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# Comparison of Simulated Nitrogen Balance Under Drained And Un-Drained Catchment Conditions

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**Abstract**: The USDA - Erosion/Productivity Impact Calculator (EPIC) model has been adapted to simulate the influence of tile drainage on nitrogen losses from the soil. The methods used are both analyses of water quality data as well as numerical simulations from selected pilot catchment in Lithuania. The results show that the direct effect of drainage is an increase in the quantities of nitrogen leached from agricultural land. However, it was also concluded that the choice of land management practice was extremely important for the leaching process.

**Key words**: nitrogen, drainage, catchment, runoff

## Introduction

Following World War II, agriculture in Lithuania was rapidly and forcefully collectivized according to the Soviet model. In order to keep high agriculture productivity, large investments were made in the frame of collective and state farms. All the farms were electrified and tile pipes drained large areas. The drained land increased significantly for the period from 1970's to 1990's. In total, 2.62 million ha or 77% of the cultivated areas has artificial tile drainage systems installed. However, such activities also had negative impacts on the environment. Since most of the chemical materials dissolve in water, the hydrological ecosystem regime is closely connected to chemical and nutritive material metabolism. In this way, the possibility has arisen for chemical elements from the soil to reach the primary hydrographic net via drainage systems, and then to reach rivers and other water bodies.

Worldwide, the introduction of drainage systems has conserved or improved thousands of hectares of land for agriculture or other purposes. At the same time the benefits of drainage (i.e. better quality land, higher crop productivity, better trafficable conditions for seedbed preparation and planting in spring, suitable environment for plant growth during growing season etc.) are associated with certain disadvantages. Examples of that can be the environmental problems created by the disposal of drainage effluents polluted with salts, nitrates, herbicides, pesticides or harmful minor elements. One of the direct effects of drainage is that it introduces a discharge through the drainage system. Water in this case acts as a vehicle for all kinds of soluble elements that are stored in the soil. Which effects these elements have on the environment depends, among other things, on climatic conditions, on agricultural practices (quantity and quality of the fertilizers and pesticides used), and on the type of soil. In this respect drainage is undesirable as it leaches fertilizer nutrients, carrying them to receiving streams where they act as pollutants as well as cause eutrophication.

The objective of this paper is to evaluate the effect of tile drainage on nitrogen balance in the soil. In order to achieve this objective, data on drainage flow as well as nitrogen quantities in drainage runoff from small-scale catchment under different land management practices were collected and analyzed. Computer simulations under drained and un-drained scenarios were also performed and analyzed.

## Material and Methods

### Modeling Approach

The effect of drainage on soil mineral nitrogen and leaching of nitrate could be estimated by experiments. However, dynamic processes of mineralization and transport would be difficult and expensive to study based on measurements only. The use of simulation models based on available knowledge of the fundamental processes governing nitrogen transport could help in explaining the behavior of a certain cultivation system during the measurement period. A relevant model that has been properly parameterized for a certain system can also allow quantitative predictions of the future behavior of the system under different scenarios. For example, the effects of different management practices, inputs and climatic conditions may be evaluated (Jansson et al., 1987; Skaggs 1986; Lewan 1994).

Field scale simulation models describe physical, hydrological and biochemical processes in a dynamic way. Consequently, such models describe mathematically water fluxes and associated nutrient fluxes from the land soil profile to the closing outlet section. The complexity of a specific field scale simulation model depends on the temporal and spatial resolution, and on the extent to which important hydrological and biochemical processes are considered (Krysanova et al., 1998). The progress in coupled hydrological/water quality modelling at the field scale or in small homogeneous catchments is evident. Models like CREAMS (Knisel et al., 1980), EPIC, GLEAMS (Leonard and Knisel 1987), SOIL-N (Bergstrom et al., 1991) and OPUS (Smith 1992) are among the most well-known in this respect.

The model chosen for this study is the EPIC - *Erosion/Productivity Impact Calculator,* developed at the Agricultural Research Service (ARS), United States Department of Agriculture (USDA). It contains a set of sub-models which describes the processes associated with water movement and nutrient cycling in a soil. Recently, the model has been adopted and tested for several applications under Lithuania's climate conditions (Povilaitis 1997). It has shown to be a useful tool to evaluate the effect of various management practices and scenarios on nitrogen and phosphorus cycle.

The drainage area considered by EPIC is generally small (1-10 ha) because soils and management are assumed to be spatially homogeneous. In the vertical direction, however, the model is capable of working with any variation in soil properties, the soil profile being divided into a maximum of 10 layers. It consists of numerous component models that pertain to the following major aspects of the erosion/productivity relationship: hydrology, weather, erosion, nutrients, plant growth, soil temperature, tillage and plant environment control. It continuously simulates the processes using a daily, monthly, yearly or long-term time step and readily available inputs. Brief descriptions of the model components are given below. However, the mathematical formulas describing all these processes in detail can be found in the EPIC user's guide (Sharpley and Williams 1990), consequently, they are not repeated here.

The hydrology module is based on the water balance equation, taking into account precipitation, evapotranspiration, percolation, surface runoff and subsurface flow for the soil column subdivided into several layers according to the data available. *Surface runoff* is simulated for daily precipitation by using the SCS Curve Number Equation based on non-linear function of precipitation and a retention coefficient. *Potential evapotranspiration* is estimated using the Priestley-Taylor method, which requires solar radiation and air temperature as input. The *actual evapotranspiration* is estimated following Ritchie concept, separately for soil and plants. Plant *transpiration* is simulated as a linear function of potential transpiration and leaf area index. *Percolation* component uses a storage routing technique to simulate flow through soil layers in the root zone. Flow from a soil layer occurs when the soil water content exceeds field capacity. Water drains from the layer until the storage returns to field capacity and as long as the layer below is not saturated. *Lateral subsurface flow* is calculated simultaneously with percolation. Lateral flow occurs when the storage in any layer exceeds field capacity after percolation. *Drainage* via underground drainage systems is treated as a modification of the natural lateral subsurface flow of the area. Drainage is simulated by indicating which soil layer contains the drainage system and the time required for the drainage system to reduce plant stress. This parameter has to be calibrated. The drainage time replaces the travel time in the lateral subsurface flow calculations for the layer containing the system. The snowmelt component is modeled according to a simple degree-day equation. If snow is present, it is melted on days when the maximum temperature exceeds 0°C. Melted snow is treated like rainfall for estimating runoff volume and percolation. To estimate the N contribution from rainfall, EPIC uses an average rainfall N concentration for all storms. The amount of N in rainfall is estimated as the product of rainfall amount and concentration.

The module representing crop and natural vegetation growth is an important interface between hydrology and nutrients. A simplified EPIC approach is used for simulating all the crops considered (wheat, barley, potatoes, oats and others), using unique parameter values for each crop. The potential increase in biomass is estimated as the product of intercepted energy and a specific plant parameter for converting energy into biomass.

The N mineralization model considers two sources of mineralization: fresh organic N pool associated with crop residue and microbial biomass, and the stable organic N pool, associated with the soil humus. Mineralization from the fresh organic N pool is estimated as a function of crop residue, soil temperature and soil water content. It is assumed that nitrogen associated with the stable N pool flows slowly into the active (readily mineralizable) pool. As one of the microbial processes, denitrification is a function of temperature and water content. The denitrification rate in the model is estimated as an exponential function involving soil temperature, organic carbon content and a coefficient of soil wetness.

A supply and demand approach is used to calculate crop uptake of nitrogen. The daily N demand is estimated as the product of biomass growth and optimal N concentration in the plants. Soil supply of N is assumed to be limited by mass flow of $NO_3$-N to the roots. Actual nitrogen uptake cannot exceed the plant demand when mass flow estimates are too large. Amounts of $NO_3$-N in direct runoff, subsurface flow and percolation are estimated as the products of the volume of water and concentration.

## Study Area

The model was tested with measured data of precipitation, soil parameters, drainage runoff and nitrogen transport from a small experimental catchment (named as M7) which covers an area of 17.3 ha and is located about 15 km north-west from Kaunas city (middle Lithuania). This is a part of a larger catchment (Fig. 1) which consists of eight sub-catchments and covers 408.8 ha in total.

The research catchment M7 was chosen as an area where the field-scale models may be tested and validated. This is a relatively flat area with surface slopes varying from 0.5 to 2.1 per cent. Dominating soil types are sod-gleyic dernopodzolic clay loam and clay. Ceramic drainage pipes of 40 mm diameter installed at a depth of 1.0 m and with a spacing of 24 m are dominating in the catchment. The whole area is used for agriculture.

Data on water flow measurements from the research area was available for the period 1968-1990. The surface runoff and drainage flow were measured using Thompson weirs and continuous recorders of type "Valdaj" were installed to register water discharges. Surface runoff events occurred very seldom. Snow measurements in the catchment were done at the end of snow accumulation periods.

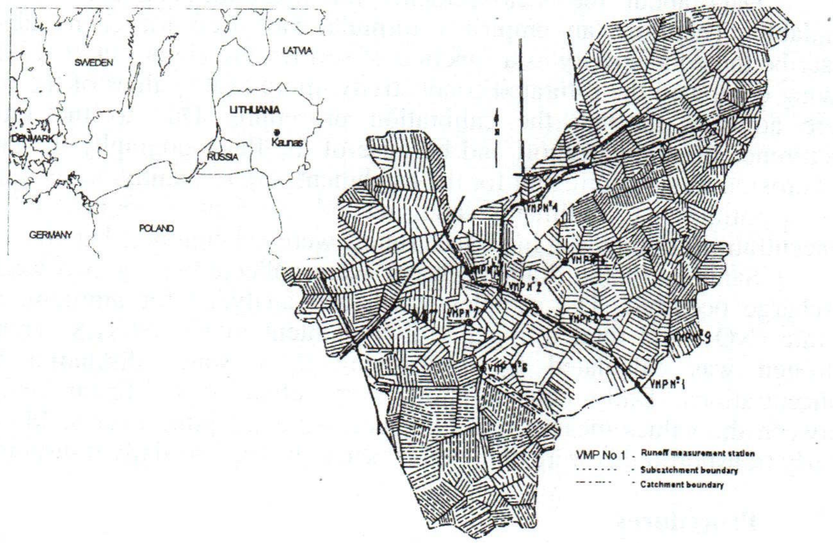


Fig. 1. Study area with surrounding sub-catchments

The Lithuanian Agrochemical Research Laboratory provided site-specific information about the soils in the study area. Samples for evaluating physical as well as agrochemical properties of the soils were taken from two columns. Each column was 1.0 m deep and represents different types of soils inside the catchment. Each column was divided into five sub-layers starting at the ground surface. Thus, the soil database, which represents the parameters for the whole catchment was created according to the data obtained from the two columns (Table 1).

Table 1 - Soil parameters for experimental catchment M7

| Descriptor | Depth to bottom of layer, m | | | | |
|---|---|---|---|---|---|
| | 0.01 | 0.25 | 0.50 | 0.80 | 1.00 |
| Bulk density, t/m$^3$ | 1.660 | 1.660 | 1.500 | 1.530 | 1.500 |
| Wilting point, m/m | 0.060 | 0.060 | 0.048 | 0.045 | 0.050 |
| Field capacity (33kPa), m/m | 0.180 | 0.160 | 0.120 | 0.120 | 0.120 |
| Sand content, % | 36.6 | 36.6 | 47.1 | 47.1 | 50.9 |
| Silt content, % | 45.9 | 45.9 | 28.2 | 28.9 | 25.4 |
| Organic N concentration, g/t | 720 | 718 | 275 | 100 | 60 |
| Soil pH | 5.7 | 5.7 | 5.7 | 5.8 | 6.1 |
| Sum of bases, cmol/kg | 136.9 | 136.9 | 136.9 | 110.0 | 86.5 |
| Organic carbon, % | 0.60 | 0.60 | 0.22 | 0.10 | 0.07 |
| CEC, cmol/kg | 24.0 | 24.0 | 36.0 | 24.0 | 24.0 |
| Coarse fragment content, % vol. | 0.0 | 0.9 | 1.0 | 3.2 | 0.0 |
| Nitrate concentration, g/t | 30.0 | 25.0 | 20.0 | 16.0 | 6.0 |
| Saturated conductivity*, mm/h | 106.0 | 30.0 | 15.0 | 5.0 | 0.5 |

* Values based on calculations (James et al. 1992)

Data about the measurements for hydraulic conductivity were not available. Therefore an empirical formula was used for each sub-layer to describe the conductivity as a function of soil texture (James et al. 1992). After having computed the saturated conductivity, more exact values of the parameter were adjusted through the calibration procedure. Due to the rather rare occurrence of surface runoff and because of the flat topography the process of soil erosion was insignificant for the catchment. Consequently, measurements on the quantities of sediments and related nutrient amounts as well as concentrations of nitrogen in surface runoff were not conducted at all.

Samples of the drainage water were collected every two weeks when discharge occurred. The samples were then analyzed for ammonium ($NH_4$), nitrate ($NO_3$) and total nitrogen (tot-N) content in laboratories. Transport of nitrogen was calculated by multiplying daily water discharge by daily concentrations. Daily concentrations were obtained by linear interpolation between the values measured at two successive sampling events. Monthly and yearly transport totals were obtained by accumulating the daily transport values.

### Procedures

Model *calibration* was performed sequentially for hydrology and nitrogen transport in drainage water for the period 1$^{st}$ February 1978 - 30$^{th}$ November 1980. The hydrological module is fundamental for any model, thus, hydrological parameters were calibrated first. It means that a certain number of parameters related to the processes of water balance were adjusted until reasonable agreement was obtained between simulated outputs and measured values. The main part of input data was determined from the available site-specific information mentioned before.

The soil profile of 1.0 m depth was taken into account. As an input to the upper boundary condition water flux through precipitation and snowmelt was evaluated on a daily time step. The lower boundary condition was simulated assuming a network of drainage pipes at a depth of 1.0 m. The root zone was considered as a soil column measured from the ground surface and up to 1.0 m downwards. All necessary corrections, including assumptions about the root zone depth, were made in the EPIC's Crop Parameter file to adjust unique values for a crop. This pertained to the data on dynamics of the leaf area index, optimal temperature for plant growth, crop height and etc.

The research catchment was cultivated with *sugar beet, barley and winter wheat* crops on rotation during the 3-year period mentioned before. Any information about land management operations was very important to the sensitivity of all the processes involved in the model and their interactions. Each crop required a separate year in the rotation. The month and day of operations were entered in chronological order.

The calibration procedure was carried out on the basis of surface (Q) and drainage flow ($Q_d$) measurements. The simulations started on February 1$^{st}$ 1978 with soil moisture content corresponding to field capacity and with the ground water table depth at 1.0 m. Since the runoff volume is estimated using the Curve Number procedure, the model requires that the Curve Number (CN) parameter acceptable for the whole catchment has to be specified. So far, the CN was adjusted to the optimal values for hydraulic conductivity in each soil layer as well as to the drainage time required for the drainage system to eliminate plant stress caused by poor aeration. These parameters were subject to calibration. The graphs of daily measured vs. simulated surface runoff for the calibration period are presented in Fig.2. It is evident from the figure that surface runoff in the catchment occurs in springtime and in small quantities only. Only the year of 1979 can be characterized as having substantial surface flow.

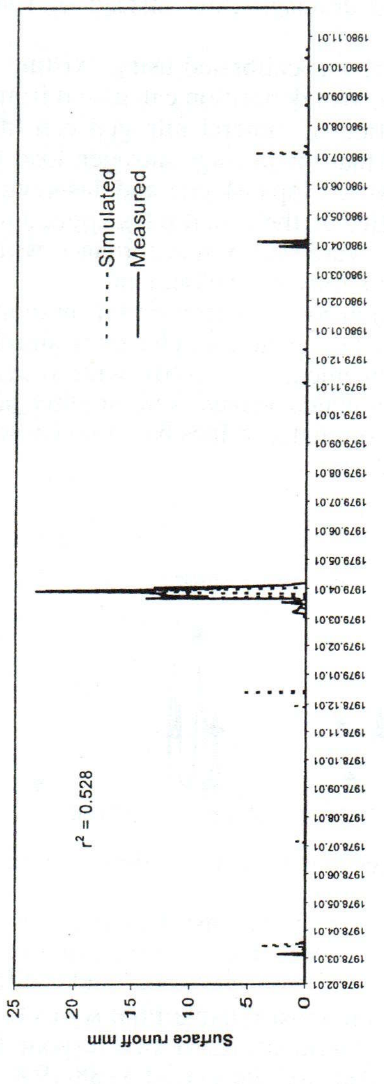


Fig. 2. Simulated vs. measured surface runoff, model calibration

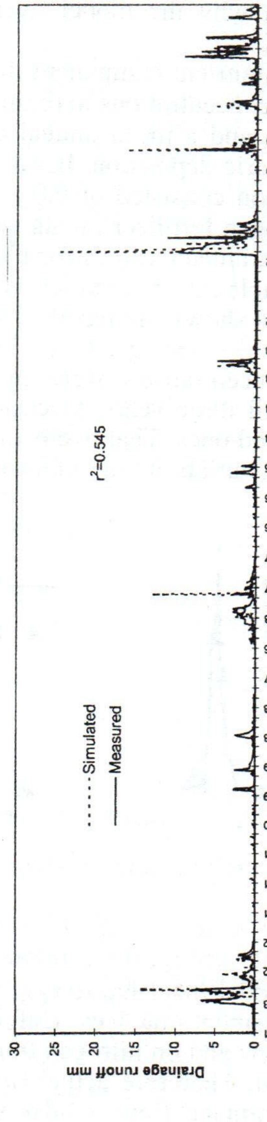


Fig. 3. Simulated vs. measured drainage runoff, model calibration

In general, the model produced reasonable results ($r^2$ = 0.528; N=1034) for daily surface runoff. However, the measured and simulated totals for the calibration period differed significantly ($Q_{sim}$ = 105.4 mm; $Q_{meas}$ = 200.7 mm). This showed that the soil temperature as well as the snowmelt component in the model had to be improved. This shortage was clearly noticed from the comparison of simulated and measured daily drainage flow (Fig.3) dynamics too. The shape of the simulated drainage hydrograph was close enough to observed one ($r^2$ = 0.545; N = 1034), but the peak runoff values differed significantly. The model underestimated peak surface flow, however, it remarkably overestimated peak drainage flow. Soil freeze-thaw cycle was the main reason why the model overestimated drainage flow instead of routing surface flow.

The nutrient component in the model was calibrated using existing data on nitrogen concentrations in drainage water. Wet deposition calculated from net precipitation and a mean annual concentration of mineral nitrogen constituted the atmospheric deposition. It was assumed that the average nitrogen load from wet deposition consisted of 9.0 kg/ha per year (Sopauskiene and Jasikeviciene 1997). Nitrogen fertilizer inputs were specified as the actual rates applied to the field. The parameter values for plant uptake were set up in accordance with the actual available information for a certain crop growth development.

Fig.4 shows simulated and measured nitrogen concentrations in drainage water. Generally, nitrogen levels in runoff and the pattern of change in simulated nitrogen concentrations were in good agreement ($r^2$=0.840) with measured values for all three years. Occasionally, simulated values were slightly higher than measured ones. That overestimation in simulated values has been caused by a substantial variability in drainage flow patterns.

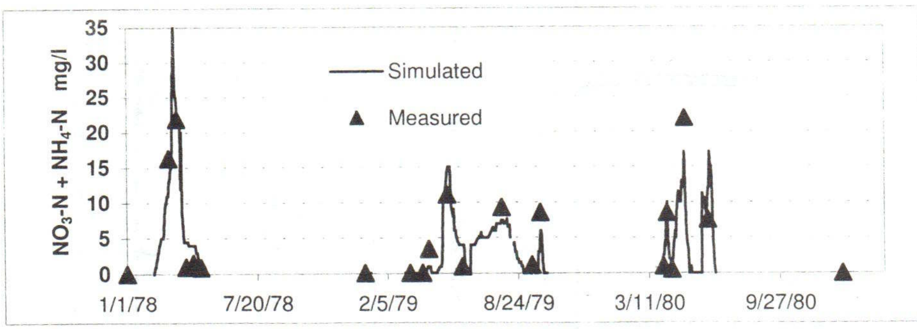


Fig. 4. Nitrogen concentrations in drainage water for the calibration period

A traditional *validation* test was conducted against data from the same catchment as used for the calibration. In this respect, a new simulation was made for the period 1985-1990 to evaluate the goodness of the model under different land management practices. Unfortunately, the most reliable time series data on drainage flow and on nitrogen transport measurements did not correspond to the same period. Therefore actual time series data for the period 1988-1990 were used for drainage flow validation, while the period from 1985 to 1986 was chosen to evaluate nitrogen transport issues. Land management practices, including fertilizer inputs, were specified according to the actual activities, which took place during a certain year. Nitrogen load from wet deposition was taken to be the same as for the calibration procedure. The validation results for daily drainage volumes are shown in Fig. 5.

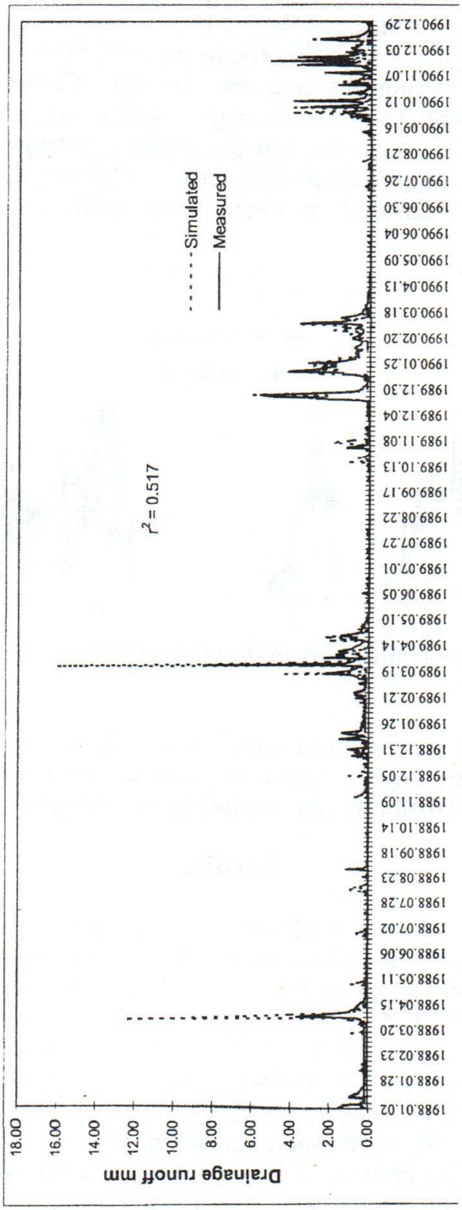


Fig. 5. Daily drainage flow for the validation period

The results indicated that the totals of measured and simulated drainage runoff were essentially equal ($Q_{d\ meas}$ = 339.7 mm; $Q_{d\ sim}$ = 336.2 mm). However, the model tended to overestimate, once again, daily runoff quantities ($r^2$ = 0.517; N=1096). In particular, large differences between measured and simulated daily runoff peaks were noticed during spring floods. This kind of error can be attributed to the errors in calculating snowmelt as it was described before.
The measured leaching of nitrogen during the 2-year validation period was most often less than 15 mg/l per day (Fig.6). Moreover, higher concentrations were always appearing in springtime and late autumn. Compared with measured values, nitrogen simulated in drainage water seemed to be slightly overestimated. Such an overestimation in simulated nitrogen was caused by the model's reaction to the application of mineral fertilizers each year. Nevertheless, the simulated and measured nitrogen concentrations showed reasonable agreement ($r^2$=0.672).

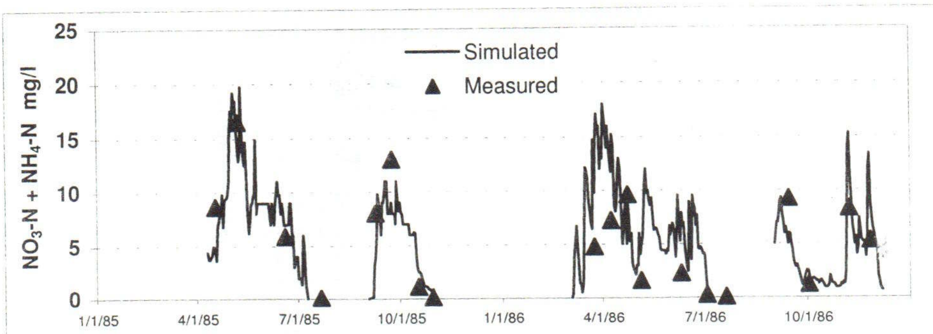


Fig. 6. Nitrogen concentrations in drainage water for the validation period

Therefore, in spite of some inaccuracies in simulating water quantities based on the daily time step the model was considered as valid for the prediction of water quality under different land management scenarios.

## Results

After the model was validated, it was adopted to evaluate the nitrogen balance in the soil under tile drainage conditions. The modular construction of EPIC makes it possible to replace any component with another version. Therefore the subsurface drainage component can be included or excluded. In this respect, it is much easier to evaluate the tendencies in how the system behaves under drained and un-drained conditions. The numerical experiments described below were built up for two options - with drainage and without it. The model set up for the un-drained condition was the same as for the drained one, but the drains were omitted. It is important to emphasize that in this chapter results of *two different simulations* are compared with themselves but not with measured values.

The purpose of these simulations was to establish relationships between the tile drainage and nitrogen losses from the soil. The same procedures are time-consuming and expensive for field studies. Moreover, there is most often no data available. Therefore, in this study, two scenarios according to different land management options (under both drained and un-drained conditions) were compared: 1) nitrogen loss in the soil taken under barley crop and 2) nitrogen loss in the soil taken under perennial grass cover. In addition to that, the effect of changes in land use on nitrogen losses was analyzed. Simulations were performed using the same meteorological data and soil parameters for both the un-drained and drained modeling options. Thus, climatic conditions for the period of 1988-1990 (the same as for validation) have been used. Land management practices (crop, fertilizer application, planting and harvest days or tillage) were the only parameters changed. All the results obtained through the simulations were provided on a monthly time step.

### Nitrogen Losses in the Soil Taken Under Barley

The nitrogen balance approach was used to evaluate the effect of drainage on nitrogen losses in the soil. In this scenario, nitrogen coming from wet deposition, added as mineral fertilizer and mineralized in the soil profile both through crop residue and humus (stable organic matter) was considered as input. At the same time, the processes of denitrification, $NO_3$-N moved out by surface runoff, mineral N transported in drainage water and loss in percolate as well as N uptake by the crop were assumed as losses. Land management activities were sequentially repeated every year through simulations for two options - with and without drainage. The fertilizer rate of 50 kg N/ha applied every year in April was assumed according to the assumption that the barley crop needs to take up such amount to produce the harvest yield of 2000 kg/ha. The indicated harvest yield is relatively small, however, this was based on the mean annual barley yield in the research catchment. The soil's reserves of nitrogen were assumed to be composed of more or less easily mineralized fractions from fresh organic material (plant residues and soil biomass) to very old and stable residual substances (humus) left over after many years of decomposition.

The simulations showed that *mineralization* proceeded more rapidly if there is enough oxygen in the soil and the soil temperature is high enough. So far, N mineralized from organic soil pool as well as from humus material significantly increases (up to 6-8 kg/ha/month) during the months from May to August each year. During winter, mineralization was limited by low temperatures, but as the soil warms in spring the rate increases rapidly, roughly coinciding with crop growth. It was evident that mineralization continued long after uptake by the barley crop had ceased, causing a considerable accumulation of nitrogen during late summer, autumn and early winter. There was only a very slight difference between drained and un-drained conditions for nitrogen mineralization. This difference can be explained by the fact that drainage significantly reduces the soil water content in early spring and late autumn while not having such an important influence on soil water content in summer time when the soil temperature is high and the process of mineralization is most intensive. Moreover, the soil temperature during early spring, late autumn or in early wintertime under drained and un-drained conditions differs insignificantly. In this case low temperature limits mineralization for both options. Consequently, tile drainage did not show a significant effect on organic nitrogen mineralization in the soil.

There were three main processes of mineral nitrogen loss from the soil - leaching, surface runoff and denitrification. Under the given climatic and hydrologic conditions, leaching was mainly a spring phenomenon. Since mineral nitrogen is dissolved in the soil solution, precipitation and the soil's water holding capacity and permeability determine how much nitrogen can be leached. Under drained conditions rain and/or snowmelt water percolates down through the soil and it pulls dissolved nitrogen ($NO_3$-N and $NH_4$-N) into drainage pipes. Mineral nitrogen is not bound to the soil particles but stays in the soil solution where it is subject to leaching or gaseous losses. According to the model, in case when drainage is excluded from the simulations the dissolved nitrogen can be transported by the lateral subsurface flow. Modeled conditions indicated that much more nitrogen was removed from the soil under drained field scenario. It was noticed from the results that large amounts of nitrogen in the percolate water exist only for short periods of time after the incorporation of fertilizers in April. The vegetation period for barley is relatively short since it is usually sown in April and harvested in August. This creates preconditions for nitrogen leaching. The background to nitrogen leaching is, that the soil delivers (through mineralization) nitrate nitrogen also during the part of the year when none is required by the crop. It seems like nitrogen leaching depends largely on how much nitrogen is in the soil during the parts of the year when crops are not absorbing nitrogen. Naturally, significant amounts of nitrogen in drainage or percolate water were produced either in early spring (March) or just after fertilizer application in April. This process was greatly influenced by hydrological conditions in the catchment. It is important to notice that nitrogen in percolate water under un-drained conditions was always produced in some period later as it was for nitrogen in drainage flow.

Nitrate nitrogen ($NO_3$-N) in surface runoff was found as insignificant for both scenarios. This part of nitrogen loss depends closely on nitrogen content in the upper soil layer, soil texture and on the amount of nitrogen in precipitation. The amount of nitrate nitrogen transported by surface runoff under drained and un-drained conditions was slightly different (up to 0.02-0.05 kg/ha/month). Larger quantities of $NO_3$-N were simulated under un-drained field conditions. These differences in the quantities of N were caused by the differences in hydrological conditions on the ground surface for drained and un-drained modeling options. A subsurface drainage system reduces surface runoff and subsequently decreases transport of nitrogen. This was the main reason why larger quantities of $NO_3$-N were simulated under un-drained field conditions.

The denitrification process for both scenarios was relatively intensive in early spring until the ground began to dry out and the oxygen levels increased. More intensively (20-35%) denitrification proceeded for un-drained conditions.

The process was faster at higher temperatures, thus, the maximum values (1.4-1.6 kg N/ha/month) occurred during late spring in May.

After the simulations were completed, attempts were made to calculate the monthly differences between nitrogen input and loss under drained conditions. The same procedure in calculating nitrogen balance was conducted under un-drained modeling scenario too. So far, when the results from two options were known, a brief comparative analysis for nitrogen balance was performed.

The quantities of mineral nitrogen available *in the soil* were analyzed by summarizing monthly differences between nitrogen income and loss (N balance) under drained and un-drained conditions, respectively. It was done by an integration procedure applied to the nitrogen balance curves. In this way two curves which indicate the cumulative nitrogen in the soil for the 3-year period were obtained (Fig.7). The graphs clearly indicate that the quantities of nitrogen available in the soil are higher (25 kg N/ha) under un-drained conditions. The increase in cumulative N was affected by the smaller quantities of nitrogen loss. It proves that larger quantities of nitrogen can be removed from the soil taken under summer cereals if drainage systems are installed.

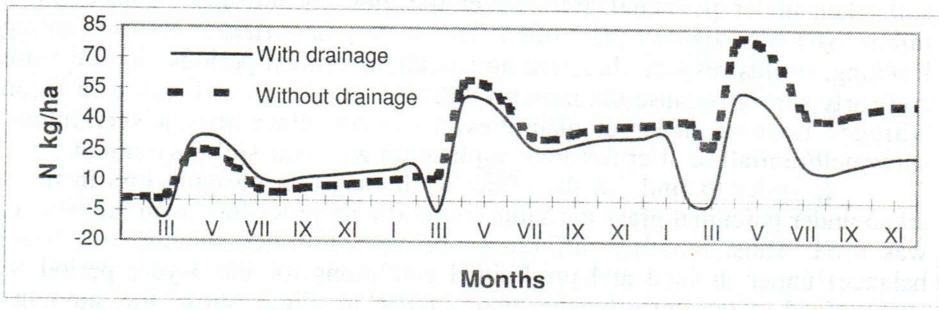


Fig. 7. Cumulative nitrogen balance for barley scenario, simulation

### Nitrogen Loss in the Soil Taken Under Perennial Grass

The same approach as for barley scenario was taken into consideration to analyze nitrogen balance in the soil taken under perennial grass. Perennial grass in this case means the mixture of pasture plants such as *white clover* and *bulbous meadow*. All the steps discussed in the previous scenario were sequentially repeated for perennial grass scenario through the 3-year simulations under drained and un-drained conditions. Only land management practices and plant growth parameters had to be adjusted to obtain ordinary conditions for the land permanently covered with grass and herbaceous plants. A fertilizer rate of 100 kg N/ha was applied every year in April. Such amount of mineral nitrogen was taken into account according to the needs of perennial grass to produce a hay yield of 5500 kg/ha. The indicated yield is based on the mean annual hay yield harvested in the middle part of Lithuania.

The microbial decay of organic material in the soil releases and binds, or in other words mineralises and immobilises, nitrogen simultaneously. If the mineralization is larger than the immobilization there is a *net* mineralization. The reverse situation is referred to a *net* immobilization. It made clear from the results that net mineralization occured throughout the year. Thus, mineralized nitrogen contributed to the crop's nitrogen supply in larger or smaller quantities over the whole year. In this case a net mineralization amount from 40 to 50 kg of nitrogen per one hectare each year can be expected. However, the simulated results for mineralized nitrogen do not show any significant difference between drained and un-drained conditions.

In general, the results from simulations showed that if the soil were taken under perennial grass, the risk of leaching is reduced significantly. This is because the grass cover can absorb nitrogen late into the autumn and even winter, if the conditions permit, and then continue uptake in the early spring. So far, large amounts of nitrogen are taken up by the crop instead of staying in the soil solution where it is subject to leaching. In this respect, perennial grass showed as much more effective cover than summer cereals like barley. In the soil taken under perennial grass cover the monthly nitrogen balance (income minus loss) was negative most often. This is very important in terms of nitrogen leaching. In this respect, there are no so-called "critical periods" in late autumn and early spring because the monthly nitrogen uptake exceeds nitrogen income. Nitrogen leaching in larger quantities can occur either after a very intensive snowmelt/rainfall or after fertilizer application when the soil is saturated.

In order to find out the effect of drainage on nitrogen loss in the soil taken under perennial grass the same summarizing procedure as in previous case was used. Thus, monthly differences between nitrogen income and loss (N balance) under drained and un-drained conditions for the 3-year period were summarized. Consequently the two curves in Fig.8 show the amount of cumulative nitrogen in the soil for drained and un-drained modeling options. From the obtained results it is clear that drainage has no significant effect on nitrogen loss if the soil is covered by perennial grass. Only a very small difference (5.0 kg N/ha) between the two curves was found at the end of simulations.

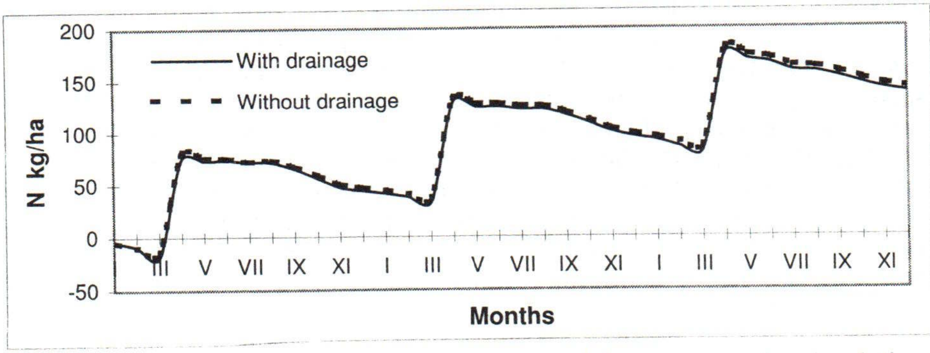


Fig. 8. Cumulative nitrogen balance for perennial grass scenario, simulation

Therefore the comparative analysis indicates that the quantities of nitrogen leached out from the soil under drained conditions strongly depend on the land management practices (crop, fertilizer application, planting and harvest days or tillage). Nitrogen loss under the same climatic conditions but for different crop scenarios (e.g. barley vs. perennial grass) differs significantly if a drainage system is present.

### Effect of Changes in Land Use on Nitrogen Losses

The numerical experiment approach was also applied to make a scenario for evaluating the effect of changes in land use on nitrogen losses under drained conditions. This scenario consisted of 10-year simulations imitating the rotation of two crops - perennial grass followed by barley. Meteorological conditions used in these simulations included the period from 1981 to 1990. The soil was assumed to be taken under perennial grass cover during the first seven years. So far, perennial grass was allowed to grow as a multiple year pasture without any ploughing for seven years. Afterwards, perennial grass was ploughed under in the spring of the next year and barley took place for the rest period of three years. In this scenario, harvest was simulated to occur in June and August for perennial grass, and in August for barley, with fertilizer rate of 100 and 50 kg N/ha each year, respectively. This scenario was meant to simulate what might happen if the long-term grass cover was ploughed under and subsequently changed to summer cereals.

Fig. 9 provides the dynamics of nitrogen loss under drained conditions for the simulated period. This includes nitrogen losses through denitrification, drainage and surface flow and in percolate. It is evident that the loss of nitrogen is small enough under perennial grass cover. However, when the pasture was ploughed under, large quantities of nitrogen were suddenly released.

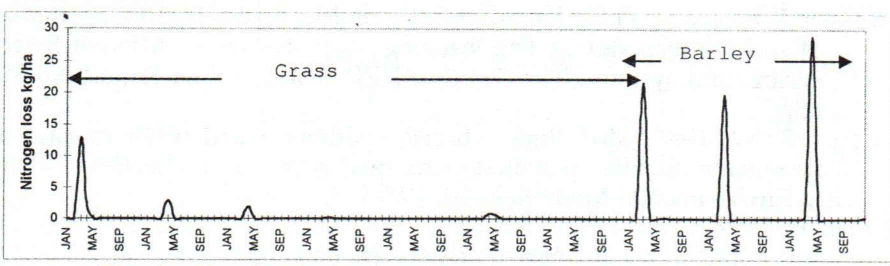


Fig. 9. Simulated nitrogen loss after the change in land use

The results from this scenario indicate that if the soil is subject to leaching, and the pasture is to be followed by a spring crop, it should be expected that large quantities of nitrogen in drainage water would occur.

## Conclusions

In this study, actual data on nitrogen quantities in drainage runoff were analyzed as well as computer simulations were performed in order to find out the effect of tile drainage on nitrogen loss from small-scale agricultural areas. Much attention was paid to compare simulated nitrogen loss between computer scenarios under drained and un-drained field conditions. After this, the following conclusions can be drawn:

1. The installation of tile drainage system changes nitrogen cycle in the soil. It slightly increases organic nitrogen mineralization while decreasing denitrification. However, the direct effect of drainage is the larger quantities of nitrogen leached out from the soil.

2. The quantities of nitrogen leached out from the soil under drained conditions largely depend on the land management. Therefore nitrogen loss under the same climatic conditions but for different crop practices (e.g. barley vs. perennial grass) differs significantly.

3. Leaching of nitrogen through drainage flow depends on how much nitrogen is in the soil during the parts of the year when the soil is un-cropped. The background to nitrogen leaching is, that the soil delivers nitrate nitrogen also during the part of the year when none is required by the crop.

4. In order to reduce nitrogen leaching via drainage flow agricultural fields have to be kept under plant cover during so-called "critical periods" in spring and autumn.

5. The changes in land use greatly affect nitrogen loss via drainage flow (e.g. if a multiple year pasture is ploughed under, large quantities of nitrogen are suddenly released).

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# KOMPARACIJA SIMULIRANOG BALANSA NITROGENA U SLIVU POD USLOVIMA DRENAŽE I NEDRENAŽE

- originalni naučni rad-

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## Rezime

USDA-model za proračun uticaja erozije/produktivnosti (EPIC) prilagođen je da simulira uticaj drenažnih cevi od cigle na gubitak nitrogena iz zemljišta. Korišćene metode su i analize kvaliteta vode kao i numerička simulacija odabranih pilot slivova u Litvaniji. Rezultati pokazuju da je direktan efekat drenaže povećanje količina nitrogena koji se isprao iz oraničnog sloja. Međutim, zaključeno je takođe, da izbor prakse u upravljanju zemljištem ima izuzetan značaj za proces ispiranja.