrate simulation from very small fields with only one or two crops each summer. The present larger study areas have a mixture of many fields and crops. Therefore the model for arable land had to be simplified with less coefficients for uptake of different crops. Thus only two types of arable land – one with some type of grass during the whole year and the other with cereals and rape – are considered. The latter type is harvested in the autumn and the soil is ploughed and bare afterwards.

The trees in the forests take nitrate from the nitrate stored in the soil, N_s, and from nitrate in the groundwater, N_w. The uptake from the soil is constant, if the mean temperature is above a threshold temperature, T_c, and if the storage N_s is large enough.

The uptake from the groundwater storage is proportional to the nitrate stored in the groundwater storage, and computed actual evapotranspiration according to Eq. (7)

$$\Delta N_w = C_c \, E_a \, N_w$$
$$\Delta N_w = 0 \quad \text{if} \quad T_5 < T_c \tag{7}$$

where

N_w – stored nitrate in the groundwater,
ΔN_w – forest uptake of nitrate from the groundwater,
E_a – computed actual evapotranspiration,
T_5 – mean air temperature for five consecutive days,
T_c – threshold temperature for uptake by forest,
C_c – empirical coefficient.

Physical and Biological Changes in Lakes

The water quality in a lake is dependent on a combination of different factors (James 1984):

– Influent quality and mixing of water in the lake,
– Physical and chemical processes in the lake,
– Biological growth.

In the lakes there is a high activity of micro-organisms, and they take up nutrients. The algae growth is a function of light, temperature and nutrients. The algae die and sink to the bottom. Some sediment can be resuspended, and there may be denitrification at the bottom. The organic nitrogen can also transform to inorganic nitrogen again in the water.

The retention time in the lake and water circulation is essential for the mixing of water and nutrients, but it also gives ample time for water quality to change due to physical, chemical, and biological processes.

Mathematical models of ecological processes have been described by, for exam-

ple, Jørgensen (in Orlob 1983) and James (1984). The problem is that an inclusion of only the most important processes still results in a great number of driving and state variables. The models also require a data base that is normally not available for both calibration and verification (Niemi 1982; Kettunen 1982).

In the light of the aim of the project and of existing data in the studied areas it was not justified to test anyone of the more complete existing ecological models. Instead very simple approach was used to describe the nitrate reduction in the lake. In the model this is governed by air temperature for ten consecutive days (as water temperature was not measured) and nitrate storage in the lake according to Eq. (8)

$$\Delta N_{lake} = C_{lake} \; T_{10} \; N_{lake}$$
$$\Delta N_{lake} = 0 \; \text{if} \; T_{10} < T_{lake} \tag{8}$$

where

N_{lake} – nitrate content in the lake,
ΔN_{lake} – nitrate losses in the lake,
T_{10} – mean air temperature for 10 consecutive days,
T_{lake} – threshold temperature for uptake etc.,
C_{lake} – empirical coefficient.

It may look like a contradiction that the most important sink in the nitrate model thus got the most simplified representation, and it can, of course, be argued whether this is correct. If the use of the model is restricted to nitrate transports from a lake or from a larger watershed to the sea, it may, however, be appropriate, provided the conditions can be considered as stationary.

A Simplified Model Approach

For areas with lakes we also tested a even more simplified method. The nitrate losses from mixed areas to the lake were not computed by the nitrate loss model. Instead the nitrate loss from different land use was calculated from measured nitrate concentration in small homogeneous research basins (as monthly means) multiplied with computed runoff and mixed together in the actual proportion of the land use in that area

$$N_{loss} = \sum (C_{land\,use} \; Q_{land\,use}) \tag{9}$$

where

N_{loss} – total nitrate loss from the area,
$C_{land\,use}$ – typified nitrate concentration in drainage water from a specific land use,
$Q_{land\,use}$ – daily runoff from the specific area.

The nitrate reduction in the lake was modelled with the same routine as for the more complex model.

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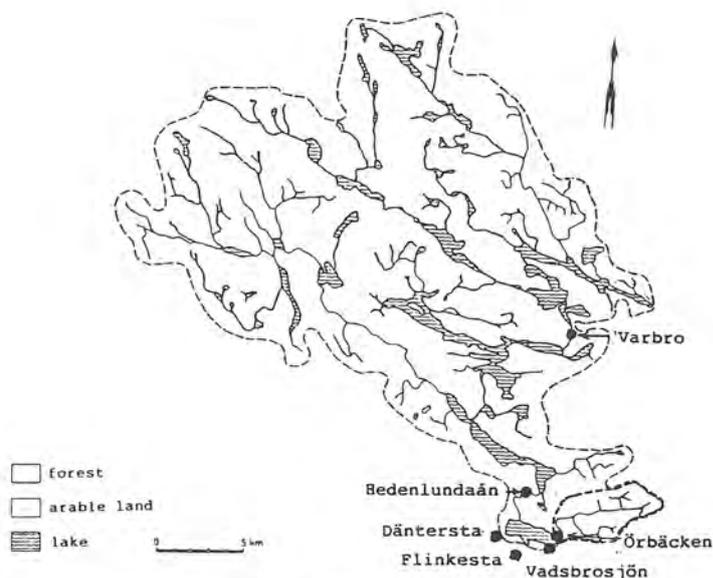
Fig. 2. The Ekenäs area. Runoff and nitrogen measurements are available at Flinkesta, Däntersta and Örbäcken, nitrate measurements at Hedenlundaån and Vadsbrosjön, and runoff measurements at Varbro (regulated). (From Ulén and Brink 1980).

Data Base and Study Area

The model was tested in the Ekenäs area in central Sweden, for which there are measurements from a small arable field, a small forested basin and from basins with and without lakes (Fig. 2). The area is situated 100 km southwest of Stockholm and has been thoroughly studied by Ulén and Brink (1980), Ulén (1982 and 1984). Specifics about the basin and available data are presented in Table 1.

The crop rotation at Flinkesta mainly comprises winter wheat, barley, oats and ley. At Flinkesta, Däntersta and Örbäcken there are both runoff and nitrogen measurements.

Table 1 – Area, percentage of forests, arable land and lakes in the studied areas

Name	Area	Forest	Arable land	Lake	Obser- vations
	km²	%	%	%	started
Flinkesta	0.066	0	100	0	1973
Däntersta	0.31	100	0	0	1976
Örbäcken	11.2	56	44	0	1976
Hedenlundaån	398	74	18	8	1979
Vadsbrosjön	418	72	20	8	1979
Stubbetorp	0.9	100	0	0	1983

For the larger areas of Hedenlundaån and Vadsbrosjön there are no runoff measurements, only nutrients measurements. SMHI has a runoff station at Varbro higher up in the watershed area, which has been used to calibrate the water balance model. One of the lakes upstream is regulated, but the correspondence between the calibrated runoff and the partially regulated runoff seems to be adequate for this study.

To test the dynamics in nitrate leaching from a forested basin data from another station – Stubbetorp – have also been used. This basin is situated 28 km south of Ekenäs.

Daily precipitation and temperature values were available from nearby meteorological stations. Monthly standard values (30 year mean) of Penman estimates of the potential evapotranspiration were taken from Eriksson (1981).

The arable land in the larger basins has been divided into two parts according to official statistics for the county of Södermanland. Ley and grazing lands form one group, and the other bigger group covers all types of cereals, potato and rape etc.

Criteria of Model Performance

The agreement between computed and recorded values can be illustrated by numerical criteria. One of the most simple and most popular of these is the sum of squares of the residuals

$$F^2 = \sum_{t=0}^{\tau} (Q_r(t) - Q_c(t))^2 \tag{10}$$

where

$Q_r(t)$ – observed discharge at time, t;
$Q_c(t)$ – computed discharge at time, t;
τ – total period of time.

If the initial variance is expressed as

$$F_0^2 = \sum_{t=0}^{\tau} (Q_r(t) - \bar{Q}_r)^2 \tag{11}$$

where

$\bar{Q}_r$ – arithmetic mean of the observed hydrograph over the time τ.

The proportion of the initial variance accounted for by the model can be expressed as

$$R^2 = \frac{F_0^2 - F^2}{F_0^2} \tag{12}$$

This criterion was defined by Nash and Sutcliffe (1970) as the efficiency of the model.

23

The values of R^2 will range from minus infinity to plus one, where plus one is representing a complete agreement between the two hydrographs or observed and calculated concentration transport values.

The R^2-criterion can not be compared when used in different catchments or during different periods of time. If the initial variance is low as usually for nitrate concentrations, small errors will cause low R^2-values.

In addition to the R^2-criterion of agreement, the coefficient of correlation is also used in this study.

Simulation of Runoff and Nitrate Concentration

The coefficients of the water balance model were first calibrated against runoff measurements at Flinkesta, Däntersta and Örbäcken. Thereafter the nitrate part of the model for forest and arable land was calibrated from the year when the measurements started until 1985 by a trial and error technique until acceptable agreement between observed and simulated concentration was achieved. The last two and a half years after 1985 were left as an independent test period.

The R^2-values for the homogeneous forest and arable basins are presented in Table 2 together with correlation coefficients for concentration and transport. There are also tabulated measured and calculated nitrate transport for the observed days.

Because of low variability in concentrations (especially for forest and mixed basins) and frequent questionable outliers we mostly trust subjective visual inspection of the results as the main tool for model calibration. The numerical criteria are, for example, very low for Stubbetorp for the period autumn 1985 to 1987, but for the period autumn 1984 to 1987 the values get somewhat better. This difference can not be seen by visual inspection and is not representative. This shows how difficult it is to use numerical criteria if the variability is low.

The calibrated model coefficients for different land use were used in the model for the mixed areas, and the simulated nitrate concentrations were tested against observations. The lake model was calibrated for the years 1980-1985 and verified for the period 1986-1988 for the larger basins. The numerical criteria are given in Tabel 3.

Forested Basins
Fig. 3 shows an example of results for Däntersta. The measurements of nitrate are made only once a month. We also tested the nitrate model in the Stubbetorp basin and got the results shown in Fig. 4 with the same model coefficients.

Arable Land
Fig. 5 shows the results from Flinkesta for the longest part of the calibration period when the field was covered with ley. The concentration drops to zero if the simulated runoff is zero.

Table 2 – Numerical criteria (explained variance, R^2, correlation coefficients, C, and nitrate transport volumes) for homogeneous forest and arable basins.

Area/ land use	Period	Runoff R^2	Nitrate concentration R^2		Nitrate transport R^2		Nitrate transport in kg/ha for the observed days obs.	calc.	No. of obs. days
			R^2	C	R^2	C	obs.	calc.	days
Däntersta forest	1977-84 calibration	0.67	0.07	0.54	-0.03	0.18	0.12	0.17	78
	1985-1.7.88 verification	0.71	0.00	0.18	0.09	0.49	0.09	0.03	36
Stubbetorp forest	1.8.1985-30.9.87 calibration	0.80	-0.81	0.49	-2.35	0.12	0.3	0.5	95
	1.9.1984-30.9.87	–	0.09	–	0.51	–	–	–	197
Flinkesta arable land									
cereal	1.10.1973-1975	0.46	0.49	0.87	0.68	0.32	2.1	1.1	30
	1.10.1980-1.9.85 calibration	0.45	-0.22	-0.13	-1.47	0.16	10.1	14.4	123
ley	1.10.1977-31.12.80 calibration	0.66	-0.53	0.06	0.54	0.18	0.9	0.8	76
	1.9.1985-1.9.88 verification	missed	-0.28	0.02	–	–	–	–	–

Table 3 – Numerical criteria (explained variance, R^2, correlation coefficients, C, and nitrate transport volumes) for mixed basins.

Area/ land use	Period	Runoff R^2	Nitrate concentration R^2		Nitrate transport R^2		Nitrate transport in kg/ha for the observed days obs.	calc.	No. of obs. days
		R^2	R^2	C	R^2	C	obs.	calc.	days
Örbäcken mixed forest and arable land	1984-1.8.87 verification (not calibrated)	0.65	-0.02	0.41	0.30	0.17	0.56	0.41	44
Heden-lundaån with lakes	1980-1984 calibration[1]	–	0.54	0.74	0.64[2]	0.16	0.14[2]	0.12	59
	1985-1.9.88 verification	–	0.67	0.84	0.86[2]	0.20	0.11[2]	0.09	44
Vadsbrosjön with lakes	1980-1984 calibration[1]	–	0.41	0.65	0.86[2]	0.16	0.13	0.11	49
	1985-1.9.88 verification	–	0.74	0.89	0.81[2]	0.21	0.11	0.09	44

1) Only the lake routine is calibrated.
2) As there are no runoff measurements, the simulated runoff has been used.
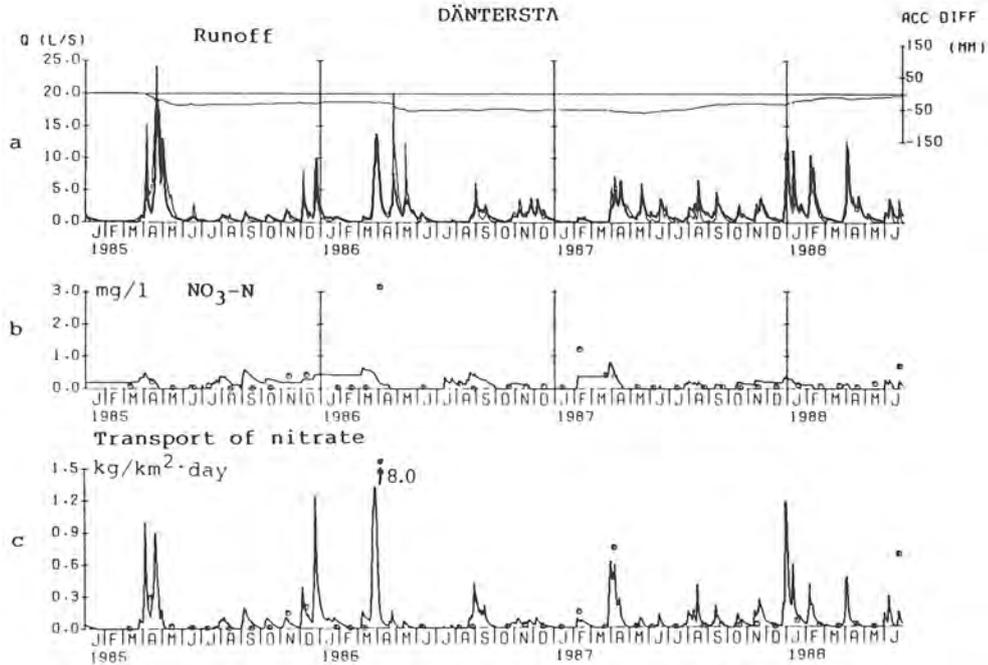
Fig. 3. Examples of runoff, nitrate concentration and transport simulation for the Däntersta basin (0.31 km², 100 % forest). a: Thin line = recorded runoff, thick line = computed runoff. Acc. diff. represents the cumulative difference between computations and observations. b and c: Dots = concentration of nitrate and the observed transport respectively, line = simulated concentration and transport respectively.

Mixed Areas

Fig. 6 shows results from Örbäcken – a mixed area without lakes. The water balance model is calibrated, but not the nitrate subroutines. The coefficient sets for forest and arable lands (cereal and ley areas) are taken from Däntersta and Flinkesta respectively. These two small areas do not belong to the Örbäcken basin but are of the same character and situated nearby.

Downstream this station there is an input of waste water from a village of about 100 persons. Nitrate measurements downstream the outlet show only very small contribution to the nitrate leaching.

Basins with Lakes

For the basin of Hedenlundaån and Vadsbrosjön the same model coefficients for nitrate turnover were used as in the Örbäcken case, but with actual land use.

Fig. 7 presents results from some of the years. Graph 7b shows the computed nitrate concentration by the model with lake processes included. There is also shown the loss of nitrate in the lake in the same graph.

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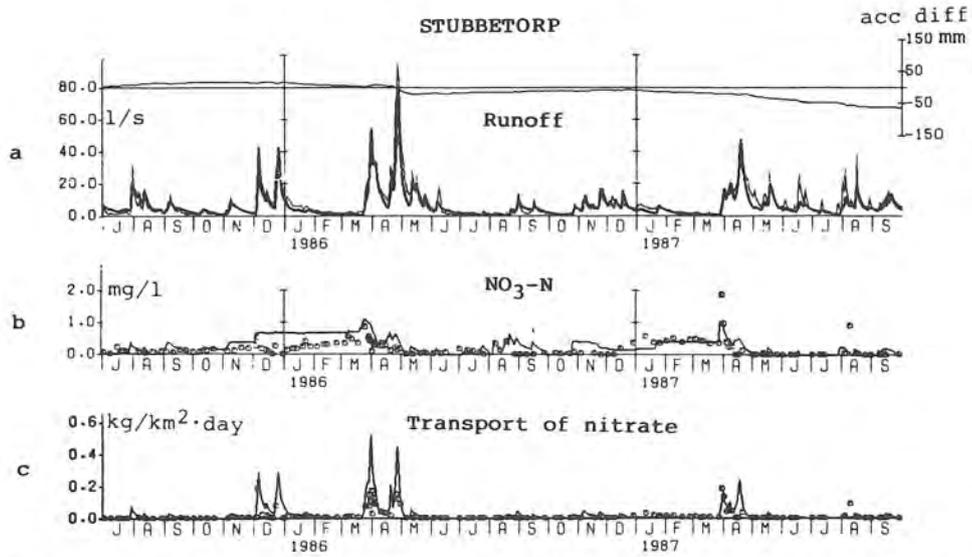
Fig. 4. Examples of runoff, nitrate concentration and transport simulation for the Stubbetorp basin (0.9 km², 100 % forest). a: Thin line = recorded runoff, thick line = computed runoff. b and c: Dots = observed nitrate concentration and observed transport respectively, line = computed concentration and transport respectively.
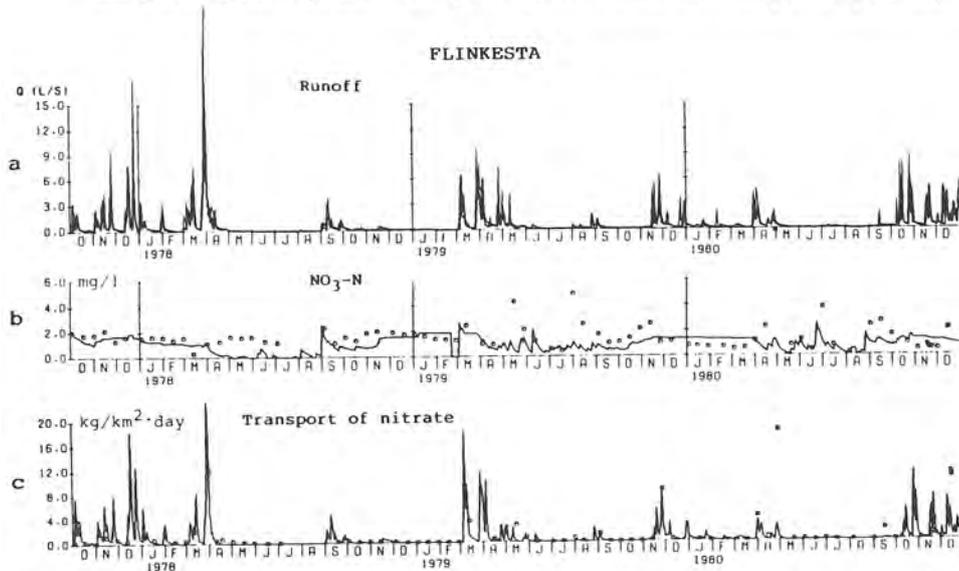
Fig. 5. Example of runoff, nitrate concentration, and transport simulation for the Flinkesta basin (0.06 km²) when it is covered of ley. a: Thin line = recorded runoff, thick line = computed runoff. b and c: Dots = observed nitrate concentration and observed transport respectively, line = simulation. The period belongs to the calibration period.

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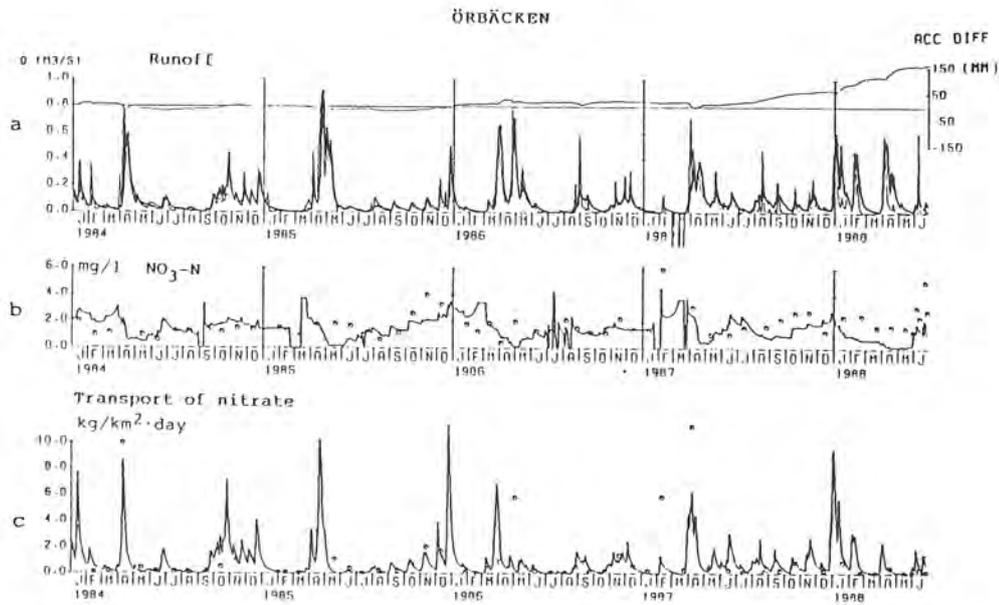
Fig. 6. Example of runoff, nitrate concentration, and transport simulation for the Örbäcken basin (11.2 km²). Örbäcken is a mixed area with no lakes. a: Thin line = recorded runoff, thick line = computed runoff, acc. diff. = cumulative difference between computed and observed runoff, b and c: Dots = observed nitrate concentration and observed transport respectively, line = simulated concentration and transport respectively. The coefficients of the nitrate model are taken from the calibration of the forest and arable land model.

Fig. 7c shows results from the simplified model approach, where typical nitrate concentration for each month from areas with different land use with the actual proportion are used as constant monthly indata to the lakes instead of the more sophisticated model. The same lake model is used as above. As can be seen from the results, this simple model performs almost as well as the more sophisticated model.

In Fig. 8, finally, are shown the complete calibration and verification periods for the lake model for the two sampling points, Hedenlundaån and Vadsbrosjön, with the complete model.

Comparative Calculations of Nitrogen Transport

In order to analyse the uncertainty in the transport calculations further we calculated the nitrate transport in three different ways at Hedenlundaån for the years 1980-87 and for the mean of all these years. In all these calculations the computed runoff from the PULSE-model is used, as there are no measurements available.

28
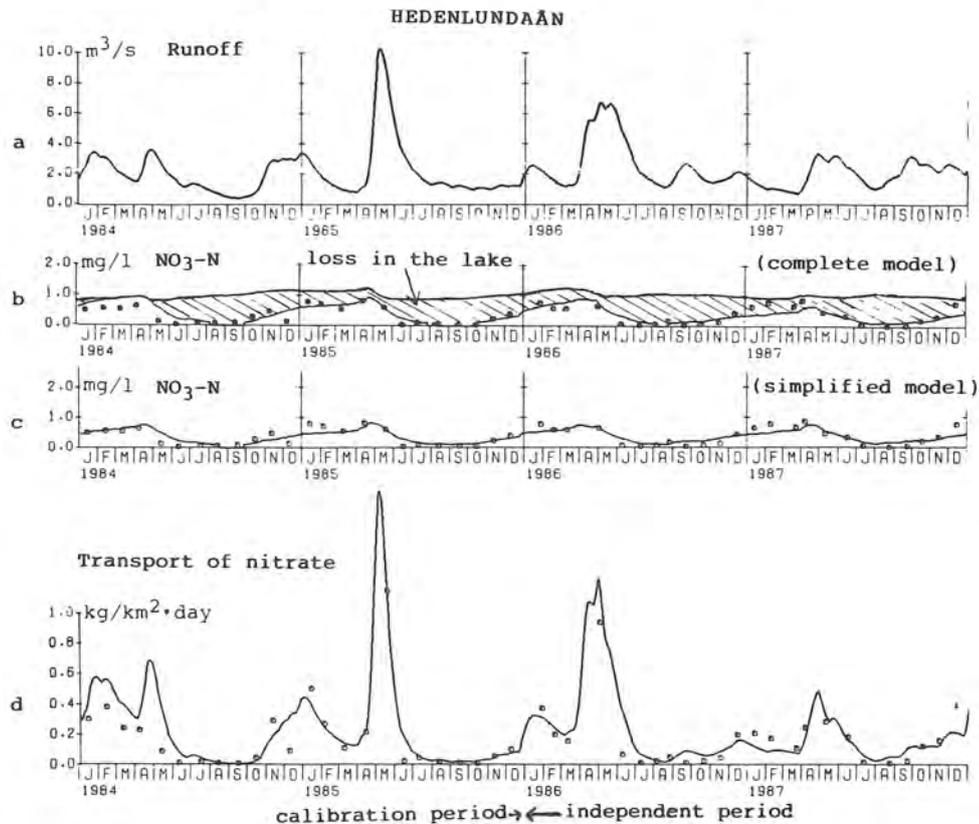
Fig. 7. Example of runoff, nitrate and transport simulation for Hedenlundaån (398 km²). a: Thin line = recorded runoff, thick line = computed runoff. b to d: Dots = observed concentration of nitrate and transport resp., lines = simulations. Fig. 7b is the simulated concentration with and without uptake of nitrate in the lakes with the complete model, and Fig. 7c with the simplified model. Fig. 7d demonstrates the computed nitrate transport with the complete model.

By the first method the transport is calculated as the product of the mean monthly runoff and the measured nitrate concentration during the same month. There can be problems due to rapid changes in runoff and nitrate concentration. For example, the nitrogen concentration in the lake normally drops between April and May. At the same time there can be a drastic drop in runoff after the spring flood, and consequently depending on the time when the nitrate measurements are made, there may be an over- or underestimation of transport in May.

By the second method the transport is a calculated by the model day by day according to the curves shown in Figs. 7 and 8.

The third method uses the simplified model with monthly standard nitrate concentration from the forest area and the two different types of arable land in the
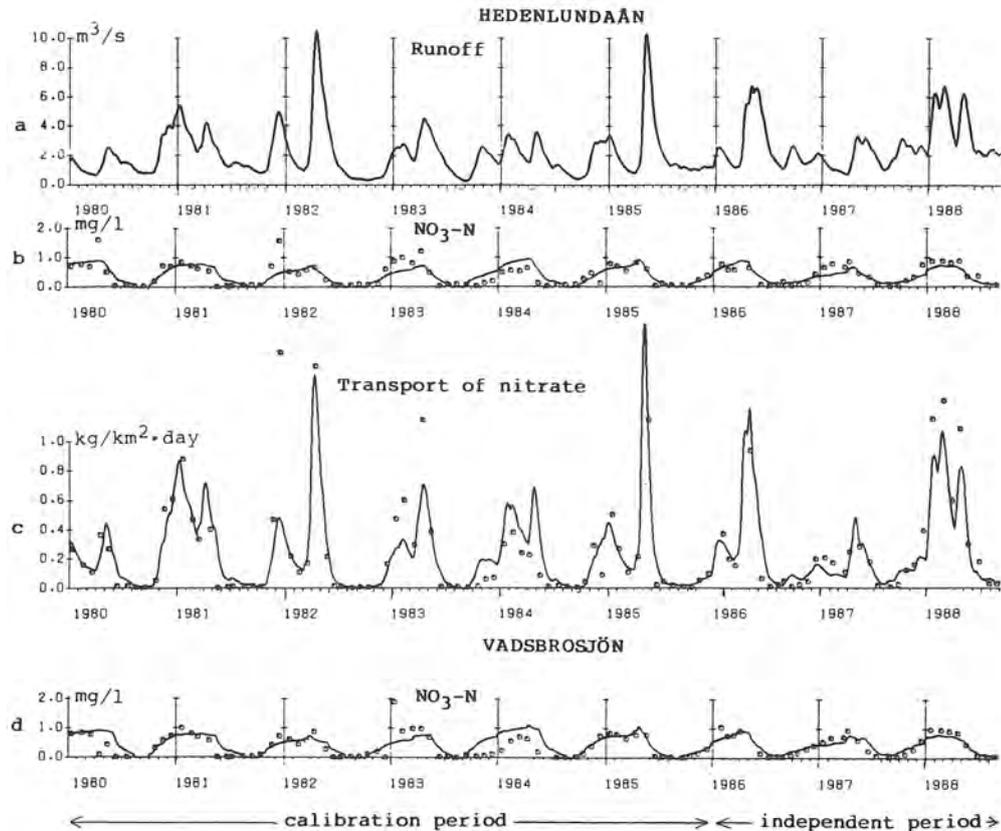
Fig. 8. Simulation of runoff (a), nitrate concentration (b) and transport (c) from Hedenlundaån and nitrate concentration from Vadsbrosjön (d) in the whole calibration and independent period. Line = simulation, dots = observations.

actual proportion in the mixed area of Hedenlundaån. These concentrations were taken from all measurements at Däntersta and Flinkesta.

The results from the three transport calculations are shown in Fig. 9.

As mean value for all eight years the calculations based on monthly mean values and on the complete model give transport values of 0.83-0.90 kg/ha year. The differences between the annual values are less than 15 % except for 1984.

Discussion of the Results

There is normally much greater variability in runoff than in nitrate concentrations. This implies that runoff data or a good water balance model are crucial for estimations of losses of nitrogen from a basin. The greatest part of the transport occurs during short periods with high flow in particular from basins without lakes.

Runoff and nitrate concentrations are damped in lakes. This together with the
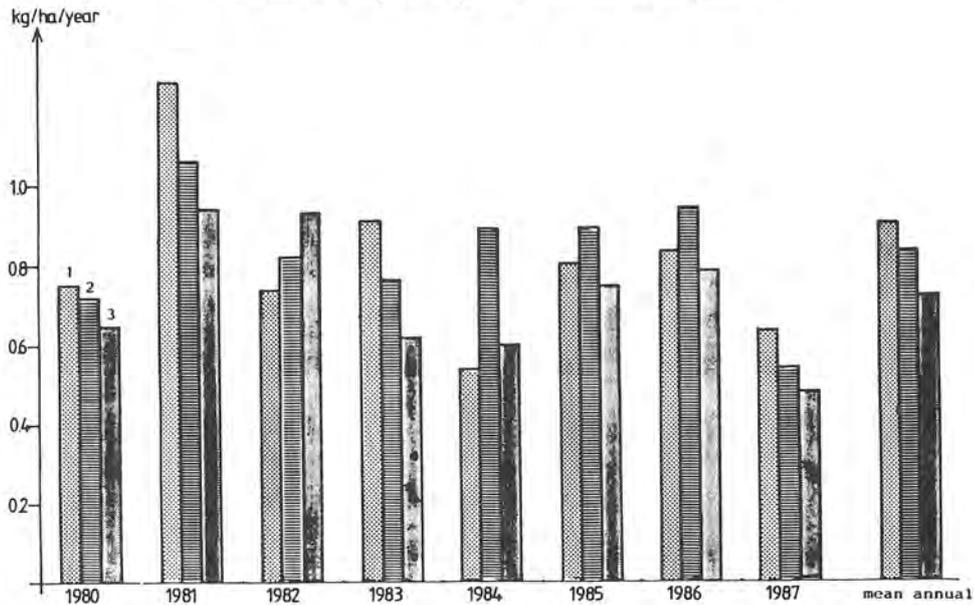
Fig. 9. Simulated nitrate transport from Hedenlunaån computed in three ways. The simulations are from left to right: mean monthly runoff * observed nitrate concentration (1), simulation on daily basis by the complete model (2) and simulation on daily basis by the simplified model (3).

nitrate loss in the lake in summer results in the greatest nitrate leaching from these basins in the period from late autumn to spring.

The model sometimes showed large deviations from the observed nitrate concentrations. As an example can be mentioned simulations for the Flinkesta field covered with ley in the spring and the early summer of 1978 (Fig. 5). It happened during low flow, so it did not influence the yearly nitrate transport much.

In the mixed areas the coefficients in the nitrate model are taken from homogeneous basins. The simulations of nitrate concentration and transport for Örbäcken show that they are of the right order of magnitude but with some deviations until the summer of 1987 (exampled for the period 1984 to 1988 in Fig. 6). From the summer of 1987 the computed concentrations are too low. The computed concentrations from the forest of Däntersta at the same time fit quite well with the observed ones, but those from the ley field at Flinkesta are a little too high. There seems to be an underestimation of nitrate leaching from fields with cereals this autumn and winter. The uptake by cereals has probably been overestimated in the model this unusually wet summer, and thus the nitrate storage has been underestimated after the summer.

The simulated runoff at Örbäcken is too high in 1988 compared to the observations probably due to some problems in the observation series, as these deviations between simulation and observation are not seen in the smaller areas. Before 1984

31

there were also problems with the runoff observations.

There is no sink in the present model for denitrification losses in the brooks. The importance of this process is disputed. Hill (1979) found a 5-6 % nitrate removal by denitrification of annual export of total nitrogen from a river basin. Danish researches in small rivers (Jeppesen *et al.* 1987) show reduction of 10-30 % of nitrate in the summer but during the year less than 2 % of the total transport. Any denitrification losses in brooks have been ignored in the model, as it can not be supported by any measurements in the actual area.

The sensitivity of the nitrate model for arable land is discussed by Bergström *et al.* (1987). It was shown that there is a considerable amount of feedback in the nitrate modelling routines, which moderates the effect of perturbations of the coefficients. The sensitivity analysis also showed that the interaction between the water balance model and the model for nitrogen turnover was not alarming. The model was most sensitive to the routine methods for calculation of nitrate uptake by plants.

The problem in this project is the transfer of information from small homogeneous areas to larger mixed areas. The first agricultural nitrate model had to be simplified to reduce the amount of coefficients, while at the same time the indata (amount of fertilizer *etc.*) are more uncertain. Still there are many empirical coefficients left, of which some are sensitive (as coefficients for uptake of crops, mineralization and leaching), and there is a great risk for overfitting of the model. As this method in first hand is thought as a calculating tool of nitrate transport from larger basins with some measurements in the basins, it can still be useful.

As is seen from Fig. 7, the nitrate turnover is strongly affected by the lakes. The nitrate concentration is smoothed out, and in summer uptake, sedimentation, and denitrification in the lakes consume nearly all inorganic nitrogen from the basin of Hedenlundaån. This means about 50 % loss of inorganic nitrogen per year. The retention time of water in the lake is important to give ample time for the processes in lake to occur. The area of the lake is known, but the depth of the mixed water body of the lake is a coefficient in the water balance model. Tests show that the actual mean depth can not be used when there is thermoclines in the lakes. The coefficient C_{lake} is the most sensitive coefficent for the transport calculation from lakes.

The main drawback of this type of conceptual models is the large number of empirical coefficients, which have to be found by calibration. In this study this has been accomplished by the use of small, more homogeneous basins, but the uncertainty is still considerable. So far the coefficients have to be considered both site specific and model specific and transfer to other basins has to be made carefully. Nevertheless it is surprising how well such a transfer succeeded in the Örbäcken basin.

The simplified model with standard loss values for different land use has the most immediate potential in Sweden. Concentrations of nitrate in watersheds and

drains from arable land and forest are measured in experimental fields spread over the country by, for example, the Swedish University of Agricultural Sciences (Gustafson 1987). Other sources of data are also available. Especially in large basins with lakes this simplified model seems to be a good tool to estimate nitrate transport out of the watershed.

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VI

Modelling Nitrogen Transport
from a Basin of Mixed Land Use
Example from the Lake Ringsjön Drainage Area, Sweden

by

Brandt, M.

MODELLING NITROGEN TRANSPORT
FROM A BASIN OF MIXED LAND USE -
EXAMPLE FROM THE LAKE RINGSJÖN DRAINAGE AREA, SWEDEN

by

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ABSTRACT

A method of computing nitrogen leaching from an area with mixed land use is described for the Lake Ringsjön drainage area (about 400 km^2) in southern Sweden. The leaching is computed as the product of runoff, calculated by a hydrological model, and typical nitrate and organic nitrogen concentrations from different land use categories. These categories are classified by Landsat satellite images.

The computed nitrate and organic nitrogen transport are checked against measurements with a sampling frequency of once a week to once a month. It is shown that lakes have a great impact on nitrogen transport in a river system. 70 to 85 % of the nitrate transport to Lake Ringsjön is retained by biological and physical processes in the lake. Retention of total nitrogen in the lake is 45 - 60 % (lake percent 10%). The seasonal and interannual variations are great and reflect different hydrological conditions.

INTRODUCTION

The Lake Ringsjön is an important water resource for the Scania region in southern Sweden. It is used for many purposes: fishing, drinking-water, recreation and as a recipient for sewage water. Its water quality has, however, degraded (1,2) in recent times. Eutrophication causes algae bloom, usually blue-green algae, aquatic plant growth, oxygen depletion and fish kills. It threatens the æsthetic quality of water and some algae can secrete toxic substances. The establishment of advanced wastewater treatment for municipal sewage in the Ringsjön area in the 1970's had little or no effect on the water quality in the lake (3).

The drainage area of Lake Ringsjön consists, as most larger basins, of a mosaic of different land uses - arable land, forests, villages and lakes. As the nutrient transport is a mixture of leaching of nutrients from all these different parts, knowledge

of each contribution is of vital importance when calculating mass transport. This is also essential when deciding counteractions to reduce nutrient pollution. The proportion of arable land in the area around Lake Ringsjön is high (34 %), and non-point pollution from agricultural areas is often regarded as one of the main sources of pollution as is also atmospheric outfall (1).

For more than ten years the water quality in Lake Ringsjön and in some of the tributaries has been monitored with a frequency of up to once a week. This type of monitoring entails the problem of interpolation of the runoff and the nutrient concentration between observations, and identification of sources. In particular the runoff from an area without lakes can fluctuate a lot from day to day, and the use of weekly measurements can introduce great uncertainty in transport values during high flows. It is also difficult to estimate the transport of nutrients from ungaged tributaries.

In this project the proportions of different land use in all sub-basins have been estimated from Landsat satellite images and a mathematical hydrological simulation model, the PULSE model, has been used to estimate water discharge from these sub-basins.

Typical monthly mean values of nitrogen concentrations as nitrate and organic nitrogen in drainage water from different land uses have been taken from monitoring programmes in small homogeneous areas. The concentration of ammonium is low and is included in the observations of organic nitrogen. The daily leaching from each land-use area in a tributary is computed as the product of runoff from the area calculated by the hydrological model and the corresponding typical concentration and added up to obtain the total nitrogen transport.

Finally lake retention processes are modelled by a simplified routine based on the change of phytoplankton quantity in the water body and an exponential decay function.

SITE DESCRIPTION AND DATABASE

The location of the Ringsjön drainage basin and the small homogeneous areas used to get typical nitrate and organic nitrogen concentration from different land uses are shown in Figure 1. Table 1 presents some background data for the small areas.

Table 1. Small homogeneous areas, size, land use, and observation period.
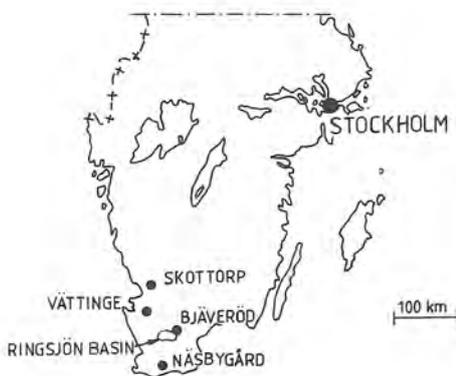
Name	Area ha*	Land use	Observation period
Näsbygård	35.7	Arable land different crops	1973 -
Vättinge	22.2	- " -	1976 -
Skottorp	14.5	- " -	1976 -
Bjäveröd	84	Forest	1983 -

Figure 1.
Geographical location of the Lake Ringsjön drainage basin and the small homogeneous basins that provided typical nitrogen concentrations.

* 1 hectare = 10^{-2} km²

The Lake Ringsjön basin (395 km²) comprises several sub-drainage basins. Sampling sites are shown in Figure 2 and characteristics are presented in Table 2. The landplains in south of the Lake Ringsjön basin include pastures, meadows and agricultural areas while the northern higher parts are dominated by forests (Figure 3). Nitrogen leaching from arable land in the area studied consists of about 90 % nitrate, but from forest areas the corresponding figure is only about 15 %.

4
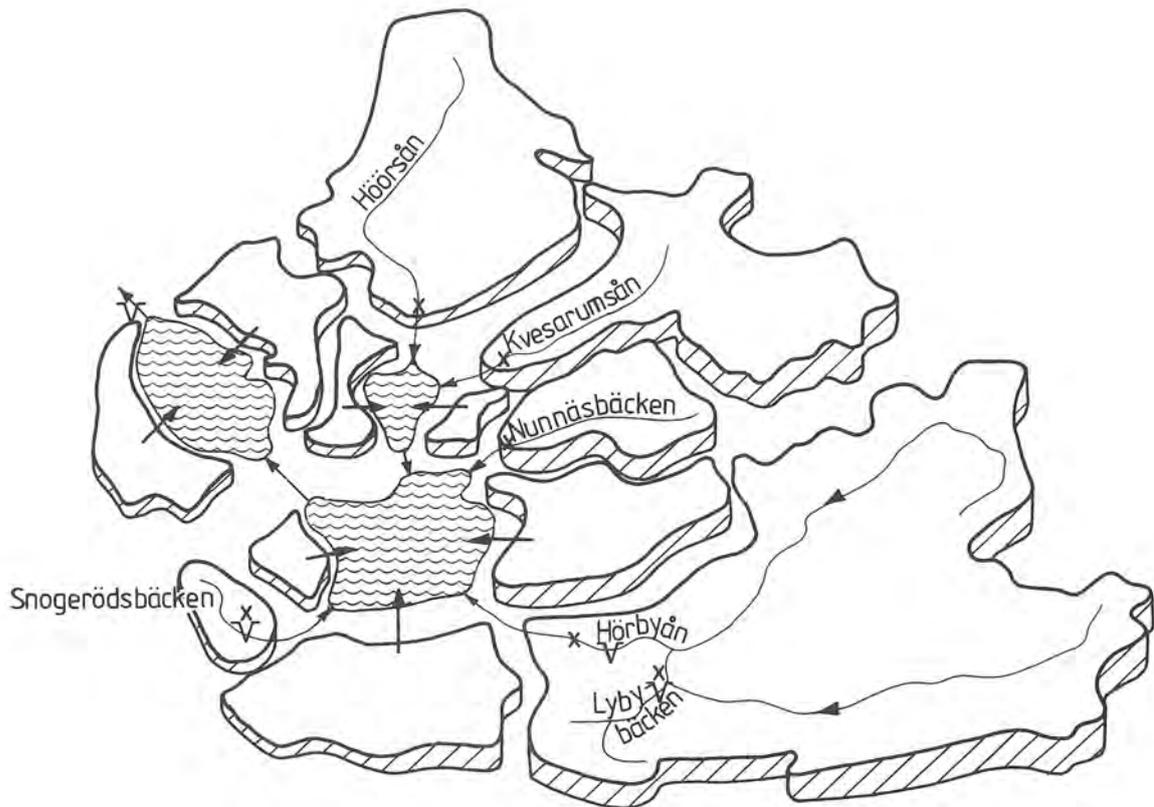
Figure 2. Sub-basins of the Lake Ringsjön basin. x = sampling points, ∨ = runoff measurements.

Table 2. Sampled sub-basins, the Lake Ringsjön drainage area and land use.

Name	Area km²	Land use Forest %	Arable land %
Höörsån	53	45	18
Kvesarumsån	39	48	15
Nunnäsbäcken	11	61	7
Hörbyån incl. Lybybäcken	145	30	40
Lybybäcken	14	2	84
Snogeröds- bäcken	7.3	8	76
Ringsjön	395	30	34
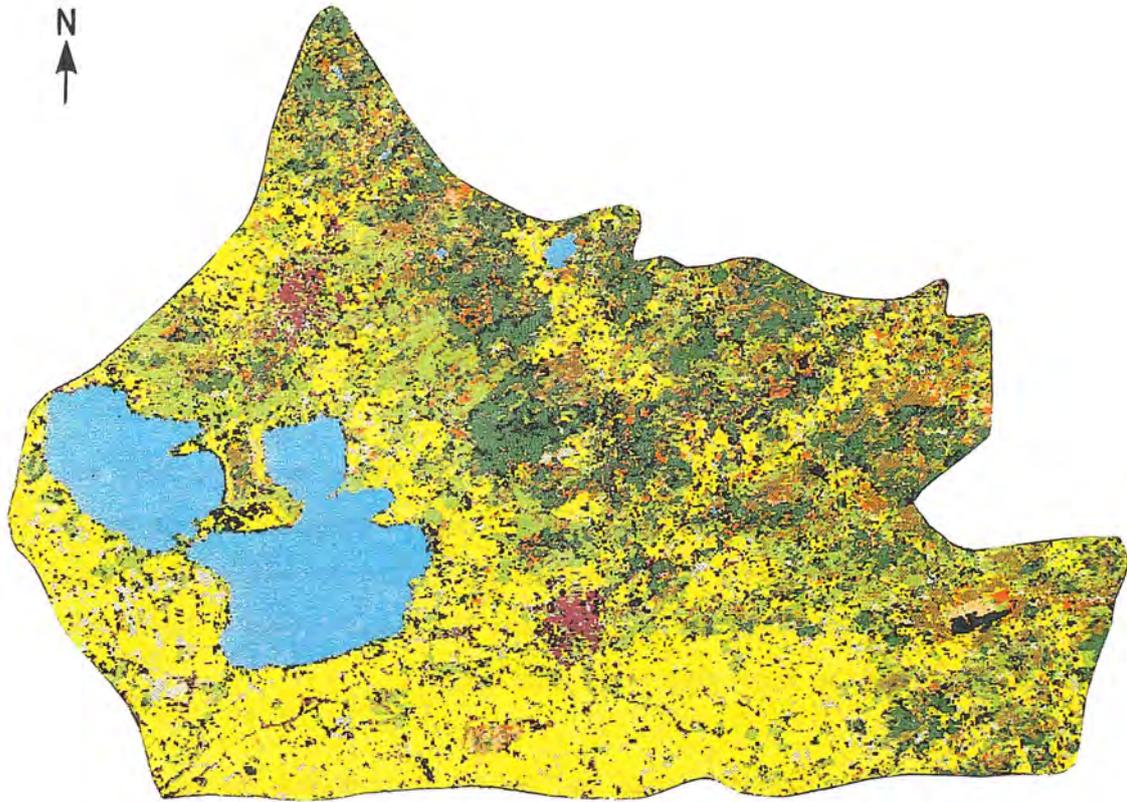
Figure 3. Land use classified by Landsat satellite images (1984-04-27). Blue = lakes, yellow = arable land, dark green = coniferous forest, bright green = deciduous forest, brown = beech forest, red = clearcut, pink = swamp, white = open field, not arable land, purple = settlement, villages, black = unclassified.

All sub-basins to Lake Ringsjön have been land-use classified by Landsat satellite images into wooded areas (coniferous and deciduous forests), felled areas, wet swamps, arable land, open fields (as meadows, farm yards, etc.), urban areas and unclassified areas. Data from the Thematic Mapper (TM) on board Landsat 5 taken 1984-04-27 have been used. Typical signatures from all TM wavelength-bands except thermal IR (TM 6) for every single land use of interest were picked out, and on the basis of the signatures a maximum-likelihood classification was made. A field survey showed a high degree of correspondence. This method gives a better estimate of actual used arable land than the usual map analysis on a 1:50 000 scale. The traditional map survey overestimates the arable fertilized land with too high computed leaching as a consequence.

Arable land is divided into fields with different kinds of seed, potatoes etc. and with ley according to a local study (4). From Landsat images it can be possible to distinguish between fields with and without crops during the winter, if the image is taken late in the autumn. But at the same time the angle of sun is lower then and there can be problem in separating other land uses.

Point sources such as manure stacks and outlets from farms are of little significance for nitrogen leaching and it has not been found worthwhile to introduce them in the present study. They might be more important if, for example, phosphorus is studied. Records of nitrogen concentrations from larger sewage plants are available and monthly mean values are introduced into the model.

For some of the tributaries water samples have been taken once a week. The weekly samples have been poured together and analysed once a month. In 1988 the samples were analysed weekly.

Phytoplankton observations in the water bodies of the lakes are available with a frequency of once a month.

Daily runoff measurements are availably for Heåkra and Lyby - both in Hörbyån -, at Snogerödsbäcken and at the outlet of Lake Ringsjön. The outlet is regulated. At Höörsån, Kvesarumsån, and Nunnesån the waterlevel is recorded once a week and

the discharge is calculated. 75 % of the drainage area of Lake Ringsjön has been covered by nitrogen measurements, but only 15 % by runoff measurements.

The input data to the PULSE model, daily precipitation and temperature values, are available from nearby meteorological stations. Monthly standard values (30 year mean) of Penman estimates of the potential evapotranspiration are used (5).

METHODS
The water balance model

The hydrological PULSE model (6) has been used to simulate the runoff from the different sub-basins to Lake Ringsjön and also from the outlet of Lake Ringsjön. The model is a modification of the HBV runoff model (7, 8) which is used in several countries for hydrological forecasting. The model runs with daily time steps with precipitation and mean air temperature as input together with monthly standard values of potential evapotranspiration. Runoff simulation involves three steps:

- snow accumulation and ablation;
- soil moisture accounting;
- generation of runoff and routing of the hydrograph through the river system.

The model controls the water balance by a set of empirical coefficients, for example: threshold temperature to decide when snow begins to accumulate and melt, maximum amplitude of soil moisture storage, recession coefficients for water in the saturated zone. These coefficients are found by calibration if runoff records are available - otherwise regional standard values are used.

The PULSE model can be broken down into sub-models, which makes it suitable for applications to areas with different land use. If the sub-model structure is defined by the outlet points of lakes, the hydrological effects of lake storage can be considered in a more physically correct way, and the number of coefficients that have to be calibrated is reduced (9).

The nitrogen model

A number of nitrogen models has been developed (see for example 10, 11, 12, 13, 14), but few have been quantitatively validated against measurements. Many of them are process-oriented and require a lot of data, and some are specific for a small basin or an arable field. There still exist uncertainties with respect to several important properties governing nitrogen turnover, such as mineralization, denitrification, nitrogen fixation, and uptake by crops which hinder further development of advanced nitrogen models.

We have earlier developed and tested a nitrate turnover model for arable land (15) and for mixed land use (16). The nitrogen turnover model only accounted for the most important processes but still resulted in too many empirical coefficients if applied to larger areas. It was further difficult to satisfy its data demand, for example the amount of fertilizer used and atmospheric outfall. Crop management and fertilizer use change from area to area and from year to year and are often not known for a basin without extensive research.

We have now turned to a simpler approach based on typical monthly concentrations values obtained from monitoring programmes in small homogeneous areas (16). These small areas have different land use, such as different kind of cereal, ley, or forests. The basins are situated outside the Lake Ringsjön basin but still thought to be representative. There are difficulties in getting representative concentrations from villages and settlements, open unfertilized fields and swamps, so they are given the same typical values as wooded areas in this study.

The nitrogen leaching from each land-use area in a tributary is calculated as the daily product of computed runoff from the area and corresponding typical nitrate and organic nitrogen concentrations and added up to obtain the total nitrogen transport.

Lake processes

The water quality in a lake is dependent on a large number of factors (17), such as:

- influent quality and mixing of water in the lake,
- internal loading in the lake,
- physical and chemical processes in the lake,
- biological processes.

In lakes there is a high micro-organism activity which consumes nutrients. Algae growth is a function of light, temperature and nutrients. The algae die and sink to the bottom. Some sediment can be resuspended, and there may be denitrification at the bottom. The organic nitrogen can also be transformed to inorganic nitrogen in the water.

Circulation of the lake water is essential for the mixing of water and nutrients, and turnover time affect the possibility of the water quality to change due to physical, chemical, and biological processes.

Several mathematical models of ecological processes in lakes have been described (17, 18). The models need a great number of variables and require a database that is normally not available for both calibration and verification (19, 20). Therefore a simple empirical approach is used in this study to describe the nitrate reduction and the organic nitrogen increase in the lake in summer. The increase of organic nitrogen is assumed to be proportional (positively) to the change of phytoplankton quantity in the lake water (mean for several years). As some of the nitrate is transformed into organic nitrogen and some is used as energy, the reduction of nitrate is assumed as proportional (negatively) to the phytoplankton quantity in the model, but it has a higher proportionality coefficient than for organic nitrogen. This is, however, not enough to describe the nitrate reduction in the lake. Sedimentation and denitrification losses of nitrate are also included by a simple exponential decay function.

Existing data in the area do not permit the use of a more complete nitrogen lake

model. If the use of the model is restricted to nitrogen transport from a lake in a larger watershed to the sea, the method is, however, appropriate provided the conditions can be considered as stationary. The model should not be used to calculate the exact proportion of nitrate transformed to organic nitrogen, sedimentation etc., as the modelling of the different processes can not be verified and model interaction can be expected.

RESULTS

The coefficients of the water balance model were first found by calibration of the water balance model against runoff measurements at Heåkra and Lyby - both in Hörbyån - and at Snogerödsbäcken. These coefficients were used for all sub-basins and tested against the weekly measured runoff. Agreement between computed and observed runoff once a week was fairly good. The typical nitrate and organic nitrogen concentration from different land-use areas were then introduced into the model. In Figures 4 - 5 are shown results from two of altogether five modelled years for two of the sub-basins investigated. The runoff (measured and/or computed) plus nitrogen transport are also presented. It is important to note that these results are independent of model calibration.

Nitrogen transport from all sub-basins to Lake Ringsjön were calculated in the same way. The retention of nitrogen in the lake were computed by the lake routine in the model and the results are shown in Figure 6. The lake model is calibrated for the years 1984 to 1987 and is verified for 1988.

In Figure 7 is shown the yearly runoff and nitrate transport to and from the Lake Ringsjön.

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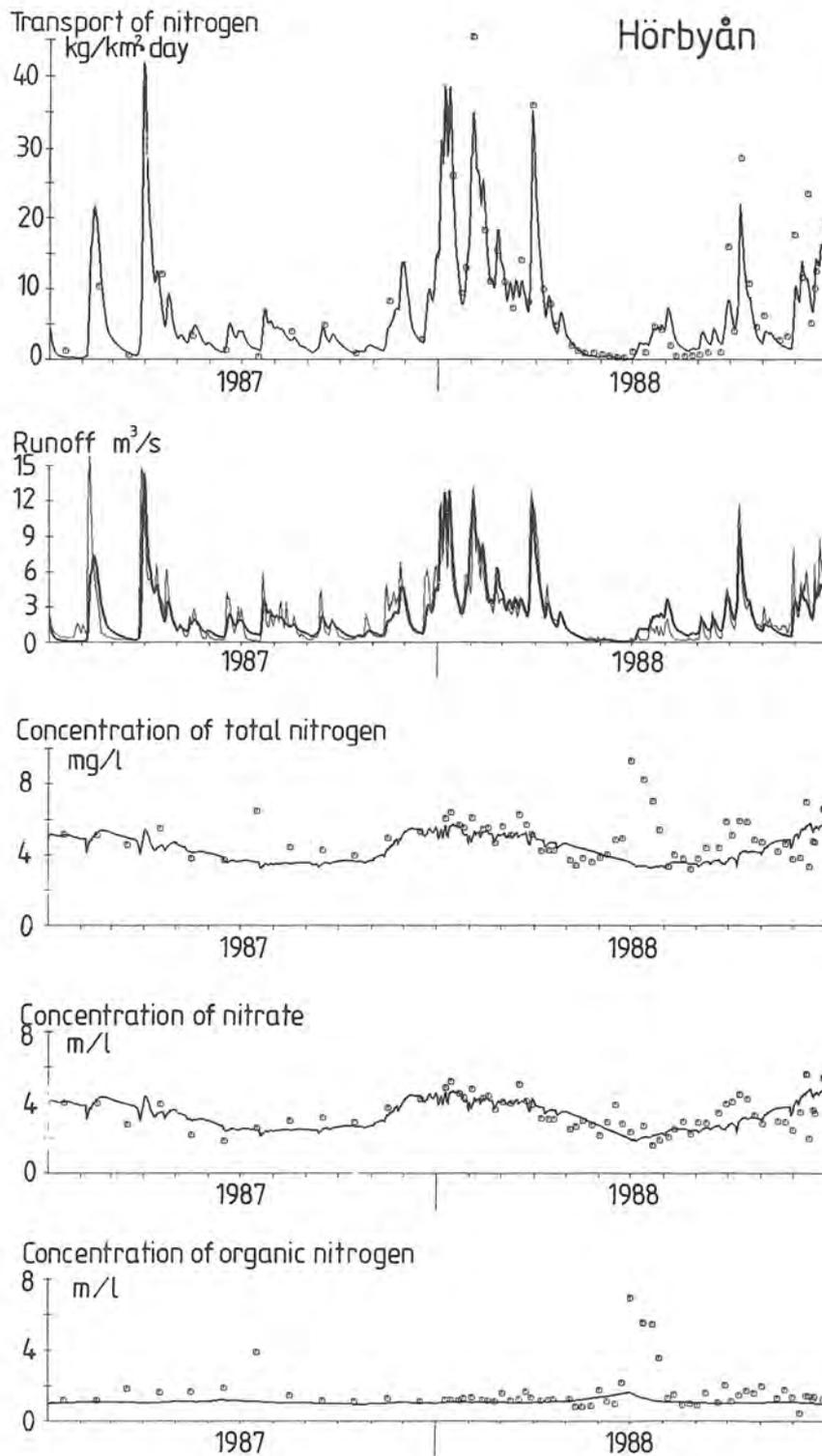
Figure 4. Results from the independent test period at Hörbyån (145 km², 40 % arable land). Thin runoff line = recorded, thick line = computed runoff, full line = computed concentration and transport, dots = measured concentration and transport.
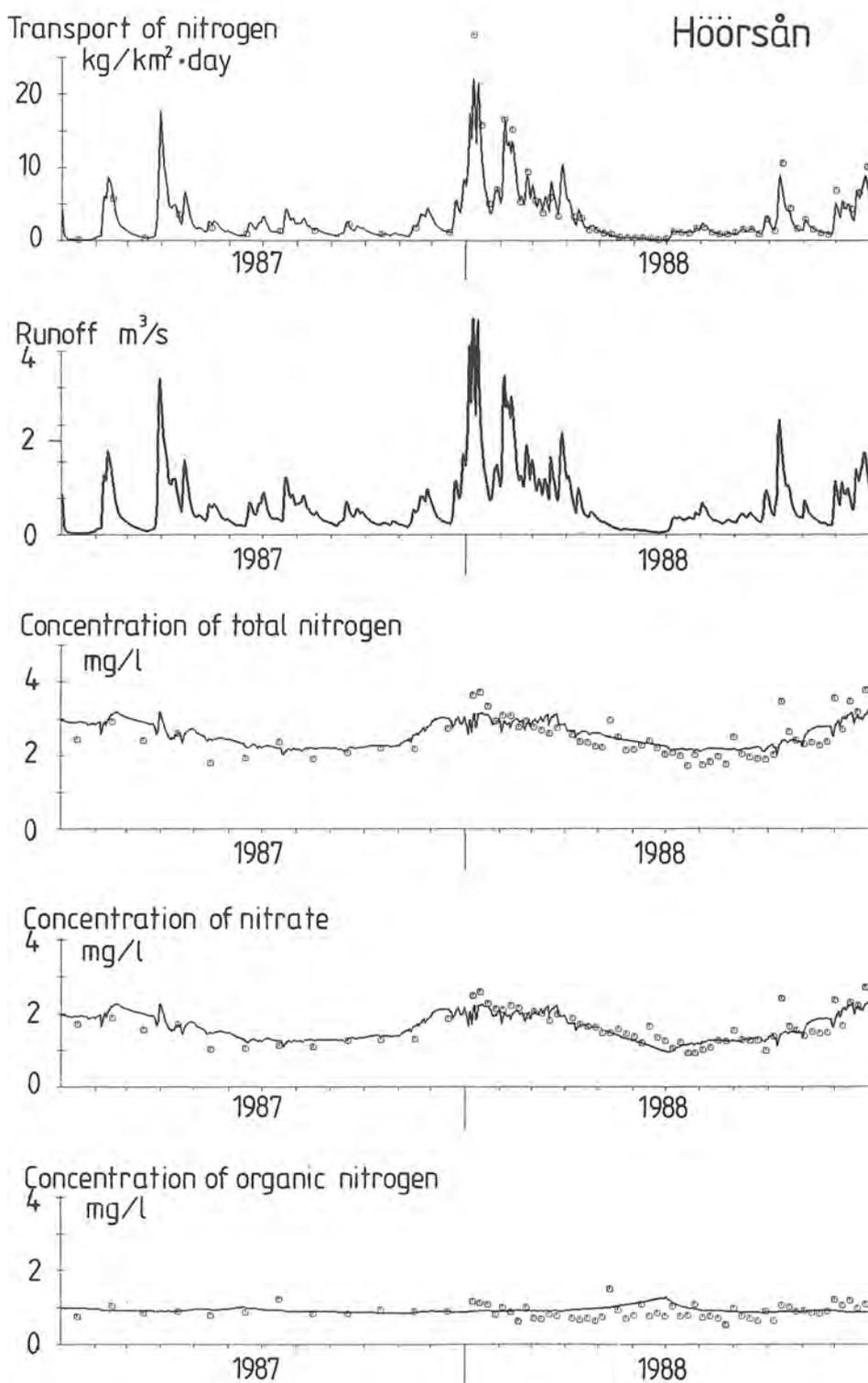
Figure 5. Examples of uncalibrated model simulation at Höörsån (53 km², 18 % arable land). Thick line = computed runoff (no continuous runoff measurements), full line = computed concentration and transport, dots in concentration diagrams = concentration measurements, dots in transport diagrams = transport calculated from measured concentration and computed runoff.
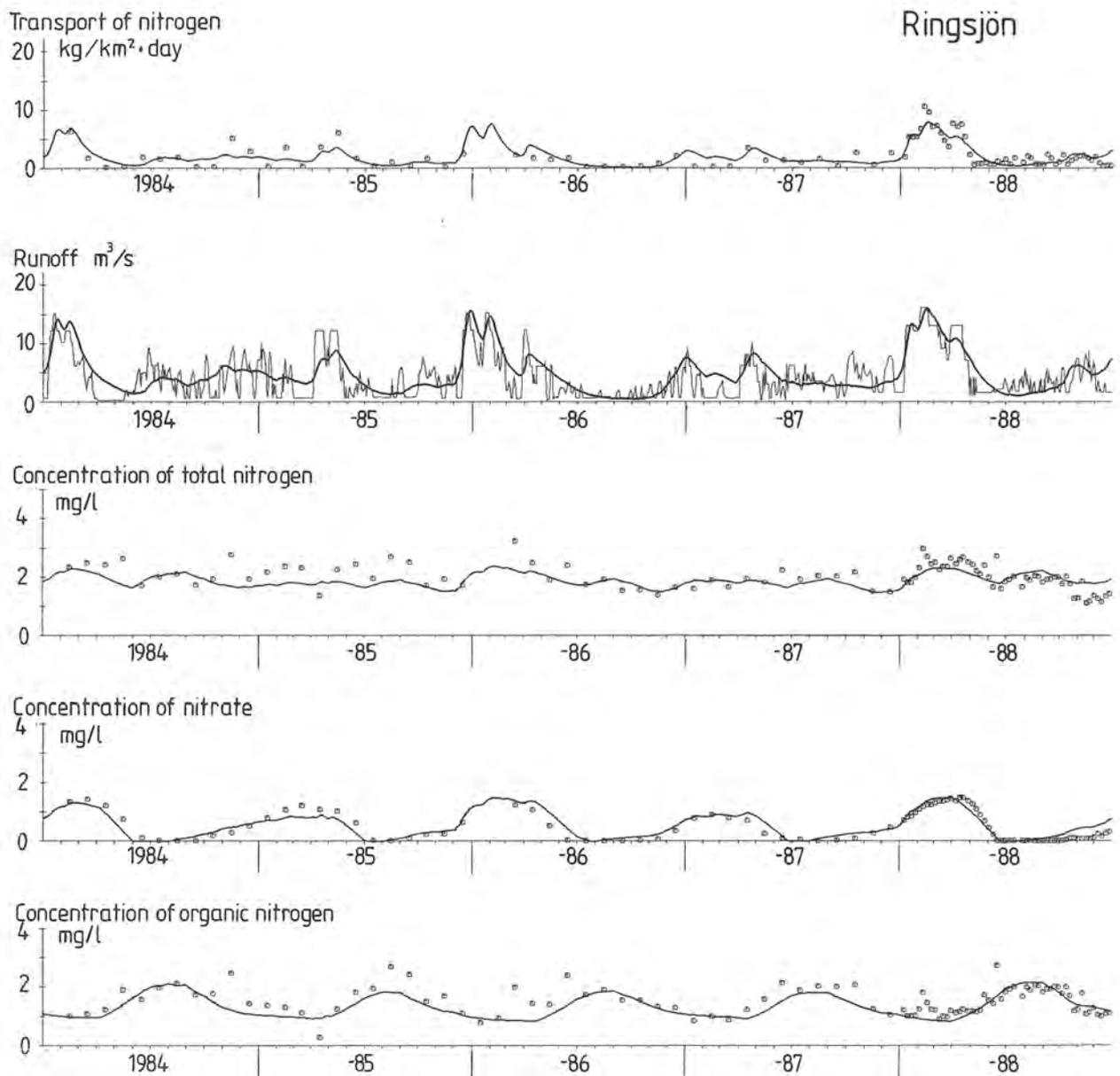
Figure 6. Results from the Ringsjön outlet (395 km²). Thin runoff line = recorded (regulated), thick line = computed runoff, full line = concentration and transport, dots = measured concentration and transport respectively. The lake model is calibrated for 1984-87 and verified for 1988.
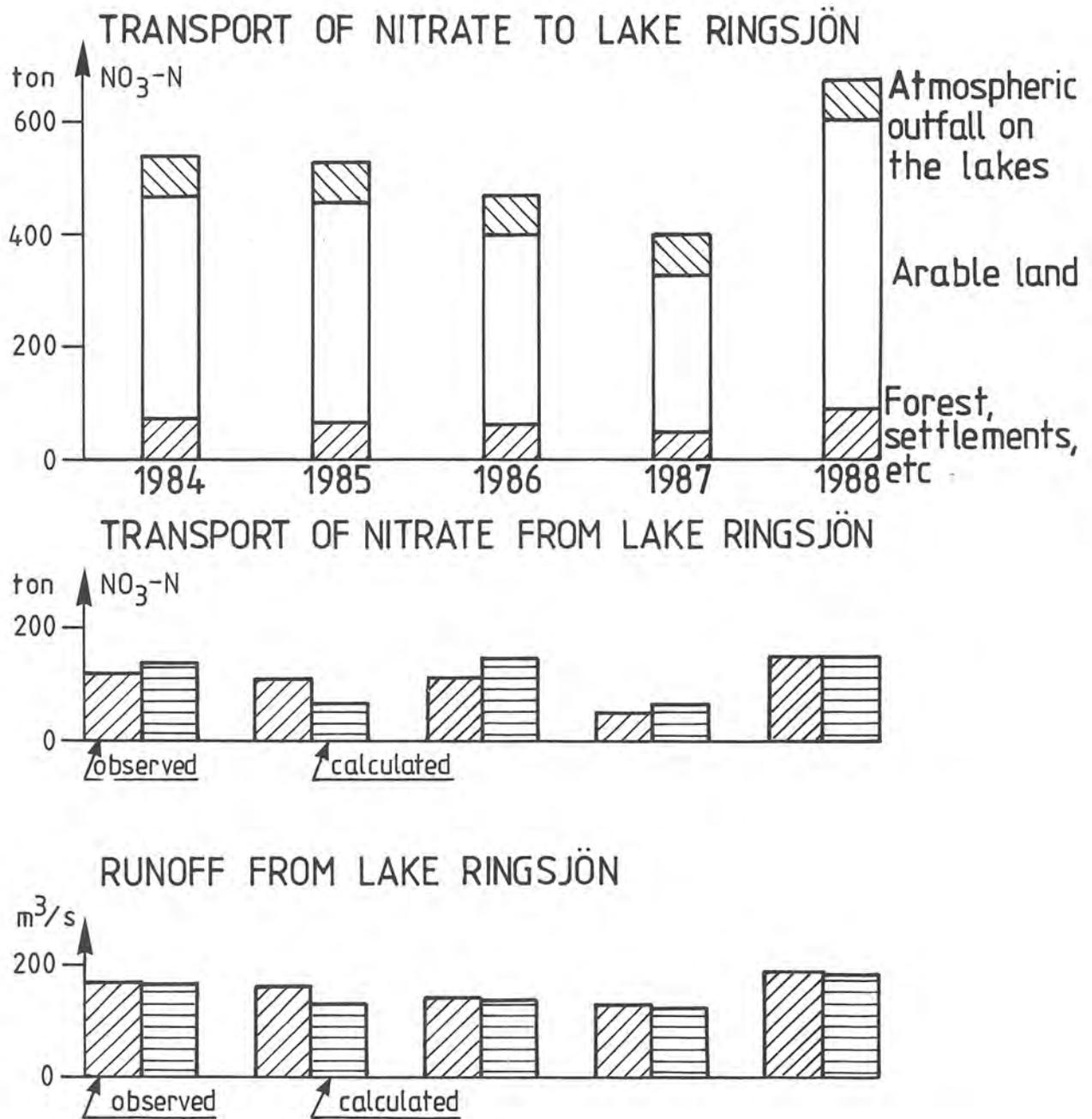
Figure 7. Yearly budgets of nitrate for Lake Ringsjön.

DISCUSSION AND CONCLUSIONS

The study underlines the importance of actual land-use statistics and illustrates the potential of satellite information. Mere analyses of maps may lead to grossly biased results. The use of Landsat-images may, however, turn out to be very expensive if the drainage basin is large.

The calculated nitrate and organic nitrogen concentrations in the different tributaries agreed fairly well with the measured (Figure 4 - 5). It is particularly encouraging as these nitrogen simulations are not subject to model calibration. I have not been able to find any acceptable explanation for the pronounced peak in organic nitrogen in Hörbyån in July 1988 (Figure 4) which also affects total nitrogen. This peak is an isolated phenomenon which could be traces of temporary biological processes of short duration. It has very little effects on the total transport of nitrogen due to the low flows.

Generally the results show that there is normally much greater variability in runoff than in nitrogen concentration. This implies that runoff data or a good water balance model are crucial for estimations of losses of nitrogen. The greatest part of the transport occurs during short periods with high flow. The deviations from observed nitrate and organic nitrogen concentrations during low flow do not disturb the transport results for a whole year.

About 34 % of the drainage basin of Lake Ringsjön is arable land but the contribution of nitrate transport from this arable land is as much as about 85 % of the total nitrate loading to the lake.

The yearly nitrate transport to the lake (inclusive of atmospheric fallout on the lakes) is reduced by 70 to 85 % by biological and physical uptake in the lakes (Figure 7). The retention value for total nitrogen is 45 - 60 %, which gives a nitrogen reduction in the lake of between 65 and 110 kg/ha per year. The lake percentage is 10 % and the turnover time is 0.8 - 1.0 year. This means that the water remains a long time in the lake and gives ample time for the processes in the lake to occur.

In smaller lakes and wetlands around the watershed with a short turnover time the retention is smaller. In one of the sub-basins to Lake Ringsjön, with a lake percentage of 1.5 %, the retention of nitrogen is found to be only 1 %. This lake is, however, less eutrophic than Lake Ringsjön. The role of wetland in reducing nitrogen from the watershed has been debated and is suggested as one of the steps to be taken against eutrophication of lakes and the sea (21). But if a wetland is to be effective in nitrogen reduction it must be wet also in the summer when there is biological uptake and turnover time must be significant. Another problem is that the largest flow and consequently the largest nutrient transport occurs in late autumn - winter - early spring, when there is little biological uptake and the most important sink is sedimentation. This is illustrated by Figure 6. The runoff for the whole years 1986 and 1987 from Lake Ringsjön were about the same, but in 1986 there was a typical winter-spring flow and in 1987 there was a wet summer. The nitrate transport in summer 1987 was reduced much more by biological uptake than corresponding transport in winter-spring 1986 and the retention value was higher for 1987 than for 1986.

The model approach of the lake processes is rough with no attempt to differentiate between uptake, sedimentation, resuspension, and denitrification. Despite this the simplified model makes a good approximation for the lake's outlet. It would demand much more input and verification data to test any more sophisticated approach in this area.

One crucial question with this method is, of course, the representativeness of the data from the typical areas which are the starting point for simulation of concentrations. Some types of land use are also missing and have been replaced by relatively crude approximations.

What are the merits of this type of model work? As we have seen, the runoff fluctuations are crucial for estimations of losses of nitrogen. It is expensive to measure runoff in every tributary that is of interest, and sometimes there can be problem to do this due to low hydraulic head, low water speed or ice damming in winter etc. If the runoff model is calibrated for an area for some years it can be used to compute the runoff retrospectively if data on precipitation and temperature are avail-

able. The calibrated coefficients can also be used in nearby areas or with general coefficients in areas with lakes without any calibration procedure but with greater uncertainties.

The nitrogen transport calculations can be used as a control tool. This means that greater changes due to human impact can be separated from natural changes caused by climate fluctuations.

The model can also be used to estimate the effects of a change in land use on nitrogen transport. For example, we can test what would happen, if ley replaced all arable land. Although this is not a realistic change, it may be of interest to know that it results in a nitrogen reduction of 10 - 20 % from the Lake Ringsjön drainage basin. A more realistic action is to use catch crops after seed in the whole area. What effect would this have? To calculate this we need typical concentrations in small homogeneous fields. Such studies are being done now in Sweden but we need more data from several seasons.

ACKNOWLEDGEMENT

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