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Abstract

The leaching of nitrate from agricultural areas and the effect of additional nutrients on both freshwater and marine ecosystems have become crucial environmental problems in large parts of Europe. Along with the increasing concern of nitrate pollution the need for predictive tools, such as simulation models, has become increasingly important. Modelling of nitrogen leaching from agricultural soils has a long tradition for north and central Europe, whereas analyses from the Mediterranean are fewer.

This study presents a methodology and a case study for the large-scale simulation of nitrogen leaching. A modelling system consisting of the distributed hydrological model MIKESHE and the one-dimensional nitrogen model DAISY was applied to a coastal agricultural catchment, draining into an eutrophic area of the Adriatic Sea. The investigated catchment was situated in the Po delta in Northern Italy. The hydrological model MIKESHE was calibrated and validated at the catchment scale using timeseries of groundwater table and channel discharges. A nested modelling approach was adopted for the DAISY model. First, the model capabilities of describing the root zone processes were carefully evaluated through a field scale calibration/validation using soil water content, mineral nitrogen concentration, dry matter yield and crop nitrogen uptake. Afterwards the impact of various catchment characteristics on nitrogen leaching was carefully evaluated, providing the necessary basic for the subsequent model application at the catchment scale.

The modelling system gave a satisfactory description of the overall hydrological processes and the nitrogen transformation in the root zone. Thus, the model provided a quantification of nitrogen leaching associated with current agricultural practices and was able to describe the overall hydrological processes in the study area. The impact of alternative management practices on nitrogen leaching and harvest yield was also estimated through scenario analysis. The modelling result has provided considerable knowledge of those nitrogen root zone processes characterising an agricultural soil with shallow groundwater table. Finally this work has proposed an upscaling procedure for the one-dimensional model DAISY to represent condition at the large catchment scale, providing the basic for a further upscaling to the regional scale.

Preface

This report present the results of my Ph.D. study at the Institute of Geography and International Development Studies, Roskilde University. The project was funded by the European Commission and the work was conducted at the Joint Research Centre, Environmental Institute, in Ispra, Italy, in co-operation between the Roskilde University. The obtained scientific results are then part of the "Environmental Science - Ph.D. program" at the Roskilde University.

I would like to thank my supervisors Giovanni Bidoglio, Niels H. Jensen, and Fransisco J. Elorza for their support, encouragement and expert guidance during the work.

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I am also grateful to my mother, husband and three children for their support, and especially for the great patience, which they have given me during the last part of my thesis.

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1. Introduction

1.1. Background

The nitrogen enrichment of natural waters can have several undesirable effects. High levels of nitrogen are significant contributors to eutrophication both of inland and coastal waters. Moreover high concentrations of nitrate in drinking water is suggested to have a negative impact on public health as they increase the risk for different types of diseases such as methaemoglobinaemia in babies (EC, 1997). The leaching of nitrate from agricultural areas and the effect of additional nutrients on both freshwater and marine ecosystems have created crucial environmental problems in large parts of Europe. Large portions of the coastline both of the North Sea, the Baltic Sea and the Adriatic Sea are suffering from eutrophication due to high levels of nutrient discharge (Stålnacke, 1996; Viaroli, 1993, EEA, 1995). An assessment of the state of environment in Europe indicated that 22% of agricultural areas in Europe has nitrate groundwater concentrations above the maximum concentration limit of 50 mg/l, and that 87% of the areas has nitrate groundwater concentrations above the guide level of 25 mg/l. (EEA 1995).

Various action programmes have been established in order to reduce the nitrate pollution deriving from agricultural non-point sources. In the late 1980s, target of 50 % reduction of nutrient export to the North Sea and the Baltic Sea was adopted by HELCOM (Helsinki Commission) and OSPARCOM (Oslo-Paris Commission) conventions. Similar goals were also adopted with the implementation of the Danish "Vandmiljøplan" in 1987 (Miljøstyrelsen, 1987) and 1998 (Miljøstyrelsen, 1998). Moreover, the European Community adopted in 1991 the Nitrate Directive (CEC, 1991), which imposed upon the EU-Member States to implement various action programmes for reducing water pollution caused, or induced, by nitrates from agricultural sources.

Table 1. Trends in nitrate fertiliser usage related to agricultural area in kg N/ha agricultural land (Source FAO; in EEA 1999).

	1985	1990	1994	Trend 1990-1994		1985	1990	1994	Trend 1990-1994
Austria	47	39	35.4	-9.3%	Portugal	34	37	34.4	-7.0%
Belgium and Luxembourg	128	125	113.4	-9.3%	United Kingdom	86	85	89.1	4.8%
Denmark	135	142	116.4	-18%	Sweden	70	62	61.1	-1.4%
Finland	79	81	73.4	-9.4%	Spain	31	35	36.6	4.6%
France	77	81	76.8	-5.2%	Albania	67	66	13.3	-79.8%
Germany			103.0		Bulgaria	77	73	27.0	-63.0%
Greece	49	47	38.2	-18.8%	Estonia		50	17.9	-64.1%
Iceland	6	5	5.7	14.0%	Hungary	85	55	39.5	-28.1%
Ireland	55	66	97.7	48.0%	Latvia		46	15.7	65.8%
Italy	62	52	56.1	7.9%	Lithuania		61	1.7	-97.2%
Netherlands	248	194	193.6	-0.2%	Poland	71	39	44.7	14.7%
Norway	112	113	106.8	-5.5%	Romania	47	44	15.4	-65.0%

Since the late 1980s there has been a marked decrease in the consumption of nitrogen fertiliser in many of the EU countries (Table 1). According to results of EEA (2000) there is however "no evidence that reduced application of nitrogen fertilisers to agricultural land has resulted in lower nitrate concentration in rivers"; and the nitrate concentration in the major European rivers have been about constant since 1980 (Figure 1A). A similar tendency is also seen in the groundwater, where data reported for France, Great Britain and Denmark, indicates that nitrate concentrations in groundwater still appear to be rising (Figure 1B). Moreover, a quarter of the sampling sites used in EEA databases - covering 17 European countries – had the nitrate concentration above the maximum limit of 50 mg/l (EEA, 2000).

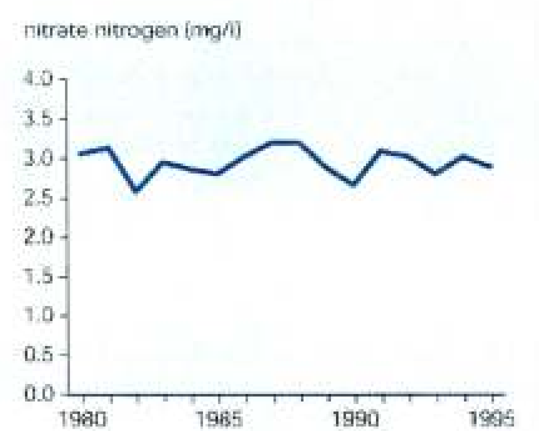

Figure 1A. Development of nitrate concentration in major European rivers (Source: CTC/IW, in: EEA 2000).

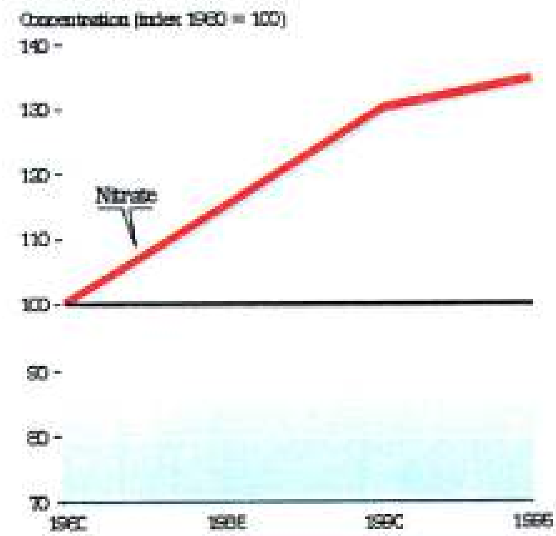

Figure 1B Development of Nitrate concentration in groundwater (France, Great Britain, Denmark); source (Source: CTC/IW, in: EEA 1997).

In the Baltic States and Poland the fertilisers level has decreased at an unprecedented rate since the late 1980s (Stålnacke, 1996). The impact of this dramatic fertiliser reduction on the nitrate levels in major Latvian and Polish rivers was analysed by Stålnacke (1996) and Tonderski (1996), respectively. In the investigated rivers there was a remarkable lack of response to the dramatic decrease in the use of fertilisers. Thus, neither Stålnacke (1996) nor Trondenski (1996) could identify a downward trend in the nitrate levels in the rivers. A similar tendency was also observed by analysing time series of nitrogen transport in 39 Swedish rivers (Stålnacke, 1996). During the monitoring period running from 1971 through 1994, the agricultural practices in Sweden were improved considerably. However, according to those results, such an improvement did not reduce the export of nitrogen from agricultural catchments.

The problem of nitrate pollution has re-emerged as an important environmental priority in Europe (EC, 1996; EC, 1997). Along with the increasing concern of nitrate pollution the need for predictive tools, such as simulation models, has become increasingly important.

During the last decade a significant progress has been made in the development of comprehensive physically based leaching models (Thorsen, 1996). These models appear to have some predictive capability, and may, with professional and caution use, be very useful for management purposes (Thorsen, 1996). Many of these models like WAWE (Vanclooster, 1994), RZWQM (DeCoursey, 1992) and DAISY (Hansen, 1991) are one dimensional. These models have been applied mostly at plot/field scale and most often to controlled experiments with extrodinary good data availablity. Nitrogen spreading in soils and waters is, however, the outcome of processes occurring in the various parts of the hydrological cycle. Recent research has also revealed a remarkable lack of response between a reduced fertiliser usage on land and improved water quality in the receiving water bodies. These results indicate an increasing need for a better understanding of the nitrogen dynamics at a scale covering the entire drainage basin. What is especially needed is an integrated analysis describing the nitrogen processes from its application at the field to its appearance in streams. Modelling analysis of this kind is also seldom reported in the literature. An example is the application of the coupled MIKESHE/DAISY modelling system in agricultural catchment in Denmark (Styezen and Storm 1993a, 1993b) and Slovakian (Refsgaard, 1998); as well as the application of the NNT-Watershed model in an agricultural catchment in Connecticut (Heng and Nikolaidis, 1998).

Modelling of nitrogen leaching from agricultural soils has a long tradition for north and central Europe, whereas studies from the Mediterranean are fewer. Examples of such modelling analysis include the work of Ceotto (1994), evaluating the model performance of the model CROPSYST (Stockle, 1992) in terms soil water and nitrogen content. The capability of the model CROPSYST and EPIC (Williams, 1984) for predicting leaf area index, crop growth as well as soil water and nitrate concentration were also compared both by Spallacci (1994) and Rinaldi and Ventrella (1997).

1.2. Objective

As a part of a project investigating interactions between upland river drainage basins and near coastal waters, the Joint Research Centre started a modelling activity of a coastal agricultural catchment called Po di Volano, draining into an eutrophic area of the Adriatic Sea. Specifically, such an activity intends to develop an integrated large–scale modelling system describing both hydrology and nitrogen transformation processes occurring along the aquatic continuum in the catchment.

The study reported here focuses on the modelling of the hydrology and the root zone nitrogen processes in a subcatchment of the investigated drainage area. The nitrogen dynamic in the root zone has been analysed with focus on the various factors affecting the nitrogen leaching processes. The outcome of the project contributes to the research on the large-scale modelling of nitrogen leaching with the specific objectives of:

- Evaluating the model performance of the MIKESHE/DAISY modelling system, with special emphesis on the applicability of the DAISY model for Southern European condition
- Estimating the nitrogen loading deriving from the agricultural area,
- Analysing which environmental improvement can be achieved by implementing alternative management practices according to the council Regulation EC 1257/1999.

2. Material and methods

2.1. Modelling system

The applied modelling system consists of two different model:

- *the one-dimensional nitrogen model DAISY,* describing the nitrogen dynamics in the root zone and providing estimates of the nitrogen leaching deriving from the agricultural land; and

- *The distributed hydrological model MIKESHE,* quantifying the hydrological processes within the study area and providing the necessary groundwater maps for modelling nitrogen leaching.

2.1.1. DAISY

DAISY (Hansen et. al., 1991) is a deterministic one-dimensional model simulating crop production as well as water and nitrogen dynamics in agro-ecosystems subject to various management strategies. The code was developed with the framework of the Danish NPO research Programme (Dyhr-Nielsen et. al., 1991), and as part of the EC Environmental Research Programme (Thomasson et. al., 1991). Subsequently, DAISY has been applied in a number of research projects at both the field scale (etc. Jensen et. al., 1994; Svendsen et. al., 1995; Jensen et.al., 1997; Heidmann, 1999; Djurshuus, 1992 and Djurshuus et. al., 1999,), as well as catchment and regional scale (Jensen et. al., 1994a; Refsgaard et. al., 1998, Styczen and Storm, 1993a and 1993b). Moreover, it has performed well in several validation exercises (Vereecken et. al., 1991; de Willigen, 1991; Diekkruger et. al., 1995).

DAISY has so far only been applied in Northern and Central Europe, and its performances in conditions typical of Southern European condition (e.g. warmer climate conditions, introduction of other cultivar types) have not yet been evaluated.

The soil part has a one-dimensional vertical structure with the soil profile divided into layers according to physical and chemical soil properties. The driving variables for DAISY are daily values of temperature, precipitation, and global radiation. In areas with shallow ground water table, information concerning depth to groundwater table is also needed. Moreover, a description of the chemical and hydrological soil properties (soil water characteristics, C-org and C/N ratios) and management practices (crop sequence, soil tillage, irrigation and fertilisation rates) are also required. A detailed description of DAISY can be found in Hansen et. al.(1991) whereas the following provides a brief summary of the main components:

The hydrological processes considered in the model include snow accumulation and melting, interception of precipitation by crop canopy, evaporation from crop and soil surface, infiltration, water uptake by plant roots, transpiration. The vertical movement of water in the soil profile is modelled by means of a numerical solution of the 1-dimentional Richards equation. The soil temperature is modelled by heat flow equation taking into account heat transfer by conduction and convection, and changes in heat content by freezing and melting processes.

The organic matter turnover is modelled by dividing the organic matter conceptually into three main pools i.e. added organic matter (AOM), soil microbial biomass (SMB), and dead native organic matter (SOM). According to their stability against decomposition, the three pools are further subdivided into two sub-pools, characterised by C/N ratio as well as turnover rate. For each sub-pool of SOM and AOM, the carbon turnover is modelled by applying a first order kinetics, assuming that the rate coefficient is influenced by soil temperature and soil water content. In case of the sub-pools of SOM, the rate coefficients are also influenced by the clay content of the soil.

The mineral nitrogen processes include nitrification, denitrification, plant uptake and vertical movement of inorganic nitrogen in the soil:

- The nitrification is described by a first order kinetics assuming the rate coefficient to be influenced by soil temperature and soil water content.
- The denitrification is simulated by defining a potential denitrification rate, related to the carbon dioxide evolution rate in the soil and to soil temperature. The potential denitrification rate is reduced according to the oxygen status of the soil expressed as a function of soil water content. The actual denitrification is either determined as the reduced potential denitrification rate or as the rate at which nitrate in soil is available for denitrification.
- The nitrogen uptake is simulated by defining a potential nitrogen demand simulated by the crop model, and the availability of nitrogen for plant uptake. Plant roots are taking up ammonium in preference to nitrate.
- The vertical movement of nitrogen is modelled by means of a numerical solution of the convection/dispersion equation for ammonium and nitrate. The mobility of ammonium in soil is considered less than that of nitrate due to ammonium adsorption, which is described by an adsorption/desorption isotherm.

The crop model module is based on the concept that the physiological age of a crop can be described as the thermal age in terms of a temperature sum. The simulation of the crop dry matter production is based on daily estimates of canopy gross photosynthesis, partitioning of assimilates, and respiration:

- The canopy gross photosynthesis is determined by the amount of photosynthetically active radiation adsorbed by the canopy and the efficiency by which the adsorbed radiation is converted into carbohydrates. The gross photosynthesis can be limited by water or nitrogen deficit only.
- The partitioning of assimilates from gross photosynthesis between crop parts is simulated as a function of crop thermal age.
- The respiration comprises both the growth respiration and a temperature maintenance respiration.

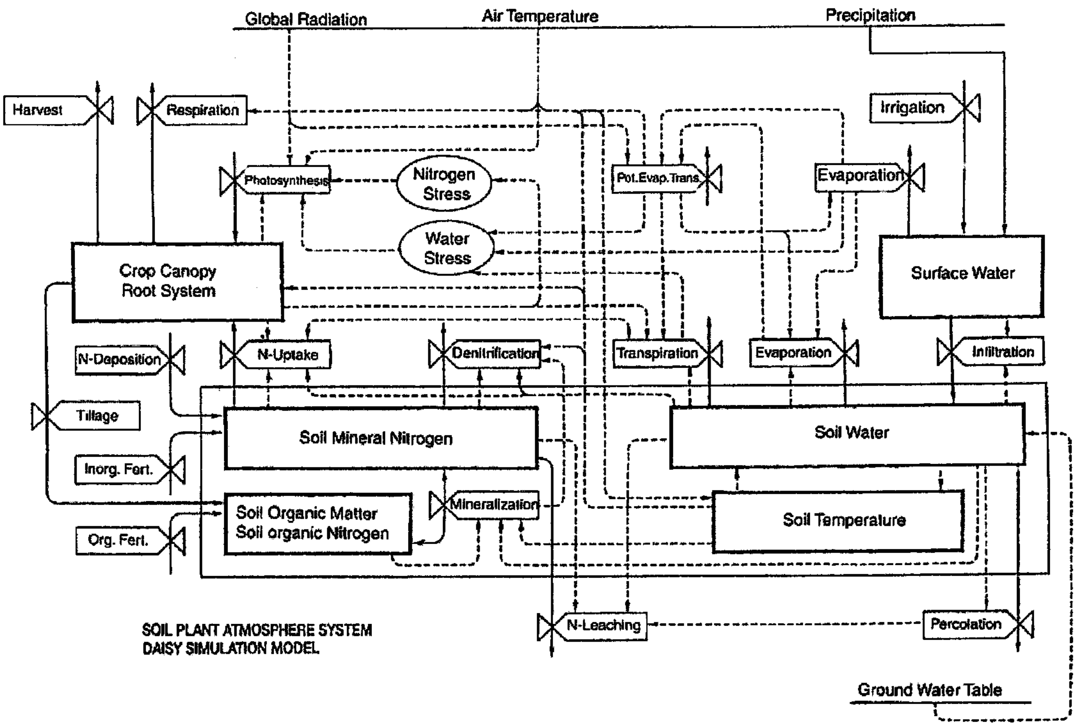

Figure 2. Relational diagram of the model code DAISY. Global radiation, air temperature and precipitation are driving variables. Rectangles represent systems variable; valve symbols represent processes while oval represent auxiliary variables. Solid lines represent flows of matter while broken lines represent flows of information. (Reproduced from Hansen et. al.1993).

2.1.2. MIKESHE

The MIKESHE code is a further development of the SHE Système Hydrologique Européen (Abbott et. al., 1986). It has been applied in a large number of engineering and research project in various countries, spanning from a detailed water flow description of a 9 ha irrigated site in Australia (Jayatilaka et. al., 1998) to the analysis of groundwater depletion in the 1145 Km2 Lake Karla watershed in Greece Kaiser et. al. (1997). Moreover, the combined modelling system MIKESHE/DAISY has already been used in previous research studies assessing large scale nitrogen leaching from agricultural land (Styczen and Storm 1993a, 1993b, Refsgaard et. al. , 1998, Refsgaard et. al., 1999).

MIKESHE is a fully distributed physically based modelling system, which can simulate all major processes occurring in the hydrological cycle. Moreover, it provides a powerful pre- and post-processing facilities with Geographic Information System (GIS) capabilities. All processes are fully coupled allowing for feedbacks and interactions between components. The spatial variation in catchment characteristics (e.g. soil type, topography, land use) and driving variables (e.g. meteorological data) are represented by a network of uniform grids. Each grid is further divided into a number of layers in the vertical direction in order to describe variation in soil and hydrogeological characteristics. The vertical flow in the unsaturated zone is described independently for each grid, and these vertical soil columns link the three-dimensional groundwater flow to the two-dimensional flow system for overland flow. A detailed description of MIKESHE is reported in Refsgaard and Storm (1995), whereas Singh et. al.(1997) and Storm and Refsgaard (1997) give a short summary of the main components:

Interception/evapotranspiration: The interception process is modelled by introducing interception storage, expressed as function of leaf area index. The calculation of the actual evapotranspiration is based upon the potential evapotranspiration using the Kristensen – Jensen model (Kristensen and Jensen, 1975). Here the actual evapotranspiration rate is further adjusted according to the vegetation density and the water content in the root zone.

Overland and channel flow: The overland flow is simulated in each grid by solving the two-dimensional diffusive wave approximation of the Saint-Venant equations. For the channel drainage network channel flow is calculated on the basis of the one-dimensional form of the Saint-Venant equation which is solved in a separate node system located along the boundaries of the two-dimensional grid squares.

Unsaturated zone: As for DAISY the vertical flow in the unsaturated zone is described by the one dimensional Richards equation. The extraction of moisture for transpiration and soil evaporation is accounted for via a sink term at the node points in the root zone. Infiltration rates are found by the upper boundary, which can either be flux controlled or head controlled. The lowest node point in the finite difference scheme depends on the phreatic surface level, and allowance is made for the unsaturated zone to disappear in cases where the phreatic surface rises to the ground surface.

Saturated zone: The groundwater flow is calculated using an implicit finite difference solution of the two- and three-dimensional Boussinesq equation. The interaction between the ground water and river system is calculated on the basis of the water levels in the river and the ground water tables. The discharge/recharge to the river occurs from all computational cell located along the river links.

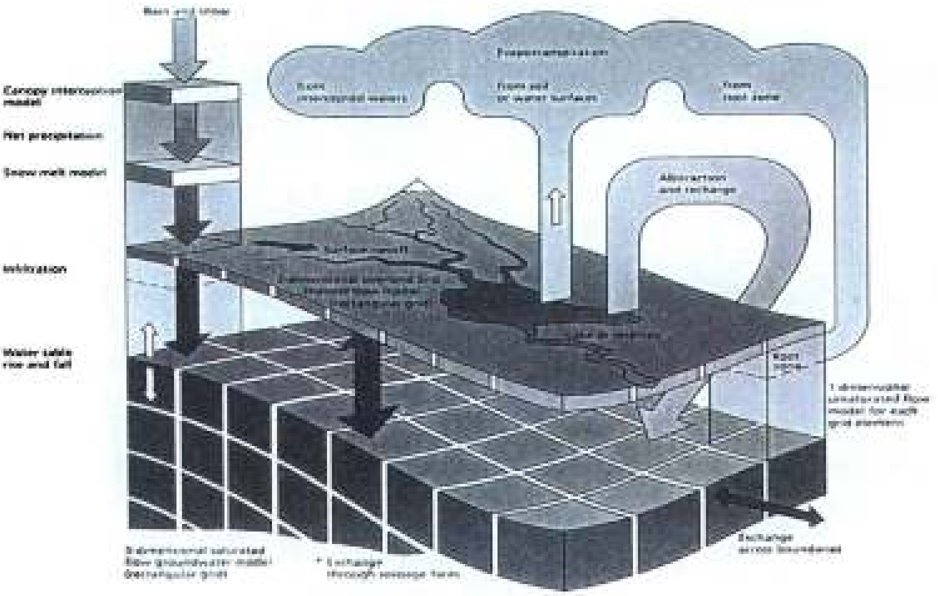

Figure 3. The structure of the MIKESHE modelling system (DHI, 1993).

Physically distributed hydrological models like MIKSEHE is characterised by an extensive data requirement and a large amount of parameters, which can be modified during the calibration process. This extensive model parameterisation has brought about much criticism of the distributed models.

According to Beven and Fisher (1996) the main problem is "that the many parameter sets, scattered throughout the parameter space, provide adequate simulation for whatever quantitative measure of performance is being use. Beven (1996) moreover claims that the problem of overparameterisation is greater for the distributed models, due to the high amount of parameters, which can be modified during the calibration process. This problem of overparameterisation is not recognised by Refsgaard et.al (1996), who claims that the key issue is the actual parameterisation procedure. In order to avoid the problem of over parameterisation, a rigorous parameterisation process, which reduces the calibration parameters to a minimum, is of outmost importance for the distributed models (Refsgaard et. al., 1996; Refsgaard, 1997).

2.1.3. Upscaling procedure for DAISY

When using a one-dimensional physically based model for estimating nitrogen leaching from large catchments, the use of an appropriate upscaling methodology is a major challenge. The subject has received a lot of attention in the literature (e.g. Blöschl and Sivapalan 1995; Beven 1995; Beven and Fisher 1996; Reefsgaard et. al. 1999). A major concern is whether small-scale physical theories can be applied also at large scale. For example, upscaling of the Darcy's law considering matrix flow can be misleading if macro pore flow becomes important with increasing scale (Sivapalan and Kalma 1995). Another critical point is the description of spatial variability. When the scale of model application increases, the amount of data available for model parameterisation generally becomes less. Thus, when small-scale models use site-specific point measurements, large-scale models generally use "effective parameters", which aggregate and account for spatial variability.

According to Beven (1995), the approach of using effective parameters in microscale equation for representing processes at the macro scale is inadequate. On the other hand, Refsgaard (1999) found that a combined upscaling/aggregation procedure provided good simulation results of water balance and nitrate concentration distribution in groundwater. However, the applied upscaling/aggregation methodology was also found to have some limitation with respect to hydrograph shape. His finding was based upon two case studies of large-scale simulation of groundwater contamination from nitrate. The adopted methodology involved an upscaling from point to field scale using effective parameters. The subsequently upscaling from field to catchment scale did not employ a different process description, but rather an aggregation procedure preserving the spatial distribution of soil types, vegetation and agricultural practices.

Figure 4. Schematic illustration of the modelling approach. The regional scale modelling was not part of this works, which provides the basic for a further exercise to be skipped out as a follow up project

In this work, a nested modelling approach has been adopted (see Figure 5), and the upscaling of DAISY to represent conditions at the large catchment scale involved a two-step procedure:

First, the capabilities of the DAISY model to describe the nitrogen root zone processes were evaluated carefully at field scale. The field was interpreted as a homogeneous soil column characterised by one set of soil parameters. The spatial variability within the field scale was assessed from measured data, aggregated and accounted for in the model by "effective parameter" values. Then, the assumption of upscaling point scale equation to field scale was carefully evaluated by comparing simulated and measured data.

Second, DAISY was applied at the catchment scale. This procedure assumed that the process descriptions validated at the field scale could also be applied at the catchment scale. A sensitivity analysis was then carried out in order to determine the impact of the various catchment characteristics on nitrogen leaching. The results of such analysis led to the identification of the various soil units assumed to be homogeneous from a leaching point of view. Various cropping sequences were then applied at the identified soil units and for each of these combinations estimates of nitrogen leaching were calculated with DAISY. Finally, an overall nitrogen balance was set up for the whole study area, and the adopted methodology for the large scale application of DAISY was evaluated by comparing measured and estimated nitrogen fluxes.

2.1.4. MIKESHE/DAISY

The two models were run separately and the data transferred from one model to the other through an iterative process.

MIKESHE is first calibrated at the catchment scale in order to ensure an acceptable description of the overall hydrological processes. In the initial calibration phases standardised time series of crop parameter (leaf area index and rooting depth) are used. These parameters are later modified according to the data obtained from the DAISY model.

Within the study area various combination of cropping sequences, soil type and depth to ground water table were identified, and DAISY simulations is carried out for each of these combinations. Subsequently, time series of leaf are index and rooting depth were retrieved from the DAISY model output and distributed over the MIKESHE grid according to their occurrence. The generated crop parameters were then used in MIKESHE to recalculate the actual evapotranspiration. On the other hand information groundwater dynamics were important for the DAISY simulations in the study area. Time series of ground water table were therefore calculated with MIKESHE, and applied as lower boundary condition for the hydrological module within DAISY

2.2. Study area

Most of the study area, situated in delta of the Po River, is below the sea level. An integrated channel network and various pumping stations are draining the land, and the water deriving from approximately 880 km^2 of mainly agricultural land is pumped directly into a coastal lagoon called Sacca di Goro (See Figure 5.).

In the Sacca di Goro conditions favouring formation and growth of algae can be found, e.g. shallow water (1-2 meters) with limited water circulation, little water exchange with the sea and with large supply of nutrients deriving from the agricultural land. The lagoon is affected by eutrophication and since 1987 massive growth of Ulva Rigida and several dystrophic crisis have been observed (Viaroli et. al. 1994). In recent years, these eutrophication phenomena have become even more intense and frequent (Viaroli et. al. 1996). All this has important economic impacts, since the lagoon is intensively used for shellfish production. The dystrophic crisis of 1987 caused an economical loss of about 10 millions of EURO (Bencivelli, 1990). In 1997, the Italian President of the Council of Ministers declared the state of emergency for the Sacca di Goro; at that time the estimated economic losses ranged between 7.5 and 10 millions of EURO (Bencivelli, 1998).

Several research projects analysing the eutrophication processes and water quality have been carried out in the Sacca di Goro (Bartoli et. a. 1996 and 1997; Christian 1998; Viaroli, 1992 and Varioli et. al. 1993; etc.). However, very little investigation concerning the actual source of the problem, e.g. the nutrients deriving from the inland drainage basin have been carried out.

To reach the objectives of the project as discussed earlier in the text, the establishment of a modelling system at the scale of the entire drainage basin of the lagoon is necessary. However, the present thesis aims at setting up a model at a smaller scale, as a first step in the implementation of the nested approach. To this purpose, a subcatchment called Bonello was selected (Figure 5.) The Bonello catchment has a surface area of about 26 km^2. On the northern and eastern side it is bordered by a branch of the Po River (Po di Goro), on the southern side by the lagoon (Sacca di Goro), and by a forest (Gran Bosco della Mesola) on the western side.

The Bonello area is one of the few subcatchments within the whole drainage basin that can be considered as hydrological independent with well-defined boundaries for the groundwater flow as well as an independent channel network. Moreover, it has the advantage of good data availability, and that very good local contact exist, which facilitate the data collection process. Finally, many of the catchment characteristics (soil types, hydrological conditions, agricultural and irrigation practices) are representative for the whole drainage basin of the Sacca di Goro lagoon.

The Joint Research Centre (JRC) has contributed with other research projects for the characterisation of the area. Piccapietra et. al. (1997) used the PESTRAS (Freijer and Van der Linden, 1996) model for analysing the transport and transformation process of selected pesticides. Moreover, Leip (2000) measured and modelled greenhouse gas emissions from both open waters (Sacca di Goro) and agricultural soils.

Figure 5 Study area - Bonello. Catchment boundaries is indicated by the blue line, the major channels by the white lines, whereas* ![pump]** ***indicate the position of the various pumping stations. The two villages "Goro and Gorino", as well as the forest "Gran Bosco Della Mesola", are also indicated on the map.

2.2.1. Catchment characteristics

The study area is very flat with a surface topography ranging from 0-3 meters below the sea level. The eastern and southern parts of the area are surrounded by high dikes, which prevent flooding from the higher located Po di Goro and Adriatic Sea. The area has a dense channel network acting both as drainage and irrigation system. Irrigation water, taken from external sources (Po di Goro and Canale Bianche), is entering the channel network through various siphons, and all drainage water is pumped into the lagoon by a pumping station situated at the river outlet. The area receives approximatly 679mm precipitation a year and during the summer period another 224 mm is applied in terms of irrigation water.

According to the Regional soil map of the "Regione Emilia Romangna" the dominating soil types in the area are Calceric Arenosols and Calceric Cambisols (FAO classification) covering 46% and 44% of the area respectively (Filippi and Sbarbati, 1994). A more thorough description of the various soil characteristics is given in section 2.3.

The study area comprises three major land use classes: agricultural, urban and forest covering respectively 85, 5 and 10 % of the area (Figure 6.). The landuse for 1994 - 1998 as provided by local farmers (Barboni, 1999; Gastone 1997, Personal Communication) are reported in Table 2. It should be noted that the data from 1995 and 1994 in Table 2. refer to the area south of the village Goro, as no data describing the northern part of the Bonello area were available.

The area is characterised by having one crop sequence a year. Mineral nitrogen is generally used as fertiliser, with ammonium fertiliser and urea as predominant types. The recommended application rates are given in Table 2.

Table 2 Relative distribution of the various crop types (% of arable land) and recommended application rates of mineral nitrogen

	1994*)	1995*)	1996	1997	1998	Application rate of mineral nitrogen (kg N/ha·year)
Maize	48	45	45	44	37	200
Soya bean	8	5	17	16	8	50
Alfalfa	15	20	10	16	25	0
Barley	0	1	2	2	5	100
Wheat	19	17	2	2	6	100
Sugar beet	9	6	6	12	11	100
Rice	1	5	3	4	5	65
Honey Melon	0	0	3	0	1	100
Asparagus	0	0	3	2	0.5	120
Tomatoes	0	0	3	2	0.5	150
Poplar	0	0	4	0	0	-
Vineyards	0	0	2	0	1	-
No data	64	64	-	-	-	-

**) The data for 1994 and 1995 refer to the area south of the village Goro, as no data describing the northern part of the Bonello Catchment were available*

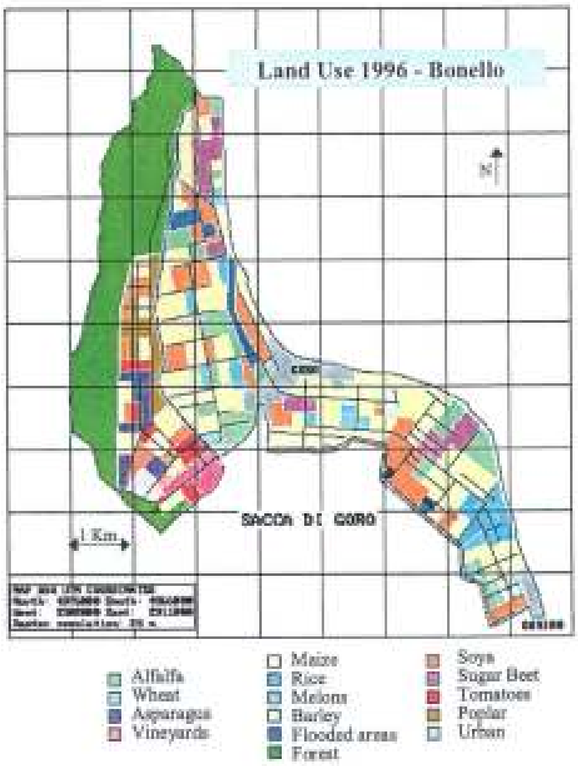

Figure 6. Land use map of the Bonello catchment 1996. The two villages "Goro and Gorino" are also indicated on the map (Modified from Piccapietra et. al. 1998).

The terrain forms an alluvial aquifer containing quaternary sediments. According to the description of 23 soil core with a depth of 6 meter (Massellani, 1984) two different sedimentation regimes could be identified:

- The eastern part of the area is characterised by a higher topographical level as well as a higher terrain slope. The available core descriptions indicated that sandy topsoil, characterised as fine sand, was overlaying sandy sediments. These sandy sediments were only separated by a very thin clay layer situated at about 5-meter below surface.
- The western part of the area is characterised by a lower topographical level as well as a lower terrain slope. Here loamy topsoil was found to overlay a highly heterogeneous aquifer characterised by a fine sequence of clayey, loamy and sandy sediments.

Unfortunately, no information concerning the precise position or the topographical levels of the analysed soil cores were available. Hence, the spatial distribution of the various layers could not be assessed from these data.

2.3. Acquisition of soil input data

Both MIKESHE and DAISY requires a description of the physical soil properties in terms of retention and hydraulic conductivity curves. In addition, DAISY requires information on clay content, total carbon content and C/N ratios. Since MIKEHE was calibrated at the catchment scale, whereas the calibration of DAISY took place at four 1 ha calibration sites (See Figure 7.), two different approached to estimate soil properties were adopted according to the scale of application.

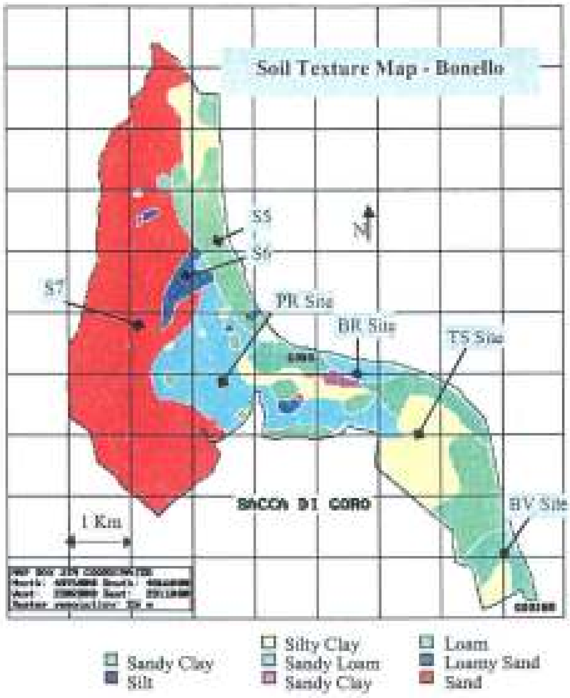

Figure 7. Soil texture map of the Bonello catchment (modified from Piccapietra et. al. 1998). The soil samples position including the four calibration sites named BR, TS, BV and PR are indicated on the map. The soil map was obtained from Consorzio di Bonifica 1° Circondario di Ferrara and the soil classification reported accordingly to the Shepard classification system (Shepard, 1954).

Catchment Scale: The available soil data comprised a 1:25.000 soil texture map based on soil texture analysis from the top 50 cm soil (Figure 7). A soil profile description was carried out for each of the main soil types identified on the available soil map. In total seven profiles description were carried out throughout the area. Four of these were situated in the calibration sites (PR, TS, BV, and PR site) whereas the remaining three (S5, S6 and S7) were situated in the remaining soil types present in the Bonello catchment (Figure 7.). The profile description focused on the hydraulic and chemical soil properties required by the models. Consequently, the soil was sampled down to one meter, and from each of the identified horizon non-disturbed samples were taken for the analysis of the retention curve and bulk density. Soil samples were also taken for analysis of soil texture, organic carbon and C/N ratios.

Field Scale: At the four calibration sites a more intensive sampling was conducted. The calibration sites BV, TS and PR were subdivided into nine equal plots. From each of these the soil was first sampled down to 90 cm depth at intervals of 0-30 cm, 30-60 cm and 60-90 cm. Soil samples were taken from each of the 30 cm layers, and analysis of organic carbon, total nitrogen, pH, cation exchange capacity (CEC), bulk density and texture were then performed on four of the nine cores (Table 3.). The texture analysis from the four cores indicated a rather small little spatial variability at each calibration site, and the measurement of the hydrological properties was carried out at one location at each calibration site. The calibration site BR was included at a later time, and all input data derived from a single soil core only.

The sample position of the various profiles is indicated in Figure 7. whereas the measured chemical and hydrological soil properties are given in Table 3 , Table 4 and Figure 8.

Table 3. Chemical soil properties. The values for the PR, TS and BV sites are averages of four replicates. The sample position is indicated in Figure 7.

Depth (cm)	C org. (%)	C/N	CEC*) pH 8,1 (meq/100 gr)	PH (H_2O)	pH (KCl)
			Site PR		
0 - 30	1.5 ±0.22	10 ±1.0	15.8 ±0.7	8.0 ±0.16	7.4 ±0.03
30 - 60	0.86 ±0.17	8.6 ±1.2	-	8.0 ±0.13	7.5 ±0.05
60 - 90	0.52 ±0.15	7.7 ±1.7	-	8.0 ±0.06	7.6 ±0.06
> 90	0.13	21	-	8.6	8.3
			Site BR		
0 - 30	1.2	9.8	14.4 ±0.4	8.0	7.5
30 - 60	1.0	9.5	-	8.1	7.6
60 - 90	0.55	8.3	-	8.2	7.8
> 90	0.083	28	-	8.6	8.4
			Site TS		
0 - 30	1.5 ±0.19	10 ±0.71	15.4 ±1.4	8.0 ±0.10	7.3 ±0.05
30 - 60	1.3 ±0.16	9.8 ±1.2	-	7.9 ±0.08	7.4 ±0.04
60 - 90	0.71 ±0.44	7.5 ±1.5	-	8.0 ±0.12	7.6 ±0.16
			Site BV		
> 90	0.12	14	-	8.5	8.3
0 - 30	1.3 ±0.25	9.0 ±0.81	12.1±1.5	8.2 ±0.10	7.4 ±0.14
30 - 60	1.1 ±0.41	8.6 ±1.6	-	8.2 ±0.16	7.5 ±0.09
60 - 90	0.7 ±0.53	7.6 ±2.2	-	8.4 ±0.32	7.7 ±0.23
			S5		
0 - 30	1.9	11	-	-	-
30 - 50	1.2	7.0	-	-	-
50 - 90	0.74	7.3	-	-	-
			S6		
0 - 30	1.9	12	-	-	-
30 - 60	1.2	11	-	-	-
60 - 90	0.77	8.6	-	-	-
			S7		
0 - 30	1.6	33	-	-	-
30 - 60	1.0	29	-	-	-
60 - 90	0.53	-	-	-	-

**) The CEC was measured in the upper 5 cm of the soil*
"-" Indicates that the given properties was not measured

Table 4 Physical soil properties. The sample position is indicated in Figure 7.

Depth (cm)	*Soil type USDA*	*Clay (%) < 2μm*	*Silt (%) 50- 2μm*	*Sand (%) 2000-50μm*	*Porosity*	*Ks [*1] (m/s)*
			Site PR			
0 - 50	Silty Loam	22	59	19	0.462	5.0E-07
50 - 80	Silty Loam	5	76	19	0.480	8.5E-07
80 - 100	Sand	3	7	90	0.468	1.0E-04
			Site BR			
0 - 50	Silty Loam	23	55	22	0.450	3.6E-07
50 - 80	Silty Loam	4	70	26	0.440	8.9E-07
80 - 100	Sand	1	0	99	0.451	1.8E-04
			Site TS			
0 - 40	Silty Loam	22	55	23	0.416	5.9E-07
40 - 80	Silty Loam	22	55	21	0.427	3.0E-07
80 - 100	Sand	1	4	95	0.422	1.1E-04
			Site BV			
0 - 40	Silty Loam	22	68	10	0.516	5.1E-06
40 - 80	Silty Loam	21	73	6	0.510	1.2E-06
80- 120	Sandy Loam	2	48	50	0.499	5.6E-06
120 - [*2]	Sand	3	7	90	0.468	1.0E-04
			S5			
0-40	Silty Loam	19	60	21	0.432	3.3E-07
40-90	Silty Loam	1	78	21	0.445	8.9E-07
90-	Sand	3	7	90	0.45	1.0E-04
			S6			
0-50	Sandy Loam	4	56	40	0.443	2.1E-06
0-85	Sandy Loam	4	56	40	0.447	2.3E-06
85 -	Sand	3	7	90	0.468	1.0E-04
			S7			
0-40	Sand	3	1	96	0.444	3.6E-04
40-100	Sand	3	1	96	0.405	1.3E-04

*[*1] The saturated hydraulic conductivity (Ks) were estimated by pedotransfer functions (Rawls and Brakensiek, 1989),*
*[*2] The soil properties were not measured in this sand layer. For the model setup the soil properties were instead adopted from the sandy layer at the PR-site*

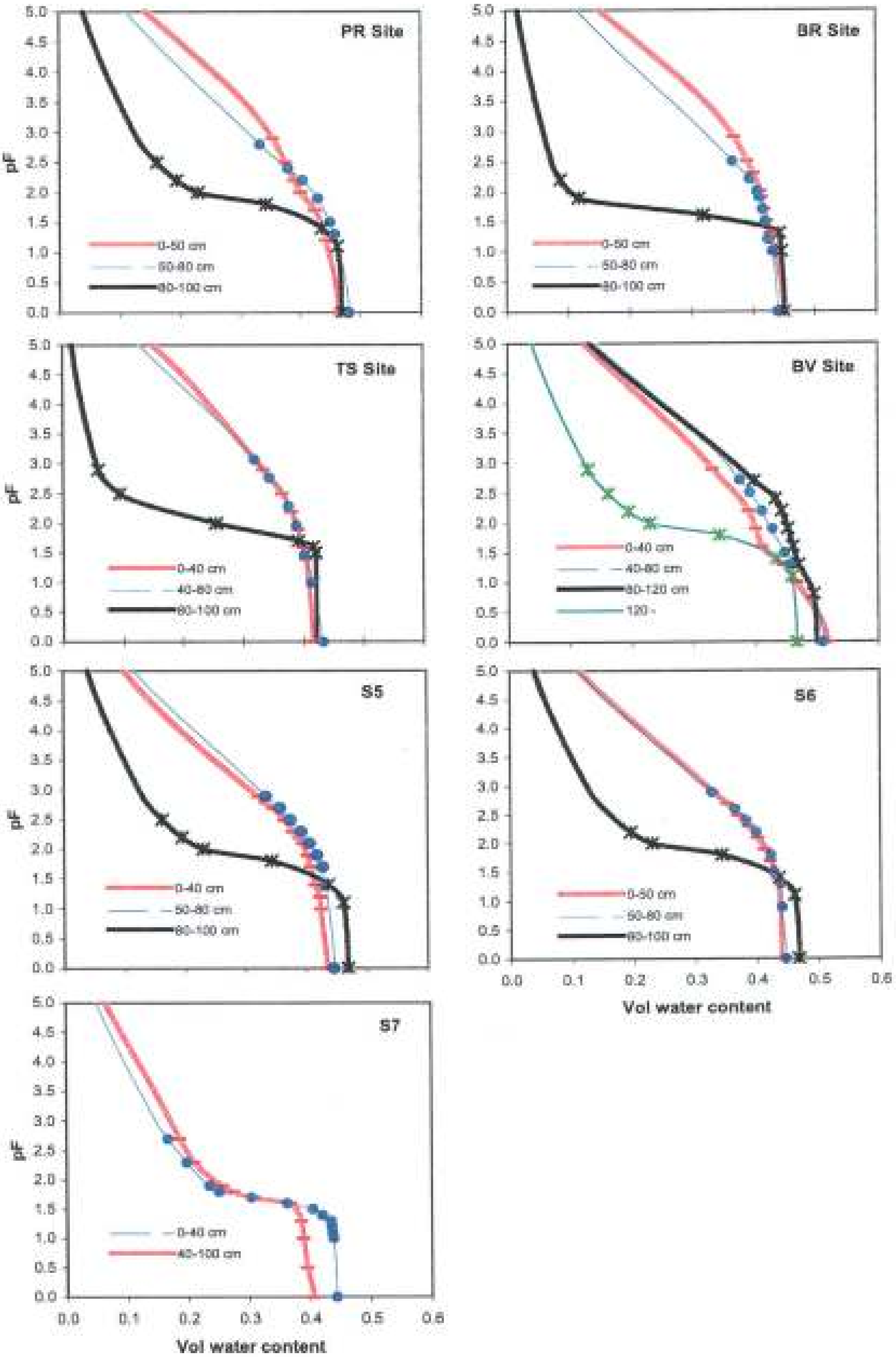

Figure 8 Soil retention curves for various soil profiles. Points indicate measured data, whereas the solid lines indicate the fitted retention curve using the Spline interpolation method (Brodlie, 1980).

2.4. Acquisition of calibration data

The hydrological model MIKESHE was calibrated using timeseries of groundwater levels and channel discharge data. The accumulated weekly channel discharge from January 1993 and onwards were obtained from Consorzio di Bonifica (1° Circondario, Ferrara). Moreover 10 piezometers were installed throughout the area in spring 1997, and the ground water table was measured on a regular basis every two weeks It should be noted, that the topographical level of the 10 piezometers were not measured in field, but estimated from the available 1:5000 topographical map. Consequently, the actual level of the measured groundwater table has some uncertainties.

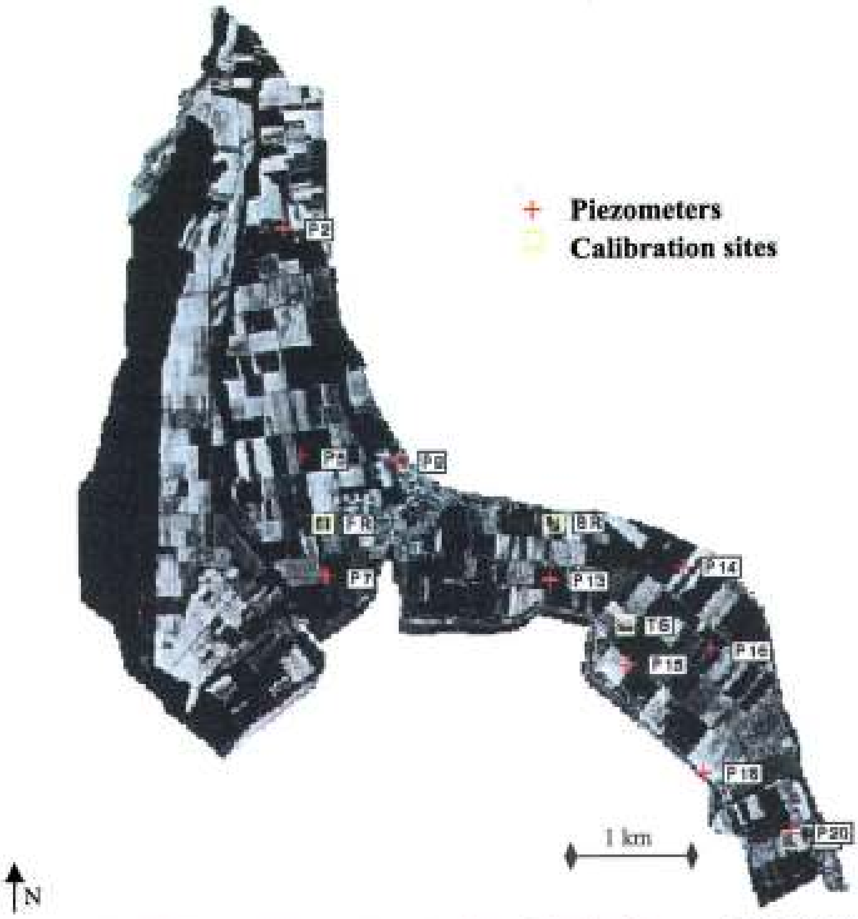

Figure 9. Position of the installed piezometers as well as the four calibration sites named PR, BV, TS and BR (Modified from Piccapietra et. al. 1998).

The nitrogen model DAISY was calibrated using soil water content, mineral nitrogen concentration, dry matter yield, plant nitrogen uptake and soil temperature

Inorganic soil Nitrogen and soil water content: For the measurement of inorganic nitrogen (NO_3 and NH_4) each of the 1 ha calibration sites were subdivided into nine equal plots. From each of this subdivision the soil samples were taken down to 90 cm using interval of depths of 0-30 cm, 30-60 cm and 60-90 cm. All nine soil cores were pooled into a single sample for each 30 cm layers. As described in section 2.3. the sample strategy at the BR site was quite different. The soil sampling concentrated here at one local spot, and the calibration data derives from one soil core only. The measurements were carried out three times during the growing season in connection with fertiliser application and two times during winter leaching period in between the growing season. These periods were selected for describing the temporal variation in nitrogen concentration in the soil, in sufficient detail to elucidate interactions of the system with fertiliser application, climate and crop growth. Each of the soil samples was gravimetrically analysed for soil water content.

Crop samples: Crop samples were selected from three of the calibration sites (BR, BV and PR). Six plants were cut just above ground and each of these was measured for dry matter yield and total nitrogen content. The plant density (plant/m^2) was determined by counting all plant present in a 30-m^2 test plot.

Soil Temperature: Values for soil temperature were obtained from Leip (2000) who measured the soil temperature in the upper 20 cm of the soil at the BR site. The soil temperature was measured 6 times during the simulation period by means of mercury or alcohol thermometers. For each camping the soil temperature was measured 18 - 24 times during the day and subsequently averaged for obtaining the mean daily temperature

2.5. Methods of analysis

The soil samples for texture and chemical analysis (Organic Carbon, Total Nitrogen, pH, Salinity, CEC) were stored at 2 -4 °C and transported to the laboratory, where they were air dried, sieved through 2 mm sieves and analysed as follows:

- *Organic carbon* was analysed using dichromate oxidation according to the Walkley-Black method (Page, 1982).
- *pH(H_2O)* was measured in a slurry having a soil: water ratio of 1:2,5 (Page, 1982).
- *pH(KCl)* was measured in a 1 M KCl slurry having a soil: KCl ratio of 1:2,5 (Page, 1982).
- *Total Nitrogen* was determined with the Kjeldahl method; the digestion was carried out at 410 °C and SeO_2, K_2SO_4, $CuSO_4$. $5H_2O$ were used as catalysts (Page, 1982).
- *Soil texture*: The sand fraction was measured directly by sieving a sample suspension. The remaining silt and clay fractions were determined by sedimentation analysis following the pipette method described by (Page, 1982). Prior to analysis, carbonates were removed by HCl pre-treatment. The presence of organic matter in the soil can cause uncertainties on the estimation of the clay and silt fraction. According to Madsen (1983) these uncertainties ranges from 1%- 2% for an organic matter content of less that 5%. As these uncertainties are acceptable for the present scope of analysis, the organic matter was only removed if it exceeded 5 %.
- *CEC* was determined by the $BaCl_2$ methods (Ministerio delle Risorse Agricole, Alimentari e Forestali, 1994), where soil samples were extracted by a Bariumchloride/Triethanolamine solution at pH 8.1.

Inorganic Nitrogen: The soil samples were extracted with 2 M KCl using 10 ml of extractant/g of soil. The suspension was subsequently filtered (Whatmann 42 filter) and the filtrate was analysed for NH_4 and NO_3. The KCl extraction and the following analysis were all carried out in three replicates. The NO_3 was analysed by ion chromatography, whereas NH_4 was analysed spectrophotometically in accordance with Graffhoff (1972). In order to avoid interference - due to the high Cl^- concentration - the samples were diluted 5 and 25 times prior to NH_4 and NO_3

analysis, respectively. The KCl extraction was preferably carried out in the field in order to minimise the changes of the Nitrogen content in the soil. However, in one situation (Autumn 1996) this was not possible. In this case, the samples were stored at 2 -4 °C and transported to the laboratory where extractions were carried out the following day. Outside the fertiliser period, the measured ammonium concentration in the soil was very low and often below the detection limit. In these cases the soil ammonium concentration was assumed to be equal to the detection limit. A procedure associated with some uncertainties. However, uncertainties in such a low concentration range did not have great impact on the results.

Soil water retention curve: Undisturbed soil samples were taken with 5 cm long metal cylinders of 98 cm^3. The soil water retention curve was subsequently determined in the laboratory using the evapotranspiration method (See Appendix I for methods description). The present methods only allowed pressure measurement down to -875 cm water (pF 2,94). Thus, the water content for lower capillary pressure was estimated by fitting the measured data with the Spline interpolation method (Brodlie, 1980), where the soil water content at pF 7 was assumed to be zero.

Hydraulic conductivity curve: Initial values for the saturated hydraulic conductivity was assessed with the Rawls and Baumer equation (Rawls and Brakensiek, 1989) based on the porosity, clay and sand fraction. The hydraulic conductivity function was subsequently approximated by a methods described by Kunze et. al. (1968)

Bulk density: Soil samples were taken at various depths with 5 cm long metal cylinders of known volume (98 cm^3). The samples were dried at 105 ^0C until constant weight for dry mass determination.

Crop analysis: Fresh plant material was brought to the laboratory and dried at 75 ^0C until constant weight. Dry mass was determined and all material was finally grounded prior to nitrogen analysis. The nitrogen content was measured with the Kjeldahl method, the digestion was carried out at 410 °C and SeO_2, K_2SO_4, $CuSO_4$. $5H_2O$ were used as catalysts. (Page 1982).

3. Hydrological modelling

Modelling of the catchment hydrology using MIKESHE aimed at quantifying the hydrological processes by deriving the water balance for the entire area, and to obtain groundwater table maps for the subsequent estimation of nitrogen leaching from the root zone.

3.1. Model setup and calibration

The model set-up of MIKESHE is described in the following, along with an indication of the selected calibration parameters.

3.1.1. Discretization

The study area was divided into 2612 grid squares of 100 m. All spatially distributed catchment characteristics (topography, soil type, land use etc.) were transformed into discretised representations at this grid scale. The vertical discretization of the soil profile in the unsaturated zone varies between 5 cm in the upper part near the soil surface to 30 cm in the lower part.

3.1.2. Meteorology

Primary meteorological variables for the model (daily values for precipitation, mean air temperature, global radiation and potential evapotranspiration) refer to the Volano meteorological station situated eight kilometres east of the study area. A station, considered to be representative for the Bonello Catchment. The potential evapotranspiration was calculated using the Penman methods. In periods of missing data the meteorological data were complemented with those possibly available from the station Bosco Mesola situated just east of the study area. The variation between precipitation data obtained from the two meteorological stations during one year was only of 6%. The daily precipitation is reported in Figure 10, whereas accumulated precipitation and potential evapotranspiration are given in Figure 11. The average precipitation during the simulation period was 679 mm with an average of 340 mm during the growing season (1 April – 1 September).

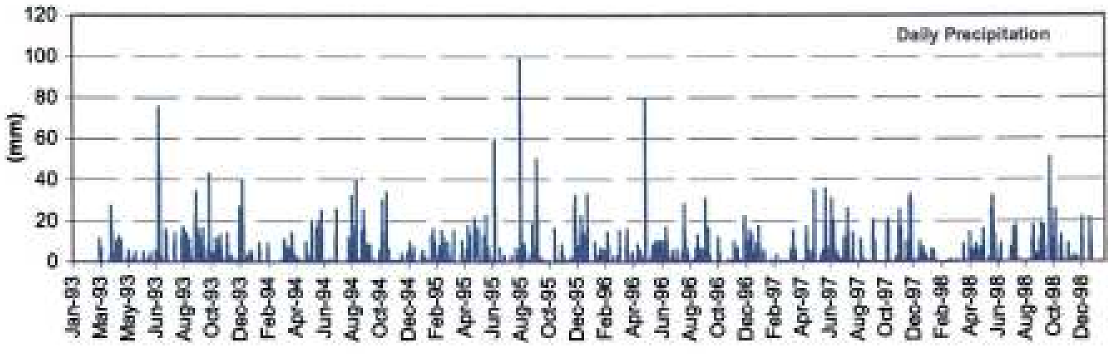

Figure 10. Daily precipitation during the simulation period (1993-1998)

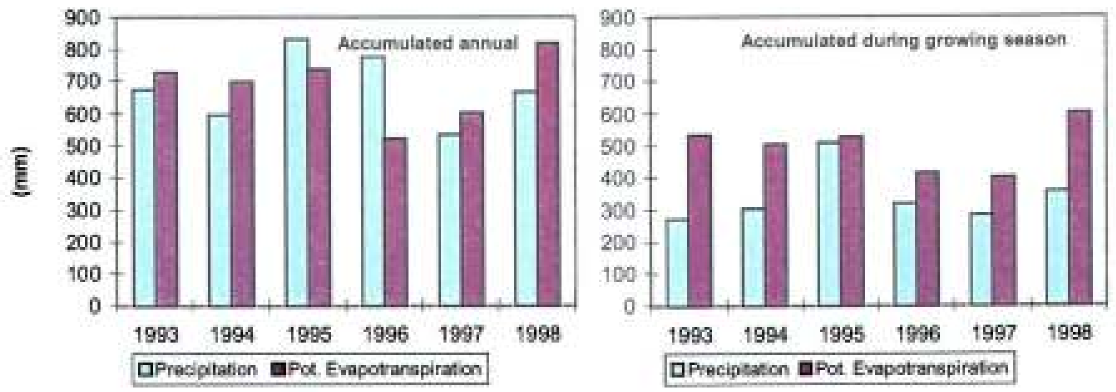

Figure 11. Accumulated precipitation and potential evapotranspiration during the simulation period (1993-1998).

3.1.3. Vegetation and Evapotranspiration

A description of the temporal and spatial distribution of Leaf Area Index (LAI) and Rooting Depth (RD) for the various crop types are used for calculating the actual evapotranspiration in MIKESHE.

For Maize, Alfalfa and Sugar Beet the data were retrieved from DAISY simulations. The temporal and spatial distribution of these crop types was described by applying 8 different cropping sequences on various soil units. These were classified as combination of different soil types and depth to the ground water table (see further description in Section 4.2.1.). The soil type and depth to ground water table did not have any notable effect on the development of LAI and RD. For each of the eight cropping sequences only one LAI/RD time series was therefore selected and assumed to be representative for all soil units with the same cropping sequence.

For Wheat, Barley and Soya time series of LAI derived from a measurement campaign carried out near Bologna (ENEL, 1997), whereas time series for RD were assessed from the literature (Neitsch, 1999). The LAI/RD time series for the remaining crop types i.e. rice, forest, tomatoes were also taken from the literature (Neitsch, 1999).

The empirical evapotranspiration and interception storage parameters were assumed to be identical to the default values given in the MIKESHE technical reference manual. According to Reefgaard (1997), these values have been successfully used in various modelling studies in many countries. These parameters were not subject to any calibration.

3.1.4. Unsaturated zone

The soil physical characteristics for the unsaturated zone were adopted directly from the large-scale application of DAISY, and data description can be found in section 4.2.1. However, the hydraulic conductivity curve used in the MIKESHE simulation was somewhat different from that used in the DAISY simulation. Indeed, DAISY used the Kunze method (Kunze et. al., 1968), whereas the MIKESHE model only allows the Brooks and Corey method (Brooks and Corey, 1964) for estimating conductivity curves. DAISY generally gave a good description of the soil water dynamics in the unsaturated zone. Therefore, the conductivity curve in MIKESHE was matched to the curve used in the DAISY simulations by fitting the hydraulic exponent “n” in the Brock and Corey equation. Apart from this adjustment of the conductivity curve, the data describing the unsaturated zone have not been subject to any calibration.

The unsaturated zone calculation was only carried out for a number of selected grid points (vertical profiles) in order to reduce computational time requirements. An initial simulation was carried out and areas of similar characteristics in terms of depth to ground water table were classified. Calculations were then carried out only in a single grid within each of the classified areas, and the results were transferred to all other grid within such area. As a total unsaturated zone calculations were carried out for 69 computational profiles.

3.1.5. Topography, overland and channel flow

The topography and the river network were digitised from a 1:5000 topographic map[1]. Moreover a large amount of geometrical data describing the cross-section and bed elevation for all major channels were obtained from Consorzio di Bonifica 1° Circondario, Ferrara, and included in the description of the river network (See Figure 12)

The channel outlet was placed outside the model boundaries and the bottom ditch level was lowered to ensure a decreasing slope towards the channel outlet. A constant head boundary condition was then applied at the channel outlet. Clearly this is not representative of the real condition where the pumping station starts to operate when the water level in the main channel exceeds a specified level. However, it was the only way that MIKESHE could simulate this “maximum level boundary condition” where water is removed when a certain level is exceeded but no water enters to compensate for falling levels during dry periods.

Large parts of the arable land consist of a complex and very dense network of drainage/irrigation ditches. Only the major channels were included in the river setup, whereas the small ditches were included through the drainage facility of MIKESHE. A drainage level was defined for each grid and whenever the simulated groundwater table exceeded the drainage level, a drainage flow was produced. Drainage water was removed from the aquifer and routed either to the channels or to the model boundaries. The amount of drainage water depends on the height of the water table above the drain and a specified time constant characterising the density of the drainage system and the permeability conditions around the drains. The drainage level and time constant applied in the model-setup were 70 cm and $4.8E^{-07}$ s^{-1} respectively. The level of 70 cm represented the minimum depth to the groundwater measured in the area, whereas the value for the time constant derived by calibration.

A Manning number of 40 $m^{1/3}$ s^{-1} were applied in all the area. However, the flooded areas in the northern part of the catchment was accounted for by applying a very small manning number of 1E-5 $m^{1/3} s^{-1}$.

The exchange flow between the river and the groundwater was calculated assuming a “full contact” between the river and groundwater. Hence, the exchange flow between the groundwater and the river was determined from the hydraulic parameters of the geological layers adjacent to the river. The river/aquifer exchange was therefore explicitly given in the saturated module.

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[1] *Regione Emilia-Romagna, Carta Tecnica Regionale, Sezione No 187160, 188130, 188090; provided by the Assessorato all'Ambiente, Ferrara*

Figure 12. Channel network applied in the hydrological model MIKSHE. Irrigation water is entering the channel network through in eight upstream points marked as Qv, whereas the channel outlet is marked as Hv.

3.1.6. Irrigation

Irrigation water is taken from Po di Goro and Canale Bianche and it is entering the channel network through eight siphons situated along the two rivers. Since both channels are hydrological independent from the remaining channel system all irrigation water derives from external sources. The yearly amount of irrigation water applied to the southern part of the area (south of the Goro village) were obtained by Consorzio Di Bonifica (1° Circondario, Ferrara), whereas data on the irrigation water for the northern area (north of the Goro village) was unavailable. Thus, the amount of irrigation water used in this area was estimated by adopting the following assumptions reflecting the general irrigation practices used by the farmers:

- The irrigation rate for the rice fields was 1300 mm/year
- The irrigation rate for the sandy soils was three times higher that of the irrigation rate for loamy soils
- The irrigation rate for the sandy area was the same for all the corresponding area
- The irrigation rate for the loamy area was the same for all the corresponding area

Approximately 10% of the arable land south of Goro are rice fields. These fields are flooded from May to September, and water inside these fields is changed three to four times during the period. According to general irrigation practices the total yearly amount of water used for rice irrigation is approximately 1300-1400 mm/year (Remari 1999, Personal communication).

The northwestern - sandy - part of the area was characterised by a much higher irrigation demand. Exact irrigation data were not available, but according to general irrigation practices

(Remari 1999, personal communication) the irrigation rate for the sandy area was 3 – 4 times higher than that for the loamy soils.

The fixed irrigation rate for rice of 1300 mm/y, as well as the ratio of 3 between the irrigation rate for sand and loam soil, was assessed through calibration.

No irrigation data was available for 1993 and 1994. In these two years the meteorological conditions during the growing season were somewhat similar to 1998 (Figure 11). An initial MIKESHE simulation also indicated that the net precipitation in 1993 and 1994 were 17% higher and 5% lower than the net precipitation in 1998,. Consequently, the irrigation rate in 1993 and 1994 was assumed to be 16% lower and 4 % higher than the irrigation rate in 1998.

Based on the above assumption the total amount of irrigation water applied to the arable land was estimated according to *Eq. 1.*, and the obtained estimates are represented in Table 5.

$$Q_{Total} = (I_{Loam} \cdot A_{Loam} + I_{Sand} \cdot A_{Sand} + I_{Rice} \cdot A_{Rice}) \cdot 10^{-3} \qquad \textit{Eq. 1}$$

$$I_{Loam} = \frac{Q_{Sifone} - I_{Rice} \cdot A_{Rice} \cdot 10^{3}}{A_{sloam}} \qquad \textit{Eq. 2}$$

$$I_{Sand} = 3 \cdot I_{Loam} \qquad \textit{Eq. 3}$$

$$I_{Rice} = 1300 \quad mm \qquad \textit{Eq. 4}$$

Where:

Q_{Siphon} = *The amount of irrigation water applied in the area south of Goro* (m^3/*year*)

A_{Sloam} = *Area of the loamy arable soil south of Goro* (m^2)

A_{Rice} = *Area of the rice field* (m^2)

A_{Sand} = *Area of the sandy arable soil* (m^2)

A_{Loam} = *Areof the loamy arable soil* (m^2)

Table 5 Irrigation water applied in the southern area (Q-Siphon) as well as the estimated irrigation rates

	Q_{Siphon} *(m3/year)*	*Rice (mm/year)*	*Loam (mm/year)*	*Sand (mm/year)*
1993	-	1300	232	697
1994	-	1300	294	882
1995	764690	1300	69	207
1996	2036549	1300	161	484
1997	2596486	1300	271	812
1998	2618125	1300	280	840
Average	2003963	1300	213	639

During the irrigation period, from April to September, the siphons are normally left open allowing a continuous water inflow from the higher located Po di Goro and Canale Bianche into the area. The siphons are only closed during rainy periods or in case of high salinity concentration in the Po di Goro (Remari, 1999 Personal communication). Hence, the temporal distribution of the irrigation water was calculated by minimising theR^2 -value expressed in Eq. 5.

$$R^2 = \sum_{i=1}^{n}(Wi - A)^2$$ Eq. 5

Where:

$$Wi = QPi + QIi$$ Eq. 6

$$A = \frac{1}{n}\cdot(\sum_{i=1}^{n} QIi + \sum_{i=1}^{n} QPi)$$ Eq. 7

$$Constraints: \sum_{i=1}^{n} QIi = Q_{Total} \text{ and } QI_i \geq 0$$

Where:

Qp_i = *Weekly precipitation (m3/week)*

QI_i = *Weekly irrigation input (m3/week)*

n = *Number of irrigation week per year*

Q_{Total} = *Total yearly irrigation input calculated from Eq. 1 (m3/year)*

The irrigation water is taken from the higher located Po di Goro and applied to the field through the network of irrigation ditches. This irrigation system requires that a minimum water level is maintained in the main channel in order to supply all the smaller distribution channels. A certain tail loss in terms of irrigation water remaining in the channel system is therefore unavoidable. Information concerning this tail loss was not available. Thus, for the model setup the irrigation tail loss (QIC) and the fraction, which actually reached the fields (QIF), were estimated by the following procedure:

1. The simulated discharge at the channel outlet (QS_i) was retrieved from an initial simulation without irrigation input,
2. The weekly irrigation input (Qi_i) was estimated according Eq. 6,
3. The tail loss in terms of irrigation water remaining in the channel system (QIC_i) was calculated by minimising the R^2 value expressed in Eq. 8,
4. The amount of irrigation water actually reaching the field (QIF_i) was calculated by Eq. 9.

$$R^2 = \sum_{i=1}^{n}(QMi - (QSi + QIC_i))^2$$ Eq. 8

$$Constraints: 0 \leq QIC_i \leq QI_i$$

$$QIF = QI_i - QIC_i$$ Eq. 9

Where:

QMi = *The measured river discharge at the channel outlet* $(m^3/week)$

QSi = *Simulated river discharge from an MIKESHE simulation without irrigation input* $(m^3/week)$

QIi = *Total irrigation water estimated from Eq. 6* $(m^3/week)$

$QICi$ = *Irrigation water remaining in the channel system* $(m^3/week)$

QIF_i = *Irrigation water applied to the field* $(m^3/week)$

Given the amount of water applied to the fields each week (QIF) the irrigation rate for the rice fields (IW_{rice}), the loamy (IW_{loam}) and sandy soil (IW_{sand}) were estimated according to Eq. 12 - Eq. 11.

$$QIF_i = (IW_{Rice} \cdot A_{Rice} + IW_{Sand} \cdot A_{Sand} + IW_{Loam} \cdot A_{Loam}) \cdot 10^{-3}$$ Eq. 10

$$IW_{Loam} = \frac{QIF_i - A_{Rice}(1300/n)\cdot 10^{-3}}{(3\cdot A_{Sand} + A_{Loam})\cdot 10^{-3}} \quad \textit{Eq. 11}$$

$$IW_{Sand} = 3\cdot IW_{Loam} \quad \textit{Eq. 12}$$

$$IW_{Rice} = 1300/n \quad \textit{Eq. 13}$$

Where:

A_{Rice} = *Area of the rice field* (m^2)

A_{Sand} = *Area of the sandy arable soil* (m^2)

A_{Loam} = *Area of the loamy arable soil* (m^2)

The river network acts both as irrigation and drainage system. During the winter period water is flowing from the fields through the channel network toward the channel outlet, whereas - during the irrigation season - water is however flowing from the major channel into the small irrigation ditches. Here the irrigation water is applied to the field by channel release, and the irrigation distribution is controlled by a number of internal channel regulators. It was not possible to simulate this situation with the applied version of MIKESHE, as it would require modelling two different flow directions in the river system. Thus, a river setup describing the drainage system was applied. The amount of irrigation water applied to the fields (QIF_i) was applied through an extra precipitation input during the irrigation period, whereas the amount of irrigation water remaining in the channel system (QIC_i) was introduced directly in eight upstream points in the channel system (see Figure 12).

3.1.7. Saturated zone

Boundary conditions: The Po River and the Adriatic Sea are the natural boundaries for the ground water flow in the area. A large quantaty of water is entering the aquifer through these boundaries because of the strong hydraulic gradient between the study area (having a topography of 1 – 3 meter below sea level) and the higher located Po di Goro and Adriatic Sea. This inflow was accounted for by imposing a fixed ground water head as boundary condition at the border facing the Po di Goro and the Adriatic Sea. The hydraulic gradient -controlling the water inflow- was created by fixing this boundary ground water head at a much higher level that the adjacent cell situated just inside the model area. A constant level to this boundary ground water head was determined by calibration (see Table 6).

At the eastern boundary, which was assumed to follow the higher surface topography, a zero-flux boundary was applied. Moreover, the simulation of the saturated zone was carried out using one computational layer of six-meter depth, and the lower boundary was also treated like a zero-flux boundary.

Initial conditions: Using the iterative procedure suggested by Refsgaard (1997), the initial condition were assessed by the following approach:

- A model run for the period 1992- 1997 was made with guessed initial conditions
- Simulated groundwater levels for December 1997 were then used as initial conditions in a second model run
- If the simulated values in the two first run were not identical, a third model run was made on the basis of the initial conditions derived from the second run, and so on.

Since 1992 was used as initialisation period, the model results for this year were not used in the calibration process.

Hydrogeological properties: The aquifer was assumed to be unconfined and to consist of one main aquifer material characterised by the same hydraulic parameters. These parameters were assessed through calibration, as no data on the hydraulic properties of the saturated zone were available. The effect of the dikes surrounding the eastern and southern part of the area was simulated by introducing an impermeable lenses (K_S =1 E^{-8} m/s) in the saturated zone bordering the Po di Goro and the Sea. The upper vertical level of the lenses was siuated at the sea level whereas the lower vertical level was situated 75 cm below the ground surface of the model area.

Calibration procedure: The selected calibration parameters and their estimated values are given in Table 6. The dynamics of the ground water table is mainly determined by the water infiltrating through the boundaries, by the hydraulic conductivity and by the specific yield of the soil where the phreatic surface is located. However, due to the coupled unsaturated-saturated zone description in MIKESHE, where the two zones overlap, the specific yield of the aquifer is, in reality, a passive parameter determined by the soil moisture retention curve of the corresponding layer in the unsaturated zone (Refgaard, 1997). Thus only two groundwater parameters were subject to calibration:

- The hydraulic conductivity
- The applied boundary groundwater head at the eastern and southern boundaries

Within the river network the exchange flow between the groundwater and the river was determined from the hydraulic parameters (e.g. the hydraulic conductivity) of the geological layers adjacent to the river. Two of the piezometers (no.7 and no. 13) was placed very close to the main channel and the groundwater fluctuations in these piezometer was mainly used for calibrating the groundwater/river exchange flow. As indicated in Table 6, a higher hydraulic conductivity was applied along the main channel in order to increase the river/aquifer exchange flow.

Table 6. Calibrated input parameters in the saturated zone

Input parameter	Values
Boundary groundwater head towards the Po di Goro and the Adriatic Sea	1.2 meters belowe sea level
Vertical and horizontal hydraulic conductivity in the main aquifer	1E-04 (m/s)
Vertical and horizontal hydraulic conductivity along the main channel	5E-03 (m/s)

3.2. Results and discussion

The model calibration and validation results are reported in this section. The model performance was assessed on the basis of

- A visual inspection of the simulated and measured channel discharges and groundwater dynamics,
- The ratio between the average discharges of measured and simulated channel discharges ($Meas_{mean}/Sim_{mean}$),
- The Nash and Sutcliffe efficiency coefficient (r^2) between observed and measured channel discharges (Nash and Sutcliffe, 1970).

The available data set was divided into two parts. The calibration of channel discharge was performed for the period 1993-1997. Since the level of the groundwater table could only be measured only after drilling of the piezometers, calibration of the model with respect to simulation of the groundwater dynamics was made only for 1997. The validation of the model was then attempted using the data collected in 1998.

3.2.1. Channel discharges

The measured channel discharges were generally well represented by the model during the calibration period 1993-1997 (Figure 13). However, the results did highlight two inadequacies of the model. Firstly the model tends to underestimate some of the largest peak flows, and secondly the model gives a less satisfactory description of the channel discharge during the summer period.

It should be noted that some of the underestimated events occur where no precipitation data were available from the nearby meteorological stations (Volano and Bosco Mesola). In these periods, the data were instead obtained from more distant stations (Figure 13). The underestimation could therefore be caused by uncertainties associated with those precipitation data that might not be so representative for the study area.

The disagreement during the summer period (especially in 1996 and 1997) might be caused by an inadequate description of the irrigation input. Indeed, limited irrigation data were available and estimations were mainly based upon general irrigation practices (see section 3.1.6.). Moreover, the model setup adopted for irrigation is not fully representative of a situation where irrigation water is applied to the fields via channel release from a large number of ditches. It is the amount of irrigation water lost through evapotranspiration that is likely to be overestimated. The evapotranspiration losses are expected to be much higher when irrigation is applied on the surface. Thus, simulating irrigation through an increased precipitation input instead of channel release from irrigation ditches can overestimate the evapotranspiration.

During the winter seasons there was a good correlation between simulated and measured data (r^2=0,897). During these periods the channel discharge derives from precipitation and infiltration from the Po River and the Adriatic Sea. The latter is controlled by the model description of the boundary condition, and the good correlation therefore indicates an appropriate model description of the boundary condition.

During the validation period in 1998 the model were found to be less accurate (Figure 13.). Especially the autumn peak flows were clearly underestimated by the model. Again the observed discrepancies are presumably caused by an inadequate description of the irrigation input. The summer of 1998 was (compared to the other years) characterised by a very high evapotranspiration.

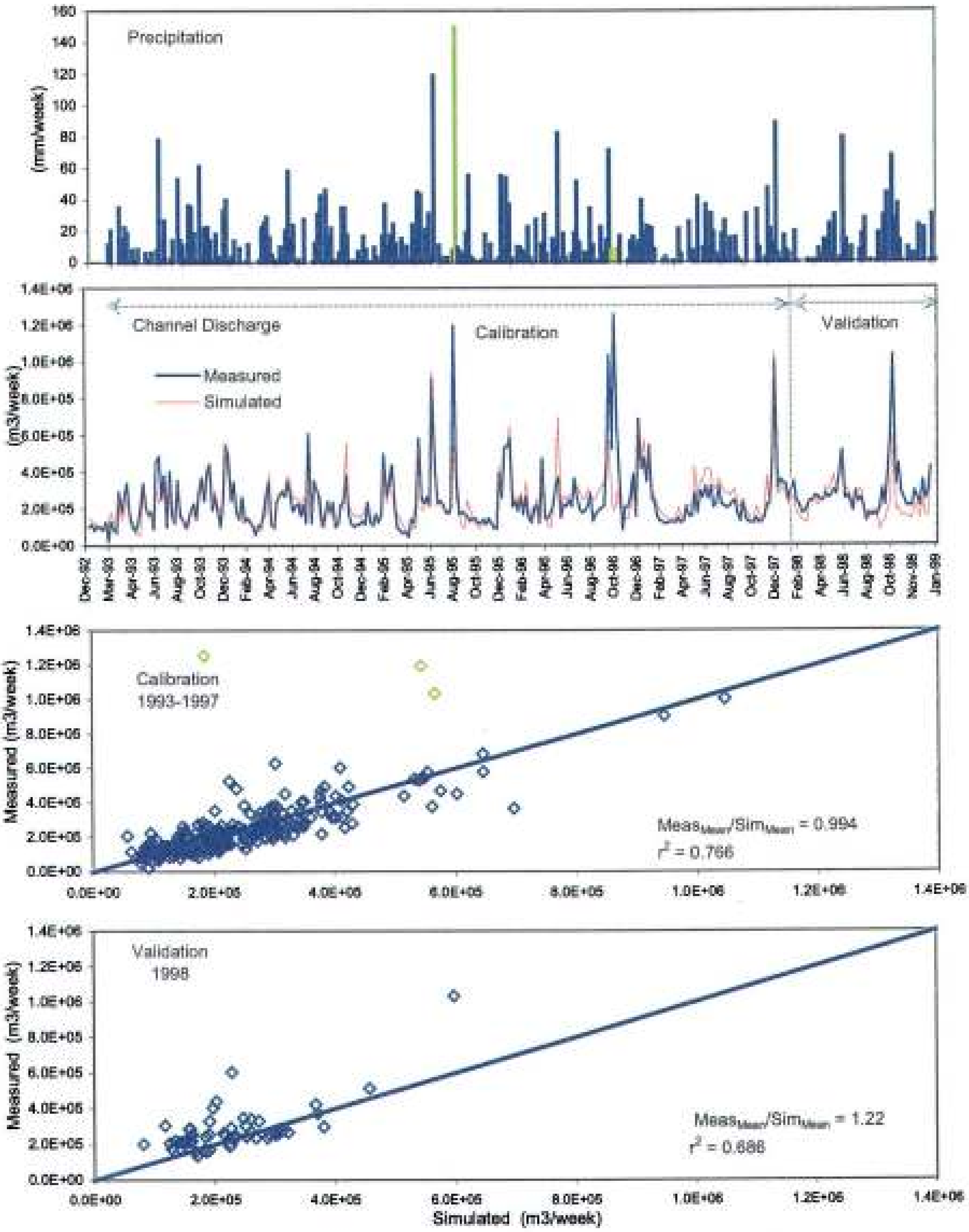

Figure 13. Simulated and measured weekly channel discharges. The bottom graphs illustrates the scatter diagrams from the calibration and validation period, whereas weekly precipitation is illustrated in the top graph. Marked in green are those periods where no precipitation data were available from the meteorological station normally used throughout the simulation period.

The estimated irrigation demand was also much higher (Table 5.), but apparently it was not sufficient to cover the large water deficit accumulated during the dry summer period. Thus, the first rain event in early autumn (September) did not result in a corresponding response in the channel discharge provided by the model. A key problem in this respect was that the meteorological condition during the summer month in 1998 was very different from the preceding calibration period (1993-1997), that was used for estimating the irrigating rates. In 1998 the summer months of June, July and August were abnormal in terms of a much higher potential evapotranspiration and a lower precipitation (See Figure 14.). The applied description of the irrigation system was sufficient for the meteorological variability found within the calibration period, but apparently it was not applicable for the more abnormal meteorological condition such as the summer of 1998.

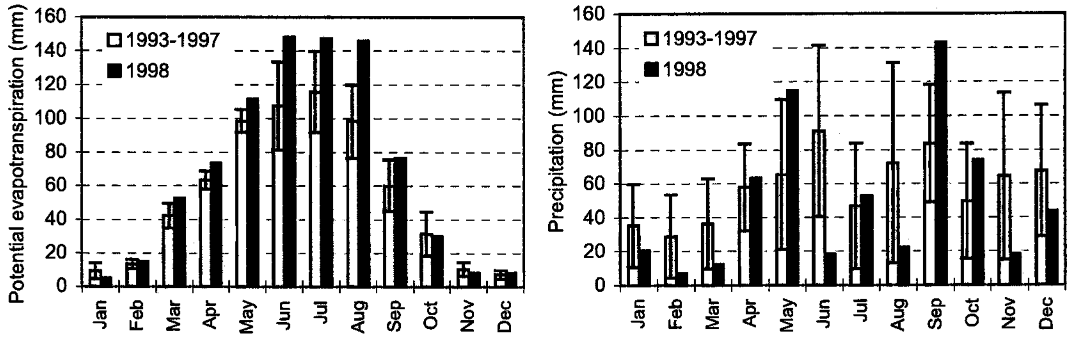

Figure 14. Monthly accumulated potential evapotranspiration and precipitation. Values from the validation year of 1998 are indicated with black bars, whereas the grey bars are averaged over the calibration period running from 1993-1997. Error bars indicate one standard deviation.

3.2.2. Groundwater dynamics

The model description of the groundwater dynamics was somewhat ambiguous. In most of the piezometers the correlation between measured and simulated groundwater table was good, while in others the correlation was less satisfactory (Figure 15 and Figure 16.). For the piezometer 5 and 16 the simulated groundwater table showed a more uneven response than the measured results. Moreover, the observed groundwater rise in July/August 1998 (Piezometer 15 and piezometer 18) and winter 1997 (Piezometer 9) was not reproduced well by the model.

The main reason for the observed discrepancies may be an inaccurate description of the spatial variability of the hydraulic conductivity. The aquifer was assumed to consist of one sedimentary bed characterised by the same hydraulic conductivity. Indeed, this is an obvious simplification, which makes it difficult for the hydrological model to capture the internal variation occurring within the catchment.

The observed discrepancies between measured and simulated data in piezometers 9, 15 and 18 are difficult to explain. The observed groundwater rise is only seen in these three piezometers, and sometimes the groundwater table starts to rise in a period with no precipitation input. It could be caused by an inadequate description of the internal channel regulation that - due to the simplified description of the river network - was not taken into account by the model description of the channel network. Moreover the irrigation rate was assumed to be the same for each soil type. Consequently, individual crop demand for irrigation water is - apart from the rice field - not taken into account by the model.

Finally, it should be noted that the topographical level of the installed piezometers was assessed from topographical map only. The actual level of the measured groundwater table therefore is associated with some uncertainties. These uncertainties make it difficult to evaluate the models capability of simulating the actual level of the groundwater table.

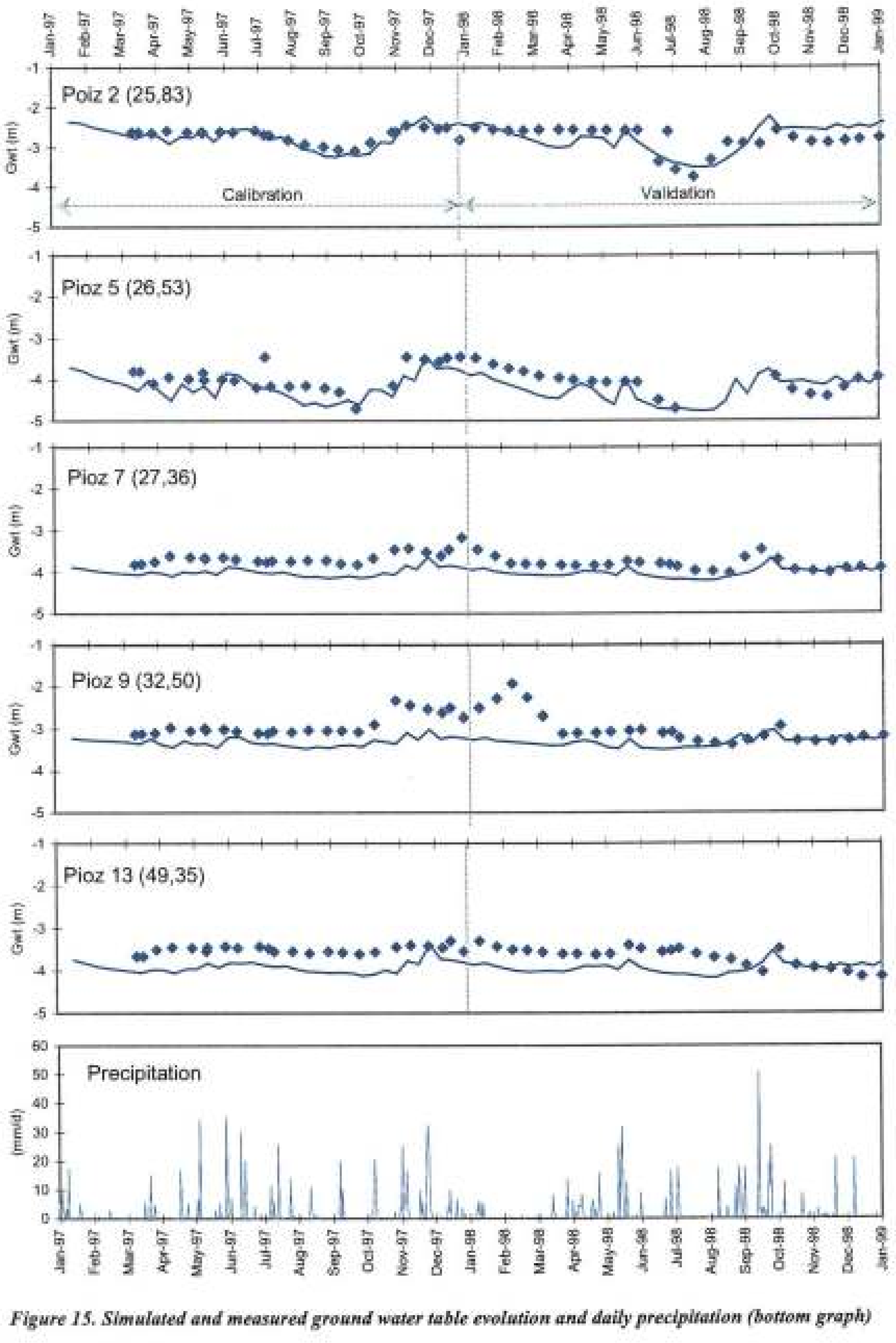

Figure 15. Simulated and measured ground water table evolution and daily precipitation (bottom graph)

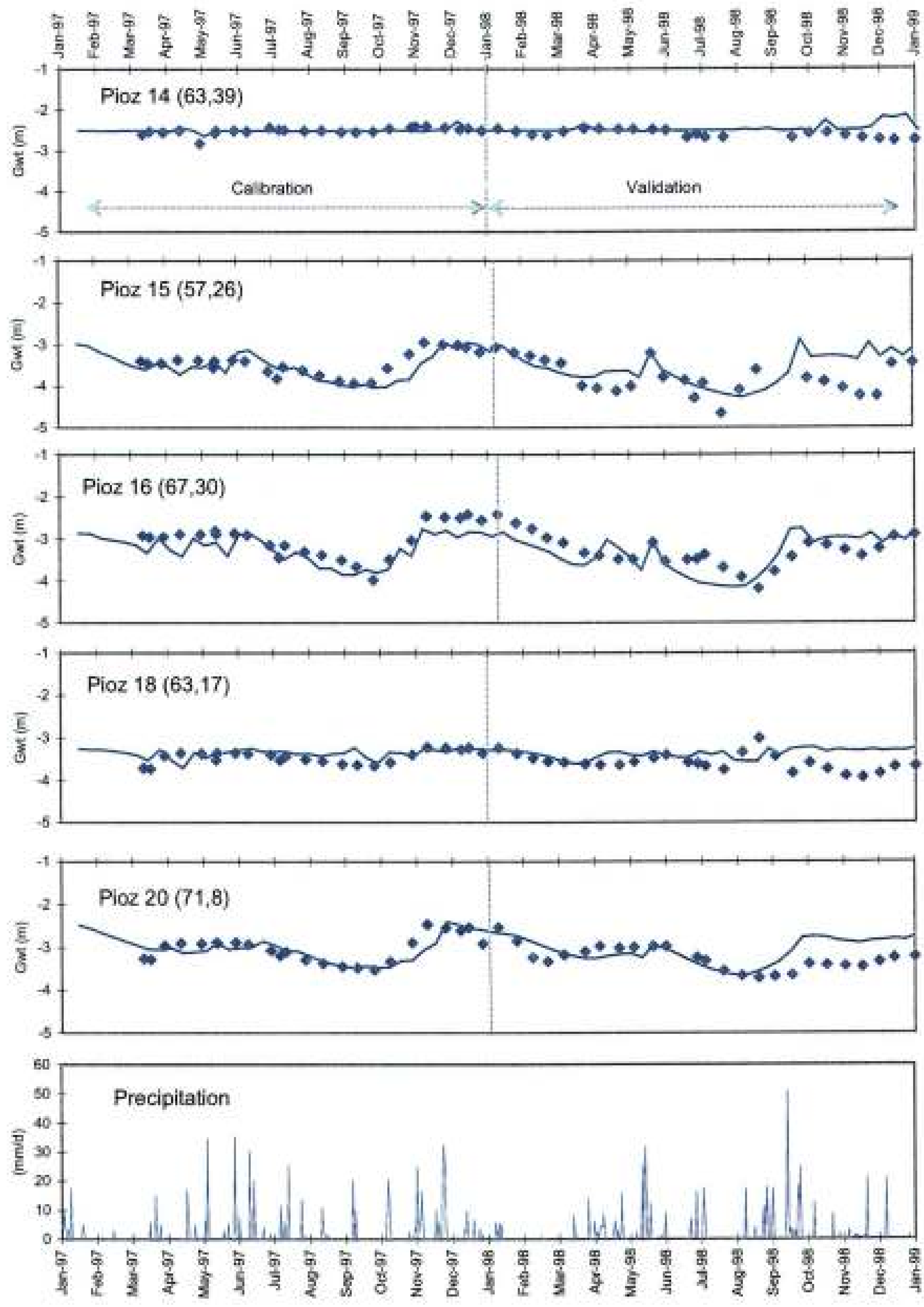

Figure 16. Simulated and measured groundwater table evolution and daily precipitation(bottom graph).

3.2.3. Water balance

The simulated water balance for the study area is given in Table 7, while specific water balances for the saturated zone and channel network are given in Table 8. Positive values indicate a water inflow to the various systems, whereas negative values indicate a water outflow.

Table 7. Annual water balance for the study area (mm/year)

	1993	1994	1995	1996	1997	1998	Average
Precipitation	675	594	831	774	533	665	679
Irrigation	197	225	84	190	305	345	224
Flow across model boundaries	182	180	180	173	183	182	180
Evapotranspiration	-603	-594	-637	-462	-511	-705	-585
Net channel outflow	-420	-437	-459	-653	-538	-507	-502
Change in storage	31	-32	-1	22	-28	-20	-5

Table 8. Water balance for the saturated zone and the channel network (mm/year)

	1993	1994	1995	1996	1997	1998	Average
			Water balance for the saturated zone				
Percolation	180	123	216	423	196	172	218
Flow across model boundaries	181	180	180	174	183	182	180
Net exchange flow to channel	-323	-339	-388	-570	-406	-364	-398
Change in storage	38	-36	7	26	-27	-10	0
			Water balance for the channel network				
Net inflow from aquifer	323	339	388	570	406	364	398
Net inflow from overland flow	1	1	9	6	1	10	5
Irrigation tail loss	96	97	62	77	131	133	99
Net channel outflow	-420	-437	-459	-653	-538	-507	-502

The water balance was characterised by two important water inflow: Water fluxes across boundaries and application of irrigation water.

The model boundaries towards the higher located Po di Goro and the Adriatic Sea was characterised by a strong hydraulic gradient and large amount of water entering the system across these boundaries (see Figure 17.). During the simulation period from 1993-1998 the average groundwater recharge was found to be 401 mm /year, and 45 % of this recharge derived from water fluxes across model boundaries.

The estimated mean irrigation input was 224 mm/year. A rather large amount accounting for approximately 20 % of the total water input to the system (See Table 7). As previously described, the irrigation water is taken directly from the higher located Po di Goro and Canale Bianche and applied to the fields by channel release from a dense network of irrigation ditches. This irrigation system requires that a minimum water level is maintained in the main channel in order to supply all the smaller distribution channels. A certain tail loss, in terms of irrigation water remaining in the channel system, is therefore unavoidable. The estimated tail loss accounts for 44 % of the total irrigation input. It should however be noted that this estimate is based upon a simple description of the irrigation system (see discussion in section 3.2.1. and 3.2.2.). Consequently, the estimated tail loss as well as the total irrigation input is associated with some uncertainties.

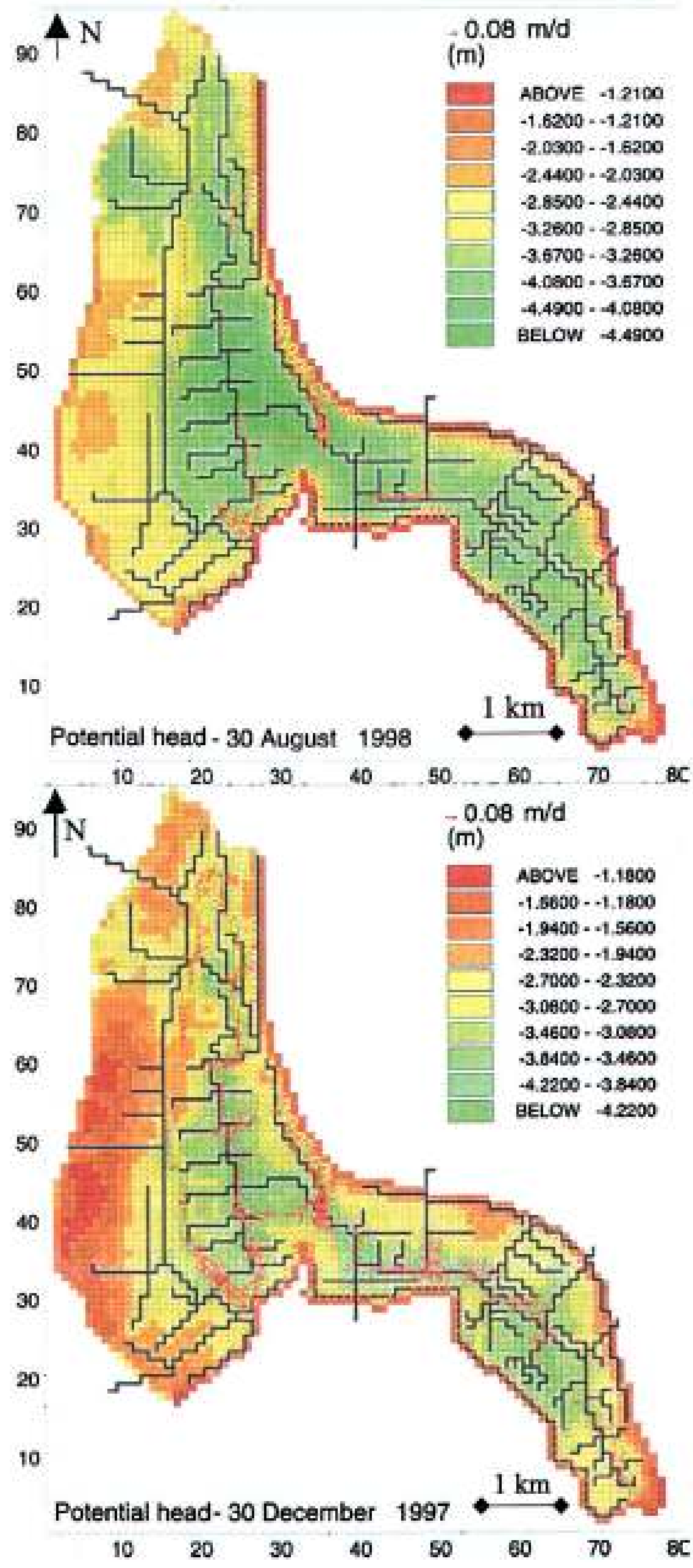

Figure 17. Simulated potential head and groundwater flow. Colours indicate the head elevation whereas the direction and size of the arrows indicates the flow direction and flow rate respectively.

3.3. Summery and concluding remarks

The model performance was considered satisfactory for describing the overall hydrological processes and the integrated hydrological response of the catchment as a whole, although the results did also reveal some model limitations.

In particular the limited data availability on the irrigation input and spatial variability of the hydraulic properties hindered the model in capturing all internal variation occurring within the catchment. Moreover, the applied description of the irrigation system was found to be sufficient for the meteorological variability found within the calibration period from 1993- 1997, but apparently it was not applicable for the unusual meteorological condition characterising the summer month of the validation year 1998. It should also be noticed that the measurements of the groundwater table had some inaccuracy.

The established water balance was characterised by two important water inflows: Water fluxes across boundaries and application of irrigation water.

The model boundaries towards the higher located Po di Goro and the Adriatic Sea was characterised by a strong hydraulic gradient and large amount of water entering the system across these boundaries. During the simulation period from 1993-1998 the estimated groundwater recharge was 401 mm /year, and 45 % of this recharge derived from water fluxes across model boundaries. The study area was furthermore characterised by a large irrigation demand and the estimated mean irrigation input was 224 mm/year, accounting for 20 % of the total water input to the system.

4. Modelling of nitrogen dynamics

The nitrogen modelling using DAISY aimed at quantifying the nitrogen loading deriving from the arable land of the Bonello catchment, with specific emphasis on the various root zone processes affecting the nitrogen leaching. Moreover, it was analysed which environmental improvement could be obtained by implementing alternative management practices.

4.1. Modelling of nitrogen dynamics – field scale

The first modelling exercise involved the upscaling of the one-dimensional model to represent condition at the field scale. Hence, the model capabilities of describing the root zone processes were carefully evaluated through a model calibration/validation. Calibration sites of 1 ha were chosen in Maize fields, the major crop type in the area (position of the four calibration sites is indicated in Figure 9.). However, during the project the crop rotation occurred at some of the sites and calibration data for alfalfa and sugar beet also became available.

DAISY has mainly been applied under climatic conditions typist of Northern and Central Europe. Thus, an interesting aspect of this project has been a test of the model applicability to Southern European conditions in terms of different climate other cultivar types, etc.

4.1.1. Model set-up and calibration

The field was interpreted as an equivalent soil column. Thus, the spatial variability within the field scale was aggregated and accounted for in effective parameter values, assessed from the measured data. The model set-up of the various sub-modules within DAISY is described in the following, along with a discussion of the selected calibration parameters.

4.1.1.1. Hydrological module

The hydrological module of DAISY considers evapotranspiration, infiltration, interception snow melting and accumulation as well as water uptake by plants. The crop growth and nitrogen transformation processes are closely related with the soil water content in the root zone, and a proper description of the soil water dynamics is therefore of outmost importance. Essential for the set-up of the hydrological module is:

- An indication of the lower boundary condition,
- A description of soil physical properties.

The area is characterised by a shallow water table ranging from approximately 1 to 3 meters below ground surface, and by strong soil capillary forces 1 (see Figure 8). Hence, the groundwater table has a great impact on soil water dynamics in the root zone and is an important input parameter for the hydrological module. Description of the groundwater dynamics was obtained from the hydrological model MIKESHE, which was setup for the entire study area (see section 3). Thus, time series of ground water tables retrieved from MIKESHE simulations were applied as lower boundary conditions for the hydrological module in DAISY.

The soil physical properties were assessed from field measurements (see section 2.3.). Thus, the soil profiles consisted of three to four horizons each characterised by physical properties as given in Table 4 and Figure 8. The saturated hydraulic conductivity (Ks) was used as calibration parameter. During the calibration period this parameter was changed in order to fit the hydraulic conductivity curve. Initial estimates of Ks were obtained using pedotransfer functions. These prior estimates indicated that the upper loamy soils has rather low permeability ranging from $10E^{-6}$ to $10E^{-7}$ (Table 9). However, the topsoil (0-45 cm) is frequently disturbed by tillage operations, which is expected to increase the permeability. In these types of loamy soil, cracking and transport through macro pores are common phenomena during the dry summer period. An example of soil crack found at one of the calibration sites is given in Figure 18. The DAISY model considers only Darcy flow in the soil matrix. Thus, the quick water transport occurring in macro pores can only crudely be simulated by increasing the soil permeability. The permeability in the upper soil layer was therefore increased considerably during the calibration period (Table 9.).

Table 9 Initial and final calibrated values for saturated hydraulic conductivity.

Depth (cm)	*Initial Ks (m/s)*	*Final calibrated Ks (m/s)*	*Depth (cm)*	*Initial Ks (m/s)*	*Final calibrated Ks (m/s)*
	PR-site			**BR-site**	
0 – 50	*5.0E-07*	*9.0E-05*	*0 - 50*	*3.6E-07*	*3.6E-05*
50 - 80	*8.5E-07*	*9.0E-07*	*50 - 80*	*8.9E-07*	*8.2E-07*
80 – 100	*1.0E-04*	*1.0E-04*	*80 - 100*	*1.8E-04*	*1.8E-04*
	TS-site			**BV-site**	
0 – 40	*5.9E-07*	*9.9E-05*	*0 - 40*	*5.1E-06*	*1.0E-04*
40 - 80	*3.0E-07*	*3.0E-07*	*40 - 80*	*1.2E-06*	*5.0E-06*
80 – 100	*1.1E-04*	*1.1E-04*	*80- 120*	*5.6E-06*	*5.3E-06*
			120 -	*1.0E-04*	*1.0E-04*

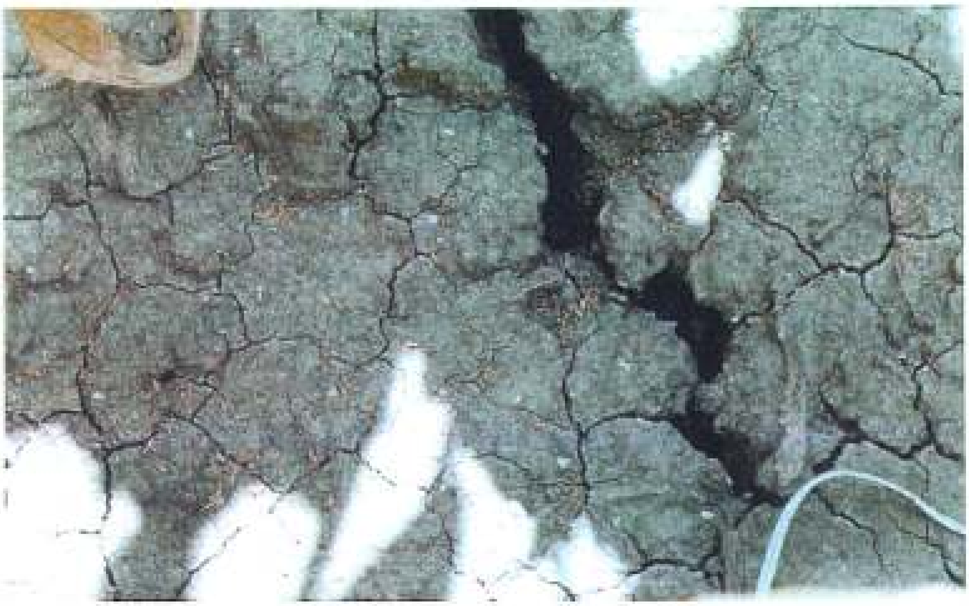

Figure 18. Soil cracking observed at the BV site June 1997.

4.1.1.2. Thermal module

The thermal properties of the soil were estimated according to de Vries (1963) and included in the model setup. The thermal module has not been subject to any calibration.

4.1.1.3. Crop module

The simulation of the crop dry matter production and nitrogen uptake for the various crop types is considered within the crop module of DAISY. During the calibration procedure simulated and measured data was fitted towards each other by changing: "Maintenance coefficient", "temperature sum at emergence" as well as the "maximum NO_3 uptake". The initial and final calibrated values of the calibration parameters are illustrated in Table 10.

The model sensitivity towards the selected calibration parameters was also analysed through two-sensitivity analysis:

An initial sensitivity analysis measuring the model sensitivity in terms of a sensitivity coefficient SC defined as:

$$SC = \frac{|Y(x_i + \xi x_i) - Y(x_i)| + |Y(x_i - \xi x_i) - Y(x_i)|}{Y(x_i)} \qquad \text{Eq. 14}$$

Y(x) is the model output for a given parameter x. In this analysis values for x_i were default given by Hansen (1993) and ξ were set to 0.25.

A final sensitivity analysis measuring model sensitivity in terms of "% change" defined as:

$$\%\ \text{Change} = \frac{(Y(x_f) - Y(x_i)) \cdot 100\%}{Y(x_i)} \qquad \text{Eq. 15}$$

Y(x) is the model ouput for a given input parameter x. x_i is the initial input parameter given by Hansen 1993 and x_f the final calibrated parameter. Hence, the final sensitivity analysis (in terms of "% change") illustrates how the actual range of variation - introduced during the calibration period- effects the model output. Interactions between the various input parameters were taken into account by simulating a "matrix-combination" allowing each input parameter to be kept, either at its initial values given by Hansen et. al. (1993), or at its final calibrated value obtained through calibration.

Both sensitivity analyses were carried out at the PR-Site and the sensitivity was evaluated during 1998. The model sensitivity was measured on four different model outputs: Net mineralisation, denitrification, plant uptake and nitrogen leaching. The obtained results are illustrated in Table 11 and Figure 19.

The overall nitrogen balance was not very sensitive to the calibration parameters (maintenance respiration coefficient, temperature sum and max NO_3-uptake). These parameters do influence the crop development during the growing season, but they do not have a great impact on the final crop growth and nitrogen uptake (See Table 11 and Figure 19).

Data concerning the crop growth and nitrogen uptake of alfalfa was not available from the field site. Thus, the calibration involved a more "coarsely procedure" using the available soil data as an indication of the crop nitrogen uptake and the statistical data on harvest yield as an indication of the crop growth.

Table 10 Initial and final values of the calibration parameters within the crop module.

	Sugar Beet		**Maize**	
	Initial value	Calibrated value	Initial value	Calibrated value
Top maintenance respiration Coefficient in respiration period II	0.03	0.02	0.006	0.007
Temperature sum at Emergence	425	405	500	450*)
Max. NO3-uptake per unit root length (mg/m/day)			0.06	0.05

**) In 1997 the measured crop growth and nitrogen uptake at the BV site were lower than the other sites. The BV site was - compared to the other sites - not very well "maintained" during the growing season 1997, as large camomile plants were present at the field until late June. It is likely that competition from these camomile plants were causing the slower crop growth and nitrogen uptake. An increased temperature sum at emergence (550) was therefore applied at the BV site in 1997 for taken this process into account.*

Table 11. Sensitivity coefficient for selected model parameter of the PR-Site during 1998. The sensitivity coefficient was calculated according to Eq. 14.

	Top maintenance respiration Coefficient	Temperature sum at Emergence (° C)	Max. NO_3-uptake per unit root length (mg/m/day)
Net mineralisation	0.08	0.11	0.03
Denitrification	0.06	0.03	0.00
Uptake	0.02	0.05	0.06
Leaching	0.00	-0.08	-0.05
Crop Growth	0.17	0.04	0.01

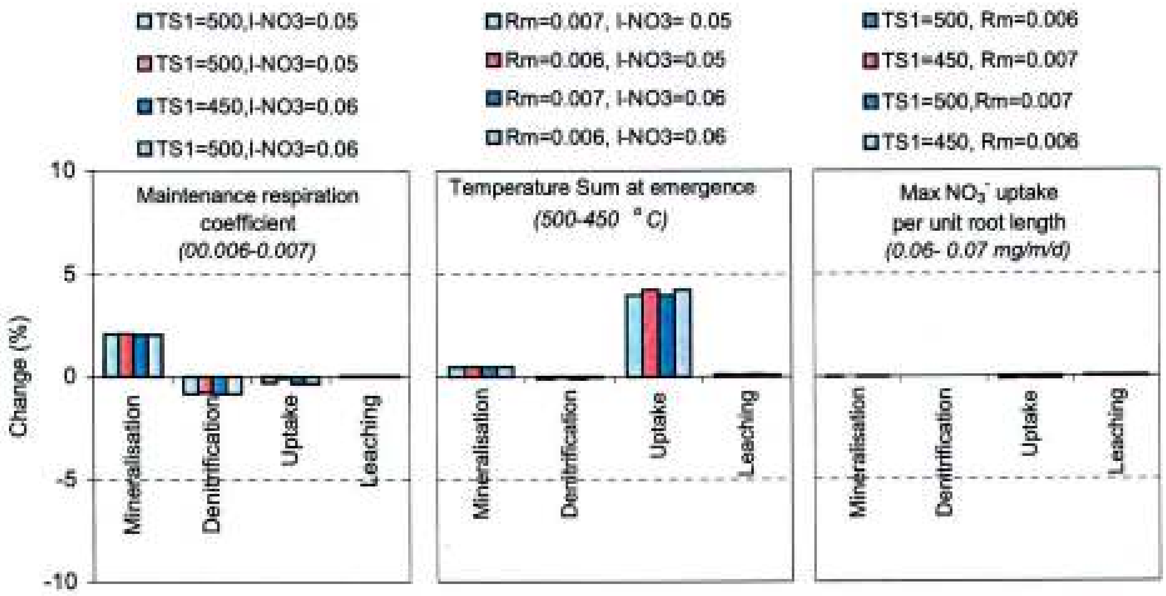

Figure 19. The percentage change (%Change) of various model parameters. The "Change %" illustrates how the actual range of variation of the three calibration parameter affect the model output at PR-site in 1998. The investigated input parameters were (from left): Maintenance respiration coefficient (Rm), Temperature sum (TS1) and Max NO_3 uptake per unit root length (I-NO_3). The number in brackets indicates the actual range of variation introduced during the calibration period, and the labels describe the various "matrix-combinations".

4.1.1.4. Management module

The management module includes information about all induced management operations such as irrigation, fertilisation, sowing, harvest and other tillage practices.

The irrigation water was applied to the fields by a dense network of irrigation ditches maintaining suitable soil humidity for agricultural purposes. Consequently, irrigation water was not applied directly in the model setup but explicit included via the applied ground water table.

Tillage and fertilisation practices were obtained by the farmers directly and included in the management module. A description of the obtained information is given appendix I. Urea was considered as ammonium fertiliser, as the hydrolyse (Eq. 16) was assumed to be immediate without significant time lag.

$$CO(NH_2)_2 \quad + \quad 3H_2O \rightarrow 2NH_4^+ \quad + \quad CO_2 \quad + \quad 2OH^- \qquad \textit{Eq. 16}$$

Initial simulation indicated that a considerable nitrogen loss occurred in connection with the fertiliser application of Urea; and that this loss could not be explained by plant uptake and leaching alone. Thus, the nitrogen loss could very well be associated with ammonia volatilisation. This process is particular important in farming system involving animals, but significant volatilisation losses have also been observed in connection with urea application (Ryden 1984, Demead 1983, Rachphal-Singh and Nye 1988, Hargove 1988, Clay et. al., 1990, Gezgin and Bayrakl 1995 and Fox et. al. 1996).

The ammonia volatilisation was not included in the DAISY model. Thus, equations deriving from the EPIC model (Williams et. al., 1984) was adopted for estimating this process. The EPIC model uses a first-order kinetics to describe the volatilisation where the rate coefficient depends on soil temperature, cation exchange capacity and incorporation depth. Detailed descriptions of the calculations are given in Appendix II, whereas the estimated volatilisation losses for the various applications are given in Table 12 The calculations indicate that volatilisation occurred up to four to five days after fertiliser application. However, it was assumed that volatilisation occurred right after application and the volatilisation loss was included by reducing the fertiliser input according to the estimated loss.

In accordance with the results of Ceotto and Donatelli (1997) the harvest index of maize was set to 0.55, and harvested part of the primary and secondary yield was set to 90%. After harvest of sugar beet the plant residue was left at the field and the harvested part of the primary and secondary yield was therefore set to 90 % and 5% respectively.

Table 12. Fertiliser application and estimated volatilisation loss at the calibration sites

Application date	Fertiliser type	Application rate (kg N/ha)	NH4 conc. (% of applied N)	Estimated Volatilisation loss (% of applied N)
		Site PR		
20 April 1996	Urea	94	100	20
20 June 1996	Urea	94	100	25
6 March 1997	N-P-K	48	100	2
23 May 1997	Urea	71	100	17
2 April 1998	N-P	72	100	1
18 April 1998	N-P-K	8	100	0
10 May 1998	Urea	118	100	19
		Site BR		
14 April 1996	N-P	54	100	1
20 May 1996	Urea	150	100	23
31 March 1997	Polline	24	75	0
8 June 1997	Urea	188	100	24
5 April 1998	N-P	90	100	2
26 June 1998	Urea	141	100	20
		Site TS		
15 April 1996	N-P	63	100	2
31 May 1996	Urea	150	100	14
14 April 1997	N-P	54	100	1
8 June 1997	Urea	141	100	22
		Site BV		
5 April 1996	Urea	141	100	20
7 June 1996	Urea	118	100	13
6 April 1997	Urea	94	100	17
11 June 1997	Urea	141	100	33

4.1.1.5. Soil nitrogen module

The soil nitrogen module within DAISY considers the transformation and transport process of nitrogen including net mineralisation, nitrification, denitrification, nitrogen uptake by plant as well as nitrogen leaching.

An important initialisation parameter within this module is the mineralisation potential of the soil, which in DAISY is described by soil organic C concentration as well as C/N ratios. These data were determined experimentally (see Table 3) and included in the model setup. Initial concentration of NH_4 and NO_3 were not available. These concentrations mainly influence the simulation results during the first year, and the lack of data has therefore been overcomed by running the model through a three years "spin up" period prior to the calibration period.

The atmospheric deposition of Nitrogen is also taken into account by this module. The wet deposition of Nitrogen was estimated by the NO3-N and NH4-N concentration in the precipitation. Based on the results of Mosello (1993), values of 1.7 ppm NO3-N and 3.2 ppm NH4-N were taken as representative for the study area. The dry deposition was estimated assuming a yearly deposition rate of 0.006 kg N/ha/y (Mosello, 1991).

The leaching depth at BV, PR and TS-site was set to 1.5 meter. A depth, which – according to the crop parameter given by Hansen et. al. (1993) – was the maximum rooting depth for the crop types present at the calibration sites. The BR-site had a high groundwater table situated above this rooting depth of 1.5 meter, and the leaching depth was therefore situated just above the ground water table (1.1 meter).

In the upper soil layers the measured ammonium concentration increased substantially as a consequence of fertiliser application. High concentration was also found in the deeper soil layers, indicating that a downward transport of ammonium occurred during the fertiliser period (See Figure *21*).

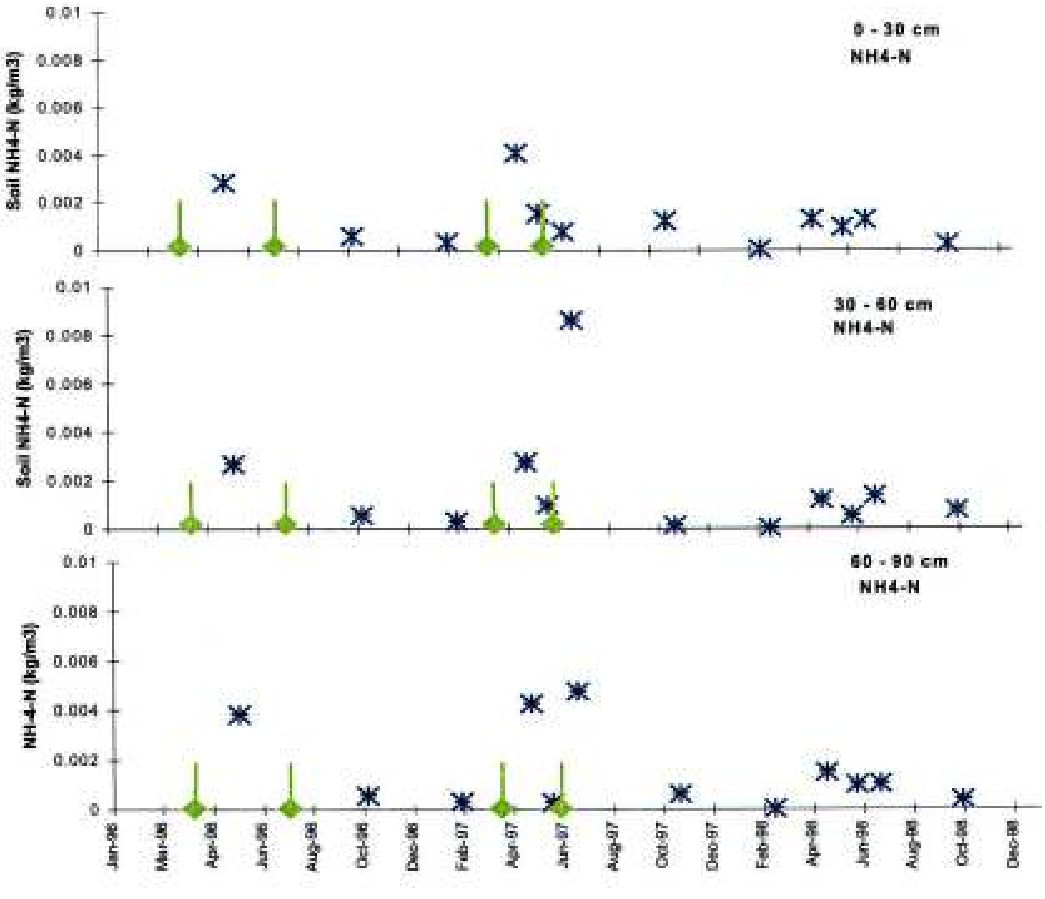

Figure 21. Measured soil NH4-N concentration at the BV site. Grey lines indicate the time of fertiliser application

The observed downward transport was not recognised in the initial simulation at all. In general a downward transport of ammonium would not be expected in such a clayey soil, as the ion tend to be retained in the soil by soil adsorption (Rauschkolb & Hornsby, 1994). However, as part of the transport apparently occurs through macro pores (see earlier discussion) the influence of this process can be minimised considerably. Moreover, the urea can - because of its zero net charges - also move readily into the soil profile before undergoing hydrolisation (Swensen and Singh 1997). A process which it not taken into account in the model setup as the urea is applied directly to the fields as ammonium (See section 4.1.1.4.).

Nitrification, adsorption and dispersivity control the downward transport of ammonium. Thus, during the calibration process the adsorption and the dispersivity was changes in the uppermost horisont in order to enlarge the vertical ammonium transport. The initial nitrification rate did not allow any downward transport of ammonium, as all the applied ammonium was nitrified in the upper soil layer rapidly after application. Thus, also the nitrification rate was reduced allowing a slightly higher ammonium concentration in the soil, as well as a downward transport of ammonium in the soil.

Table 13. Sensitivity coefficient for selected model parameter of the PR-site during 1996-1998.

	Adsorption	Dispersivity	Nitrification
Net mineralisation	0.00	0.00	0.00
Denitrification	0.01	0.00	0.00
Uptake	0.01	0.00	0.00
Leaching	0.01	0.01	0.03
Nitrification	0.04	0.00	0.05

The model sensitivity towards these calibration parameters was analysed following the procedure already described in section 4.1.1.3. The initial sensitivity analysis (Table 13) indicated, that the leaching was not very sensitive toward the calibration parameters and a change within the 50% interval was not sufficient for increasing the downward ammonium transport. Thus, during the calibration process the ammonium mobility was increased considerably by changing both dispersivity and ammonium adsorption. As indicated in Table 14., the ammonium adsorption was practically neglected. A drastic change, which – however - was found to the only way that the model could simulate the downward transport of the applied ammonium occurring through the macro pores. The effect of the introduced changes was evaluated in an additional sensitivity analysis, evaluating how the actual range of variation of the selected calibration parameter influenced the nitrogen balance. Here the model sensitivity was measured in terms of "% change" calculated according to Eq. 15 (See section 4.1.1.3.)

Table 14. Initial and final calibrated values for nitrification rate, dispersivity, and adsorption

	Initial value	Final calibrated value
Nitrification rate (days^{-1})	5.0E^{-03}	3.0E^{-03}
Max. NH4 adsorption-planar sites(meq/g of clay)	0.426	0.001
Max. NH4 adsorption – edge sites(meq/g of clay)	0.022	0.002
Dispersivity (cm)	10	60

The changes introduced during the calibration period caused an increased leaching, but did not have a major impact on other sinks and sources for the nitrogen system (see Figure *22*). Decreasing the adsorption did however change the pathway of ammonium. Before the major part of the applied ammonium was nitrified, and the produced nitrate was either leached out or taken up by the plant. By increasing the ammonium mobility a larger part of the applied ammonium was leached out directly or taken up by the plants; whereas a much smaller part was transformed into nitrate by nitrification. Thus increasing the ammonium mobility caused a decreased nitrification (See Figure *22*). The plant uptake of ammonium has preference over the uptake of nitrate. Thus, although the total amount of nitrogen uptake was only slightly effected by the decreased adsorption, the fraction of nitrogen taken up as ammonium was increased considerably.

During the calibration period only the nitrification rate, dispersivity, as well as adsorption were modified. All other initialisation and process parameters describing the soil nitrogen system were kept at the default value given by Hansen et. al. (1993).

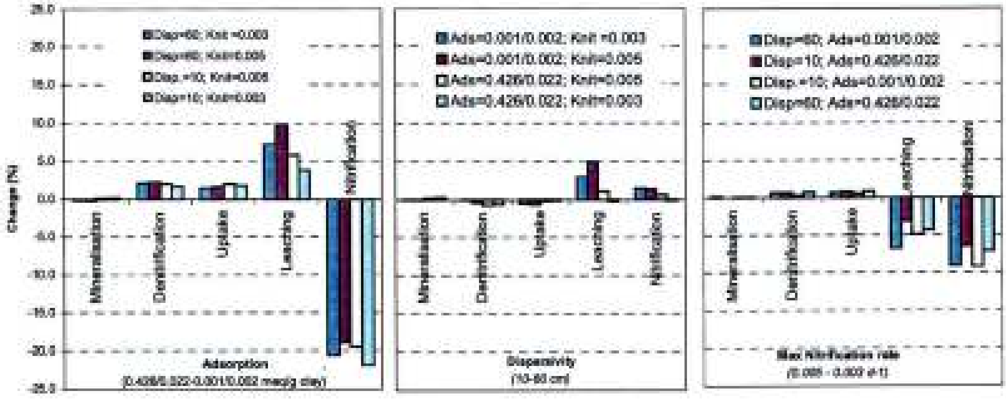

Figure 22. The percentage change (Change %) of various model parameter. The "Change %" illustrates how the actual range of variation of the three calibration parameter affect the model output at PR-site in 1996-1998. The investigated input parameters were (from left): Adsorption (Ads), Dispersivity (Disp) and Max. nitrification rate (Knit). The number in brackets indicates the actual range of variation introduced during the calibration period, and labels describe the various "matrix combination".

4.1.2. Results and discussion

The model performance of the various submodules within DAISY is discussed in the following section. The simulation results from both calibration and validation period are illustrated, whereas a discussion and indication of the various calibration parameters have been described in the previous section. Calibration was performed for the period 1996-1997, while validation was attempted using the data collected in 1998.

4.1.2.1. Hydrological module

The model simulation gave a good description of the soil water dynamics in the root zone(See Figure *23* - Figure 24). However, some inadequacies of the model were observed, particularly in representing the rapid flow occurring through macropores.

A large part of the observed discrepancies was found during the summer period, where the soil water content tends to be overestimated in the upper soil layers and underestimated or simulated correctly in the deeper soil layers. This effect is illustrated in the lower graphs of Figure *23* - Figure 24 showing the averaged soil water content from 0-90 cm. Here the simulated data generally correspond well with the measured data. Thus, despite of the observed disagreement within the 30-cm intervals the temporal variation of the soil water present in the root zone (0-90 cm) is generally simulated correctly.

The macro pore transport of water is a probable explanation of the found discrepancies. As mentioned before macropores is a common phenomenon during the summer period, and macro pores and soil cracks were observed in the field (Figure 18). The presence of macro pores do cause a rapid drainage after rain events, as large quantity of the water is rapidly transported downwards through the macro pores. During the calibration period the permeability in the upper soil layers were also increased in order to take this process into account (See Table 4). The temporal and spatial variation of macros pores do however make it difficult to simulate this "macro pores drainage" and a satisfactory result was not always obtained (See Figure *23* - Figure 24).

A simplified description of the spatial variation of the hydraulic parameters may also explain parts of the observed discrepancies. A critical point in the model setup of the hydrological module was that the hydrological parameters in terms of soil retention and hydraulic conductivity curve was measured at one location only. Indeed, aggregating all spatially variability into effective parameters, which have been assessed from one location only, is an obvious simplification. The concept of using effective parameter for field scale models has also been subjected to some discussion in the literature. Although the approach has been successfully applied in previous research studies dealing with leaching models (Djurhuus et. al. 1999; Svendsen et. al. 1995). It has also received some criticism. e.g. Smith and Diekkuger (1996) concluded that the effective soil hydraulic parameters were not adequate for modelling water flow in spatially variable fields.

Finally it should be noted that the precipitation data were not measured directly on site but retrieved from the nearest situated meteorological station situated some 8 kilometres from the study area. Thus, some of the observed discrepancies could presumably relate to uncertainties associated with the actual precipitation input.

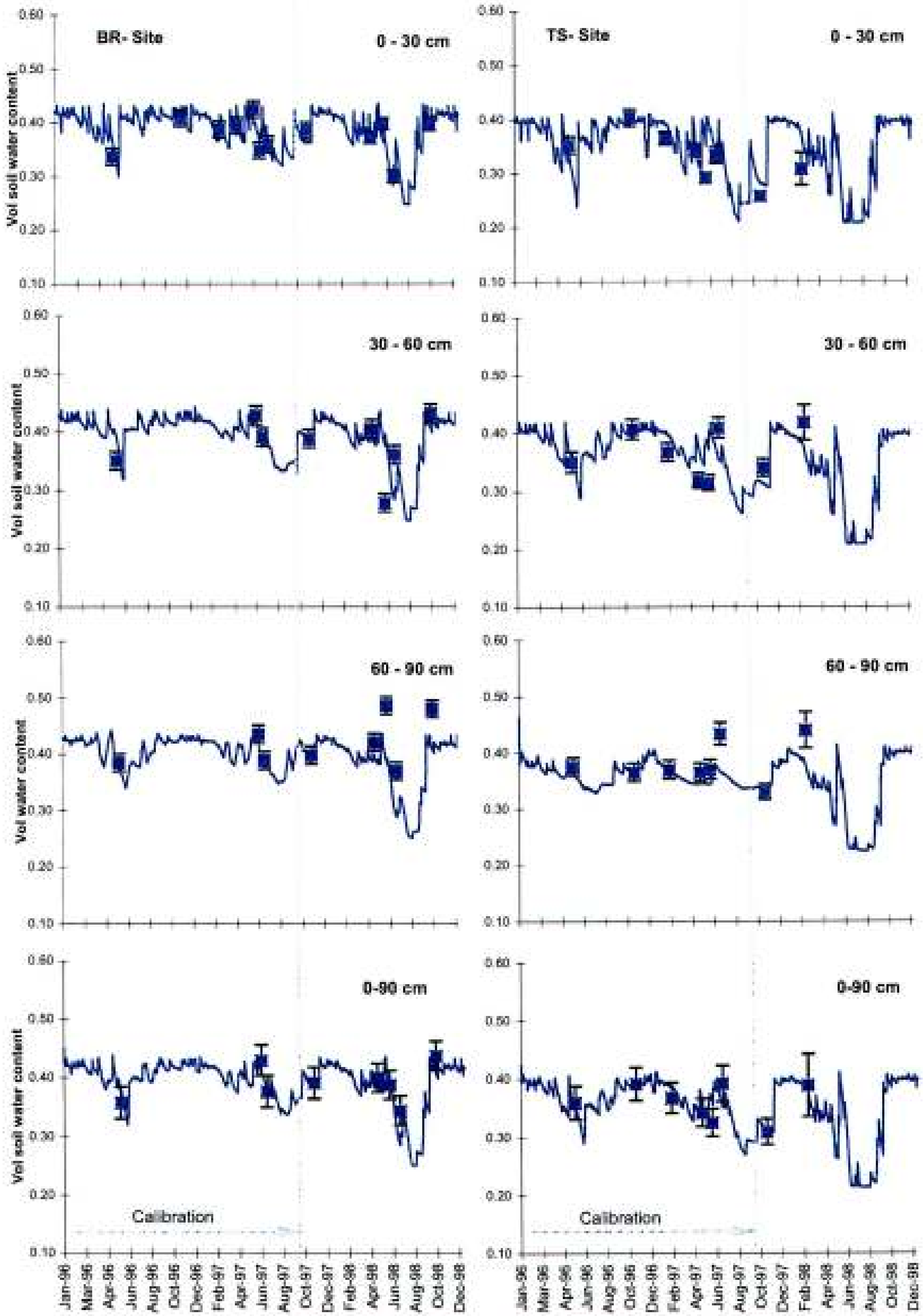

Figure 23. Simulated (solid line) and measured (points) soil water content at the BR and TS site. Error bars indicate one standard deviation. The concentration is averaged over the three 30 cm intervals in the three upper graphs, and over the whole 90 cm depth in the bottom graph

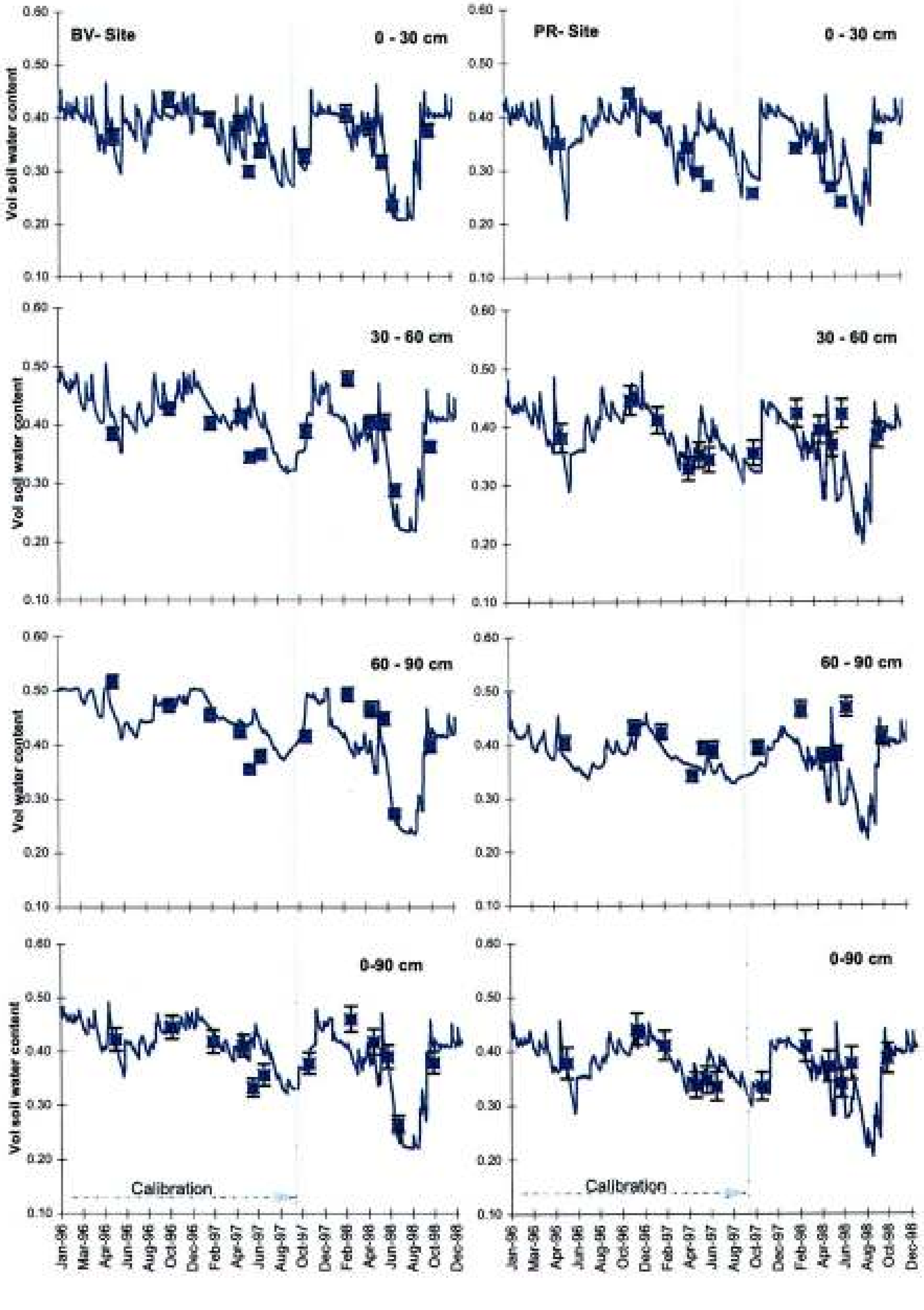

Figure 24. Simulated (solid line) and measured (points) soil water content at the BV and PR site. Error bars indicate one standard deviation. The concentration is averaged over the three 30 cm intervals in the three upper graphs, and over the whole 90 cm depth in the bottom graph

4.1.2.2. Thermal module

There is a very good correlation between the measured and simulated soil temperature, which indicates good model performance concerning soil temperature (See Figure 25.). This is purely based upon data measured in the upper 20 cm at one of the calibration sites (BR). Furthermore, it should be noted that the thermal module has not been subject to any calibration.

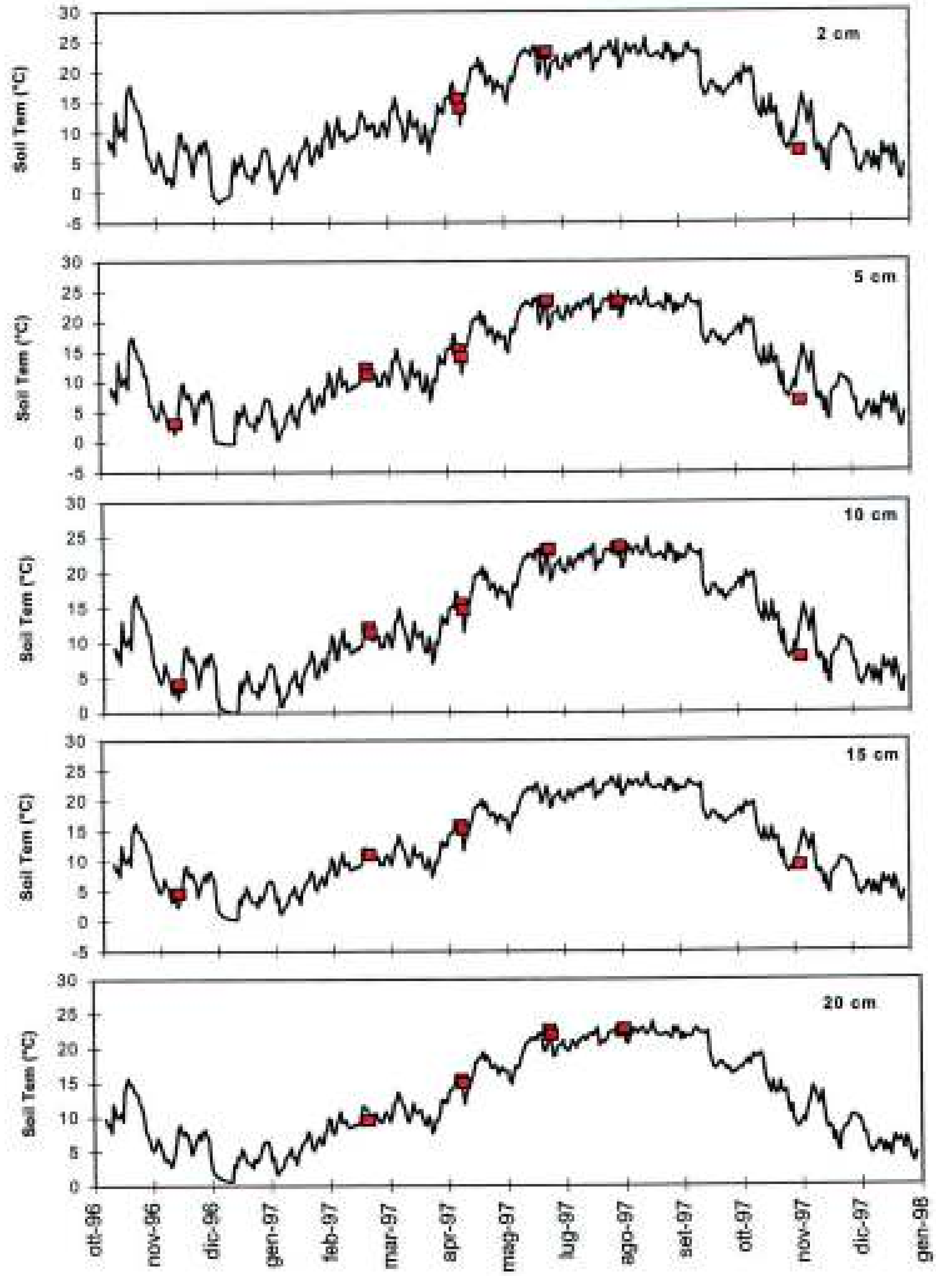

Figure 25. Simulated (solid line) and measured (points) soil temperature at the BR site.

4.1.2.3. Soil nitrogen dynamics

Model simulations were generally consistent with observed data, indicating a good model description of the nitrogen dynamic. However, the results also highlighted some model limitations, particularly in describing the transformation and transport processes of the applied NH_4-fertiliser occurring shortly after fertiliser application. Inadequacies which appear to be associated with the following two processes:

- Urea hydrolyse
- Preferential flow of ammonium and Urea nitrogen

Clear discrepancies were found in the upper soil layer during the summer and growing season (Figure 26A,E and Figure 27A,E). The soil nitrogen concentration was generally overestimated, which indicate an additional nitrogen loss from the system, which the model does not take into account. The simulation results indicated that the observed nitrogen loss could not be explained by plant uptake because the crop uptake was not underestimated (Figure 32 - Figure 34).

Most of the observed disagreement was observed in connection with application of the fertiliser urea. The urea was applied to the field as ammonium because the hydrolyse of urea (Eq. 17) was assumed to be immediate without time lag. Thus, all applied urea was immediately available as ammonium for crop uptake, nitrification and volatilisation. The urea hydrolyse and the following volatilisation are however strongly correlated with soil temperature and humidity (Hargrove 1988, Clay et. al.1990, McInnes et. al. 1986). In McInnes et. al. (1986) water deficit caused considerable delay of the urea hydrolyse and unhydrolysed urea were found in the soil two weeks after application. Also Gezgin (1995) found that the urea hydrolyse - because of water deficit - was delayed and started nearly 20 days after urea application. Also at the calibration sites residual parts of the applied urea pellets were found several days after application, indicating significant time lag of the urea hydrolyse (Leip, 2000). A time lag, which presumably was caused by water deficit, as also suggested by Leip (2000). Thus, parts of the observed nitrogen loss can be caused by a delayed urea hydrolyse. An effect which is not considered by the model, and which leaves parts of the applied nitrogen as unhydrolysed Urea.

A time lag of the urea hydrolyse will also increase the nitrogen leaching risk. Unhydrolysed urea is more exposed to leaching, as it - because its hydrophilic and uncharged molecule can move readily into the soil profile before undergoing hydrolyse (Swensen and Singh, 1997). A downward transport, which is further enhanced, as parts of the water transport apparently occurs through macro pores (See section 4.1.2.1.). A downward transport of ammonium was also observed at the calibration sites, as high ammonium concentrations were found in the deeper soil layers during the fertiliser period (Figure 21). During the calibration period the ammonium mobility was also increased in order to take this process into account. However, the higher ammonium concentrations, observed in the deepest soil layers were still not recognised in the model simulation. A discrepancy, which indicates that the downward transport of ammonium occurring through macro pores was underestimated by the model. Thus, parts of the observed nitrogen loss might be caused by underestimated preferential transport of urea and ammonium.

Finally the spatial variability of the soil hydraulic parameter may also explain some of the observed discrepancies. Djurhuus et. al. (1999) evaluated the impact of the spatial variability of the soil hydraulic parameters on model output from DAISY. Soil input parameters (texture, soil retention and conductivity curve) as well as calibration data (soil water content and nitrate concentration in the soil water) was measured at 57 point on two experimental plots in Denmark. Djurhuus et. al.(1999) concluded that the observed nitrate concentration in soil water was well described by the model when using geometric means as effective parameters for the hydraulic parameters. His result also illustrated, that the variation of the model output – caused by spatial variability of the hydraulic parameters - increased considerably during the summer period. As previously described the spatial variability of the hydraulic parameters were aggregated into effective parameters, which were assessed from one location only. Thus, some of the observed inadequacies may be associated with this simplified description of soil hydraulic parameters.

This problem of overestimated nitrogen content during the growing is also reported elsewhere. In Diekkruger et. al. (1995) the performance of various different nitrogen leaching models were evaluated. Many of the involved models had problems simulating the mineral nitrogen concentration during the growing season. Here the mineral nitrogen was overestimated during the growing season, although the nitrogen dynamic in general was reflected satisfactory (Franko et. al 1995; Kersebaum, 1995; Svendsen et. al. 1995, Vanclooster et. al. 1995, Whitmore, 1995). Similar tendency was also found in Hansen et. al. (1991) using the DAISY model for simulating the nitrogen dynamics. Here the soil nitrogen was overestimated during the growing season, whereas a good description of the soil nitrogen was found during the winter season.

It should be noted that also in this work there was a good correlation between measured and simulated data during the winter season. This indicates that the found discrepancies are confined within a short period of time. A time period which is of less importance for the nitrogen leaching, as the predominating part of the leaching occurs during the autumn/winter period where there is a high net precipitation. Thus, although the model results did reveal clear uncertainties in describing the transformation and transport of the applied nitrogen fertiliser the overall model description of the nitrogen dynamics were considered satisfactory for evaluating the long-term nitrogen leaching risk.

Finally two of the observed discrepancies requires some further explanation:

The nitrogen concentration in the upper soil layer at PR in July 1998 was remarkable high and clearly underestimated by the model (Figure 27 & Figure 22). Contemporary, the very low nitrogen concentration found in the deeper soil layers was clearly overestimated by the model. It is quite unusual to find such high level of nitrate in the upper soil layer so long time after fertiliser application. It is also difficult to give an adequate interpretation of the observed discrepancy. However, the urea application was followed by a dry period with no precipitation, and water deficit could therefore cause some delay the urea hydrolyse. At the same time the crop uptake of nitrogen was highly overestimated (see discussion in section 4.1.2.4.). Thus, a time lag of the Urea hydrolyse as well as a high overestimation of the nitrogen crop uptake can be possible explanations for the observed discrepancy. Finally, the obtained information concerning the exact dates of fertilisation, harvest and sowing may also be associated with some uncertainties.

Compared to the other sites the nitrogen dynamic was reflected less satisfactory at the BR site (Figure *26* & Figure 28). A fact, which presumably was caused by the different sampling strategy applied at this field. The nitrogen concentration was measured on a soil sample deriving from one soil core only, whereas a pooled soil samples from nine soil cores was used on the other sites. Hence, the soil samples here are less representative of average field condition, as local variation is not taken into account in the model.

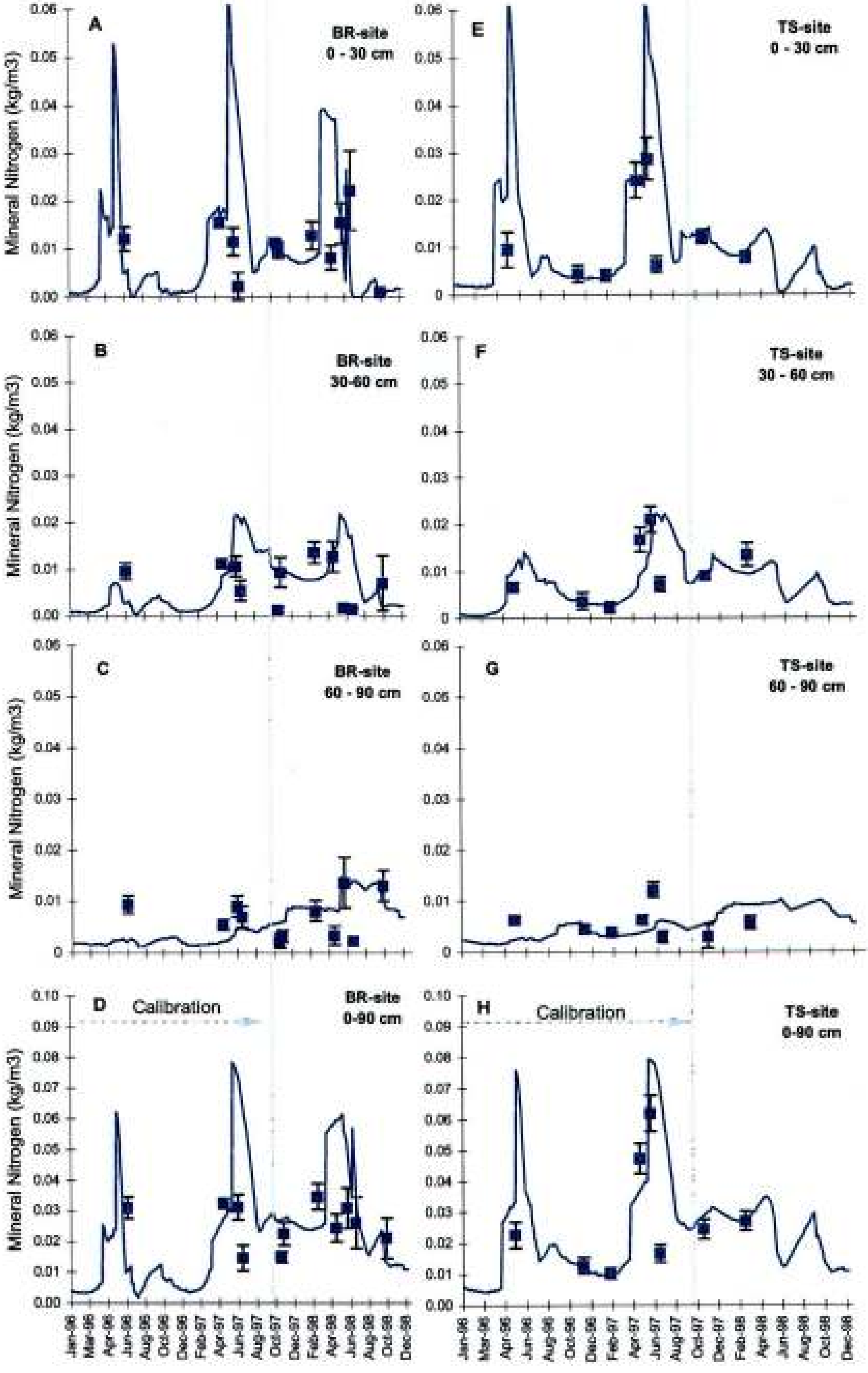

Figure 26. Simulated (solid line) and measured (points) soil mineral nitrogen at the BR and TS site. Crop BR: Maize 1996, maize 1997 and maize 1998 Crop TS: Maize 1996, maize 1997 and alfalfa 1998. The concentration is averaged over the three 30 cm intervals in the three upper graphs, and over the whole 90 cm depth in the bottom graph. Error bars indicate one standard deviation.

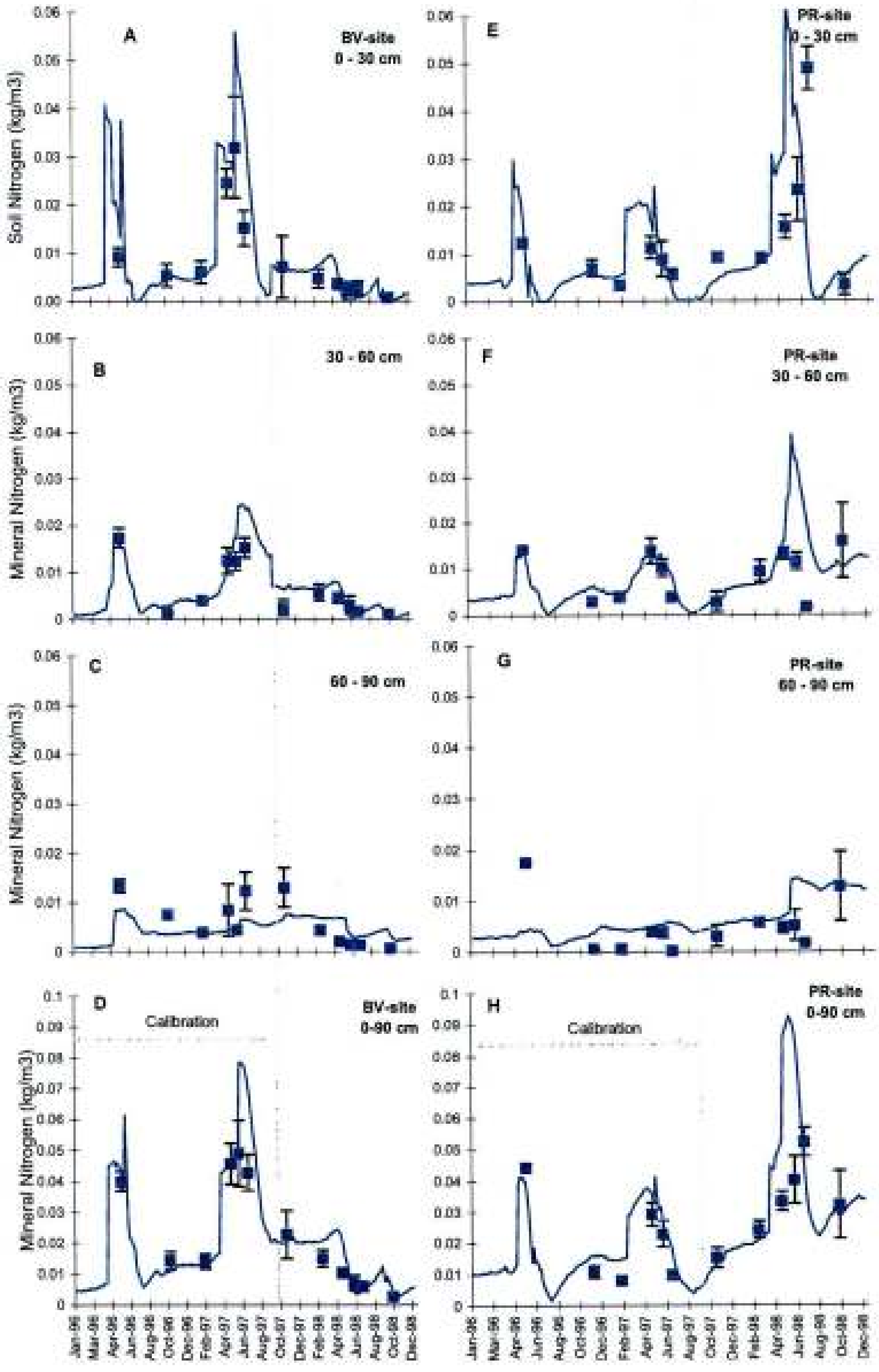

Figure 27. Simulated (solid line) and measured (points) soil mineral nitrogen at the BV and PR site. Crop BV: Maize 1996, maize 1997 and alfalfa 1998. Crop PR: Maize 1996, Sugar beet 1997 and Maize 1998. The concentration is averaged over the three 30 cm intervals in the three upper graphs, and over the whole 90 cm depth in the bottom graph. Error bars indicate one standard deviation.

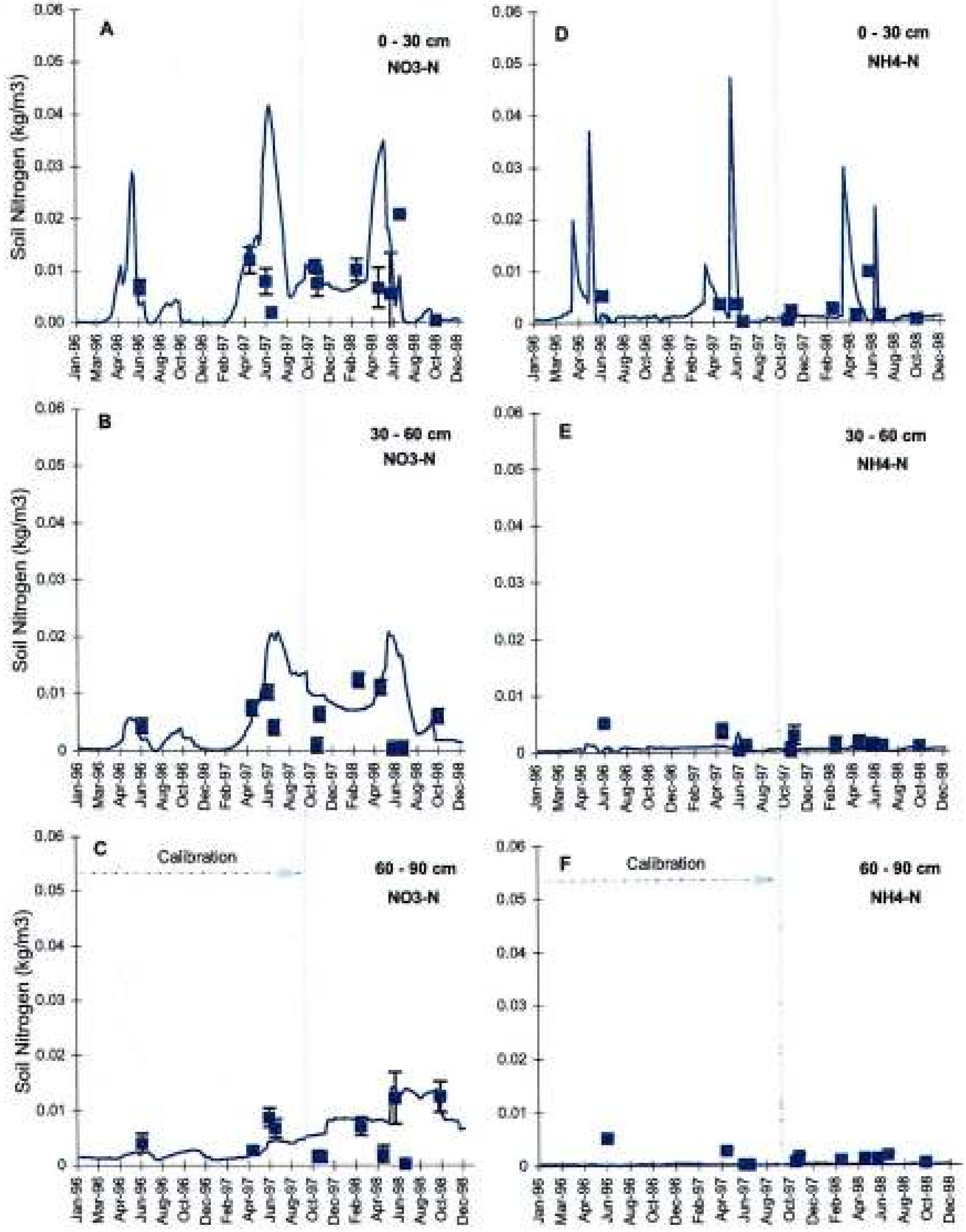

Figure 28. Simulated (solid line) and measured (points) NH_4-N and NO_3-N soil concentration at the BR site. Crop: Maize 1996, maize 1997 and maize 1998. The concentration is averaged over the three 30 cm intervals and error bars indicate one standard deviation

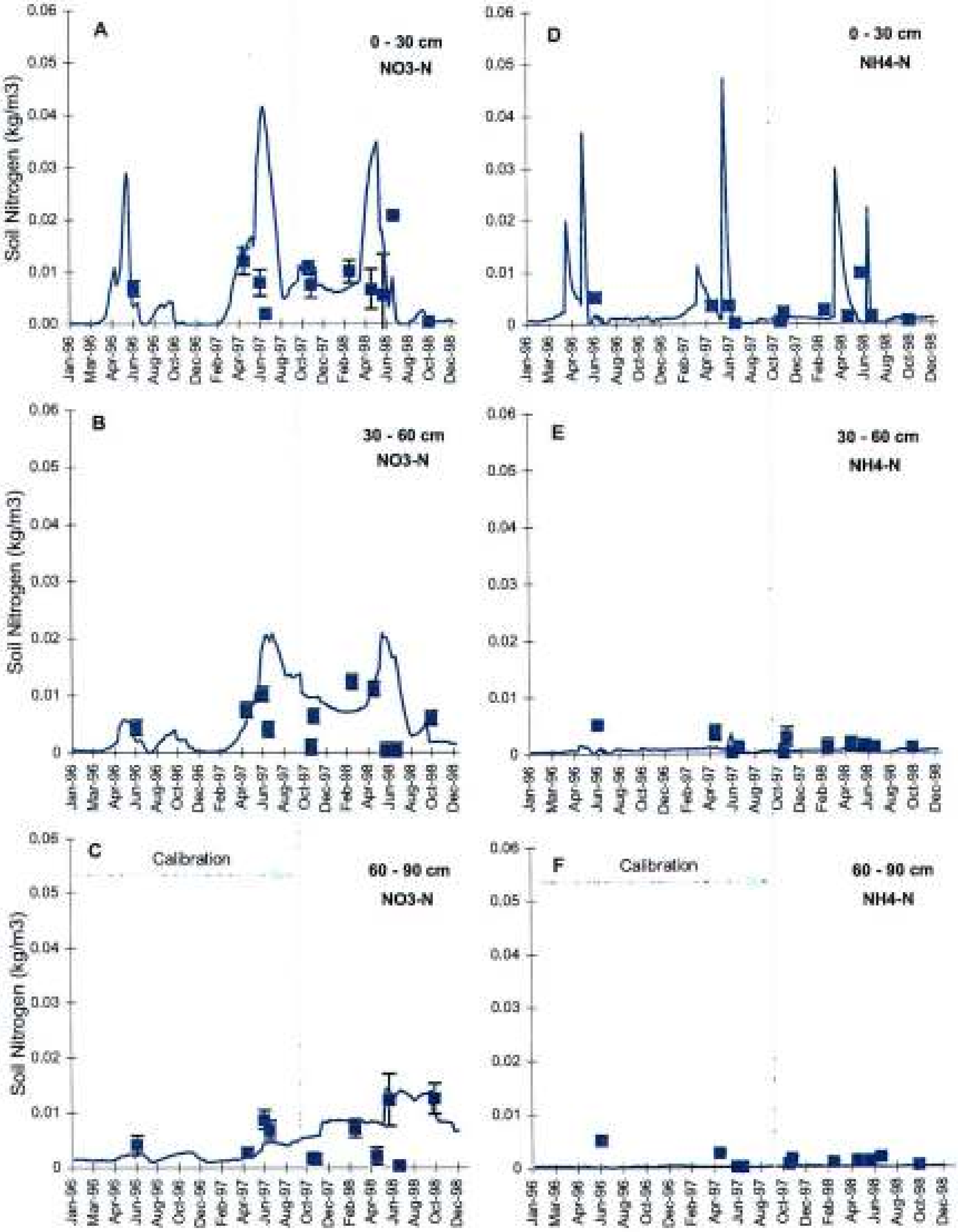

Figure 29. Simulated (solid line) and measured (points) NH_4-N and NO_3-N soil concentration at the TS site. Crop: Maize 1996, maize 1997 and alfalfa 1998. The concentration is averaged over the three 30 cm intervals and error bars indicate one standard deviation.

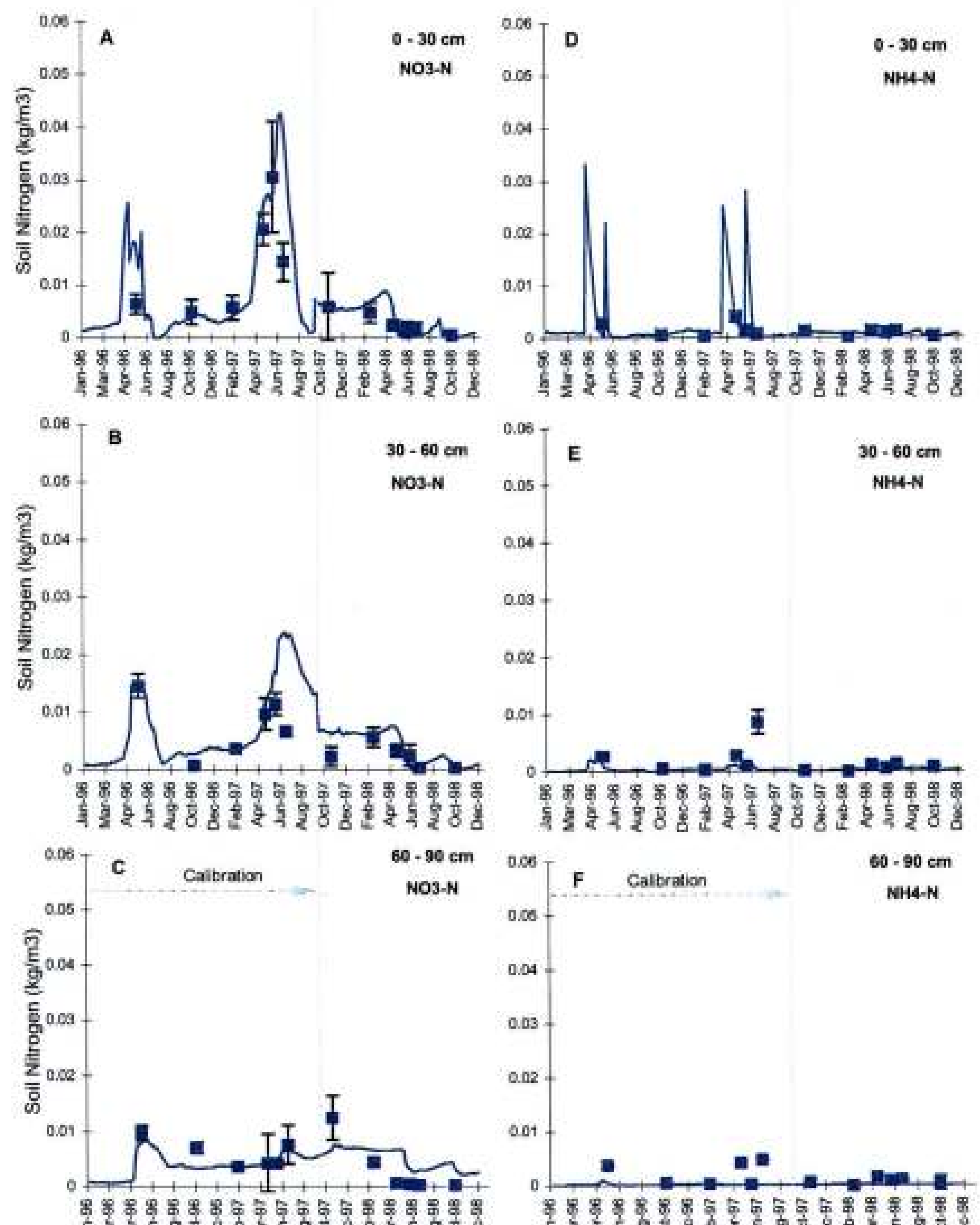

Figure 30. Simulated (solid line) and measured (points) NH_4-N and NO_3-N soil concentration at the BV site. Crop: Maize 1996, maize 1997 and alfalfa 1998. The concentration is averaged over the three 30 cm intervals and error bars indicate one standard deviation.

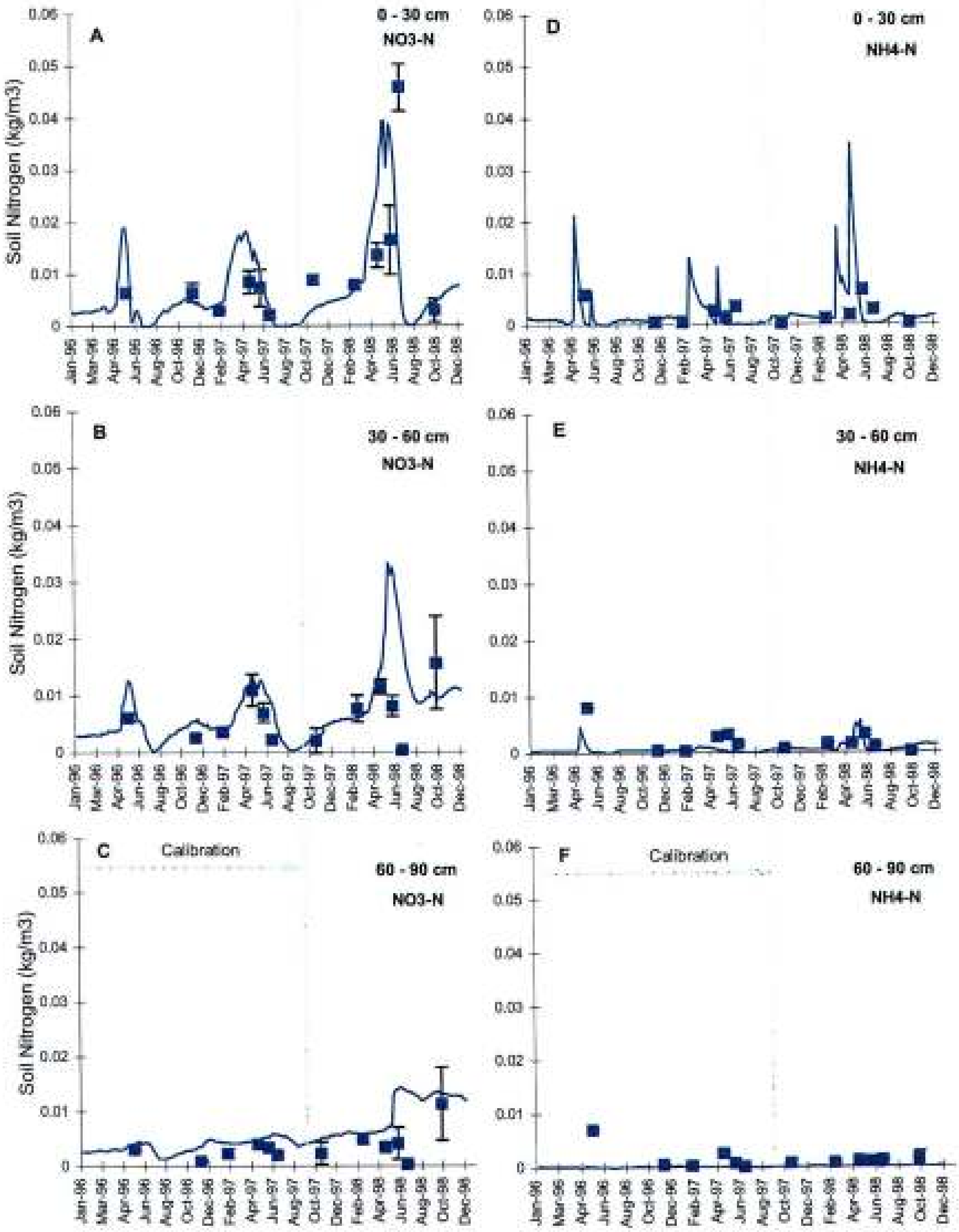

Figure 31. Simulated (solid line) and measured (points) NH_4-N and NO_3-N soil concentration at the PR site. Crop: Maize 1996, Sugar beet 1997 and Maize 1998. The concentration is averaged over the three 30 cm intervals and error bars indicate one standard deviation.

4.1.2.4. Crop module

The DAISY model gave a satisfactory description of the dry matter production and total nitrogen uptake of sugar beet (Figure 32), whereas the model results did reveal some discrepancies concerning the simulation of nitrogen uptake of Maize (Figure 33- Figure 34.)

At the BR site (Figure 33) the final crop growth and nitrogen uptake of maize are simulated correctly both in 1997 and 1998. Some discrepancies were however observed during the growing season, in terms of an underestimated crop growth in 1997 (Figure 33A.) and an overestimated nitrogen uptake in 1998 (Figure 33D.). The latter overestimation was also seen at the PR (Figure 32D.) and BV site (Figure 34B.), and it indicates that the nitrogen uptake is limited by stress-factors not taken into account in the model. Contemporary with the observed discrepancies the amount of nitrogen available for crop uptake is also highly overestimated (Figure 26 - Figure 27). Moreover the very low nitrogen concentration measured in the deeper soil layers in July 1998 indicated a nitrogen shortage at BR and PR site (Figure 26 - Figure 27). A nitrogen shortage, which could be a limiting factor for the crop growth. Consequently, the observed discrepancies are presumably caused by nitrogen deficit limiting the plant nitrogen uptake. A limiting factor which (due to the overestimated soil nitrogen) is not taken into account in the model. Unfortunately, samples deriving from the end of the growing season were unavailable from PR and BV site. Thus, it is not known whether the found discrepancies continues throughout the whole growing season, or are confined within a short time period like at BR site.

In fact, it should be noticed that calibration was based upon a rather limited data set. Crop data covering the whole growing season was only available from one site, whereas the collected crop data from the remaining sites only covers the first part of the crowing period. Moreover data describing the growth of sugar beet were only available for one year, and validation of this crop module was therefore not possibly.

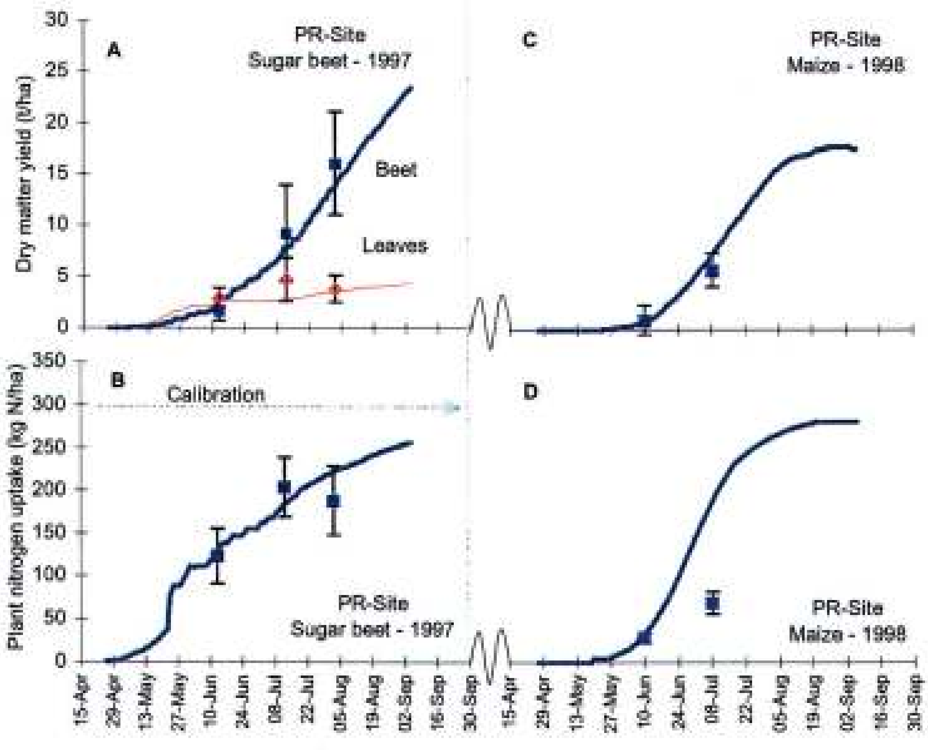

Figure 32. Simulated (solid line) and measured (point) crop growth and plant nitrogen uptake for sugar beet (A, B) and maize (C, D) at PR site. Error bars indicate one standard deviation

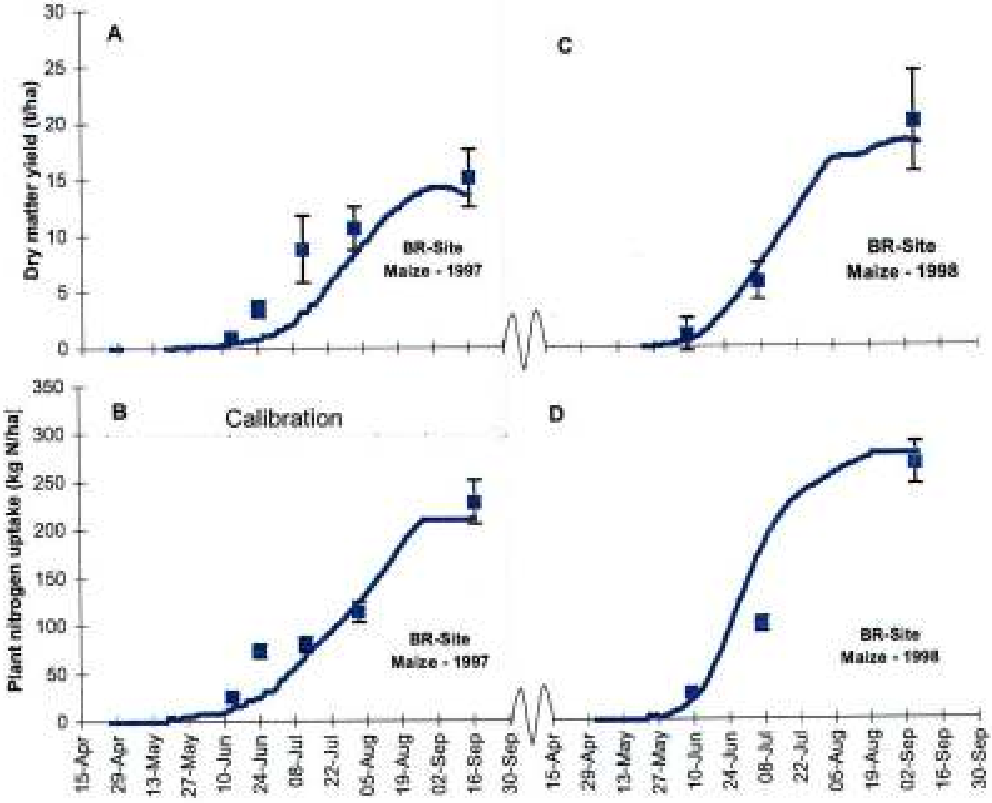

Figure 33.Simulated (solid line) and measured (point) crop growth and plant nitrogen uptake for maize at BR site. Error bars indicate one standard deviation

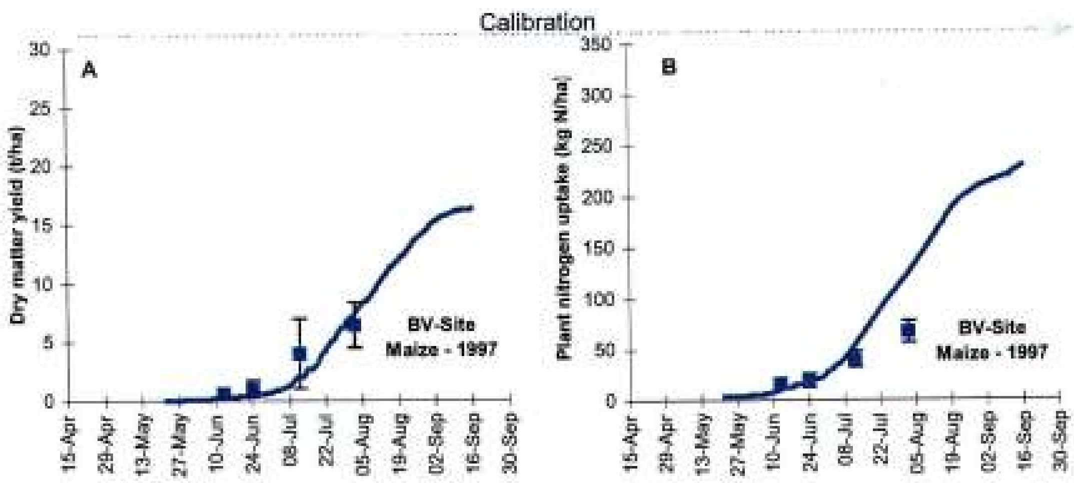

Figure 34.Simulated (solid line) and measured (point) crop growth and plant nitrogen uptake for maize at BV site. Error bars indicate one standard deviation

4.1.2.5. Nitrogen balance

Simulated annual inorganic nitrogen and water balance for the four calibration sites are shown in Table 15. Compared to the other years the net precipitation was much higher in 1996, and the estimated percolation and denitrification were therefore much higher in 1996.

Table 15. Inorganic nitrogen (kg N/ha·year) and water balance (mm/year) for the four calibration sites

	1996	*1997*	*1998*	*Average*	*1996*	*1997*	*1998*	*Average*
		BR				***TS***		
Crop type	*Maize*	*Maize*	*Maize*		*Maize*	*Maize*	*Alfalfa*	
Fertiliser input	*204*	*212*	*231*	*216*	*213*	*195*	*0*	*136*
Atmospheric input	*41*	*32*	*39*	*37*	*41*	*32*	*39*	*37*
Net Mineralisation	*203*	*163*	*103*	*156*	*191*	*148*	*150*	*163*
Denitrification	*132*	*68*	*62*	*87*	*131*	*46*	*41*	*73*
Plant Uptake	*265*	*212*	*282*	*253*	*251*	*228*	*174*	*218*
Leaching	*21*	*9*	*34*	*22*	*18*	*14*	*28*	*20*
Volatilisation	*35*	*46*	*31*	*37*	*22*	*30*	*0*	*17*
Changes in N-pool	*-4*	*72*	*-35*	*-5*	*24*	*58*	*-55*	*22*
Net Precipitation	*310*	*140*	*74*	*175*	*274*	*97*	*114*	*162*
Percolation	*315*	*144*	*115*	*191*	*244*	*118*	*200*	*187*
Changes in root zone	*-5*	*-3*	*-41*	*-16*	*30*	*-21*	*-86*	*-26*
		PR				***BV***		
Crop type	*Maize*	*Sugar Beet*	*Maize*		*Maize*	*Maize*	*Alfalfa*	
Fertiliser input	*188*	*119*	*195*	*167*	*259*	*235*	*0*	*165*
Atmospheric input	*41*	*32*	*39*	*37*	*41*	*32*	*39*	*37*
Mineralisation	*130*	*122*	*125*	*126*	*80*	*81*	*86*	*82*
Denitrification	*26*	*8*	*15*	*19*	*23*	*9*	*7*	*13*
Plant Uptake	*245*	*238*	*301*	*263*	*261*	*229*	*161*	*217*
Leaching	*10*	*4*	*3*	*6*	*10*	*6*	*1*	*6*
Volatilisation	*42*	*13*	*23*	*26*	*44*	*70*	*0*	*38*
Changes in N-pool	*36*	*12*	*16*	*34*	*41*	*36*	*-45*	*41*
Net Precipitation	*247*	*-21*	*-33*	*64*	*252*	*67*	*-6*	*104*
Percolation	*243*	*50*	*24*	*106*	*263*	*71*	*12*	*116*
Changes in root zone	*4*	*-72*	*-57*	*-41*	*-11*	*-5*	*-18*	*-11*

Large part of the nitrogen is lost from the system through gaseous losses (denitrification and volatilisation), whereas the size of the leaching itself is relatively small (6 – 22 kg N/ha·y). Important factors for the limited nitrogen leaching is the high denitrification and evapotranspiration. During the summer period the high evapotranspiration, low precipitation, shallow ground water table and strong capillary forces of the top soil (see Figure 8) causes an upward water flow. Moreover, the calibration sites are irrigated from underneath by a dense network of irrigation ditches ensuring sufficient soil humidity for agricultural purposes. An irrigation method, which further enlarges the upward water transport, and decreases both the percolation as well as the nitrogen leaching. Large amount of nitrogen is also transported upward due to the capillary rise. According to the modelling result the amount of nitrogen entering the root zone from underneath ranges from 9 to 22 kg N /ha per year.

Table 16. Estimated upward transport of nitrogen "negative leaching" (kg N/ha)

	1996	**1997**	**1998**	**Average**		**1996**	**1997**	**1998**	**Average**
BR-site	16	17	34	22	**TS-site**	17	21	14	17
PR-site	7	11	9	9	**BV-site**	5	13	11	10

The estimation of the upward nitrogen transport is however associated with some uncertainties, and the fluxes seem to be overestimated in areas with shallow groundwater tables. For calculating the upward nitrogen transport DAISY assumed equal nitrogen concentration in the deepest calculation node and in the water located just below the defined leaching depth. When the leaching depth is located just above the ground water table (e.g. the BR-site) the groundwater concentration is therefore equal to the concentration in the lowest calculation node. In this area the groundwater concentration is much lower, mainly because the saturated zone is characterised by a high horizontal flow velocity and a large amount of water infiltrating through the higher located Po di Goro as well as the Adriatic Sea (see section 3.2.). Hence, the upward nitrogen transport at the BR-site is overestimated. During the summer period – where the upward water transport is predominant – the leaching is therefore underestimated. If the nitrogen - entering the system from underneath - is not denitrified or leaving the system through plant uptake, the overestimated upward nitrogen fluxes causes an overestimated soil nitrogen concentration in the deeper soil layer. During the following winter period – where the downward water transport is predominant - the leaching will therefore be overestimated. The overestimated winter leaching would therefore counterbalance the underestimated summer leaching. In this case the overestimated upward nitrogen transport causes some uncertainties on the nitrogen leaching within the various seasons (summer and winter), but it is not expected to have a great influence on the annual leaching as a whole. This conclusion is based on the assumption that the nitrogen - entering the system from underneath - is not denitrified or leaving the system through plant uptake. The robustness of this assumption may be subject to discussion:

It can not be excluded that parts of the nitrogen entering the root zone from underneath – leaves the system through plant uptake. It should however be noted, that a large quantity of the upward nitrogen transport, occurs during the last part of the growing season, where the plant uptake of nitrogen has reach its maximum.

Nor can it be excluded that a part of the nitrogen entering the root zone from underneath is denitrified. However, the upward nitrogen transport occurs during the dry summer month. During this period soil humidity is often below the 80 % of saturation permitting denitrification to take place. Moreover the denitrification is related to the mineralisation potential of the soil. The denitrification in the deeper soil layer - below the tillage zone - is therefore minor due to the low mineralisation potential characterising these deeper soil layers.

4.1.3. Summary and concluding remarks

The overall model performance at the field scale was considered satisfactory for describing the nitrogen dynamics in the root zone. However, during short periods of time, the model overestimated the nitrogen concentration during the growing season. An overestimation caused by inadequate description of the transformation and transport processes of the applied NH_4-fertiliser occurring shortly after application. Problems arose from the description of the urea hydrolysis as well as the transport of water and nitrogen occurring through macro pores. Moreover the overestimated nitrogen concentration was found to influence also the simulated crop uptake of nitrogen, which in periods tends to be overestimated.

Modifications regarding the description of nitrogen volatilisation losses were found to be necessary in order to adapt the DAISY model to the Mediterranean conditions as investigated in this work. The ammonium volatilisation in connection with urea application was found to be an important process in the study area. This process was not included in the DAISY model and equation for estimating this process were therefore adopted from the EPIC model (Williams, 1984).

4.2. Modelling of nitrogen dynamics- catchment scale

After having proven its capability of describing the nitrogen root zone processes at the field scale the DAISY model was applied at a larger scale, with the purpose of quantifying the nitrogen leaching deriving from the arable land of the Bonello Catchment.

4.2.1. Model setup

The temporal and spatial distribution of nitrogen leaching was described by setting up the DAISY model for a number of soil colums which were representative of the study area. Different cropping sequences were applied on various soil units, classified as combination of different soil types and depth to the ground water table. Such classification criteria was based on the results of sensitivity analyses investigating the effect of groundwater fluctuations, depth of groundwater table, soil type and depth to the sandy layer on the extent of nitrogen leaching. These analyses were made for the conditions found at the TS-site, and involved the following:

Soil type: The hydraulic and chemical properties of the loamy soils were quite similar (Table 3, Table 4 and Figure 7). Modelling results also shows that even though denitrification and mineralisation were quite different, the average nitrogen leaching from soil profiles PR, TS, BV, BR and S5 were found to be very similar (see Table 17). In the model setup the loamy soil were therefore represented by two soil classes only: a Sandy Loam represented by soil profile S6, and an aggregated Silty Loam represented by the soil profile of the TS-site. Moreover, a sandy soil class, represented by the soil profile S7, was also included in the model setup. The hydraulic and chemical properties of these three soil classes can be found in Table 3 and Table 4.

Depth to the sandy layer: The loamy soils covering the western part of the study area overlay a sandy subsoil (see Figure 7). According to the soil profiles description the sandy layer started at a depth of 80 – 120 cm (see Table 4). Since the depth to the sandy layer is expected to influence the nitrogen leaching from the root zone, this parameter was also varied from 80 to 140 cm. The modelling results shows that increasing the depth to the sandy layer caused a decrease of percolation, which in turn caused a decrease of nitrogen leaching (see Table 18). The limited number of analysed soil profile made it difficult to describe the spatial distribution of the depth to the sandy layer. Thus in all the loamy part of the study area, the sandy layer was assumed to begin at the depth of 80 cm depth.

Groundwater fluctuation and level nine different time series of groundwater table were retrieved from the MIKESHE simulation and applied as lower boundary conditions for the hydrological module. For evaluating the impact of groundwater fluctuation on nitrogen leaching the selected time series represented the combination of three average groundwater depths (1.3, 1.7 and 2.3 meter) and three groundwater fluctuations (low, medium and high). The analysis was carried out for two soil types: a sand soil (S7) and a loam soil (TS-site). The hydraulic properties of these soil types are given in Table 4. For evaluating the impact of groundwater level the average groundwater table varied from 1.1 meter to 3.5, whereas the groundwater fluctuations were the same in all the time series. Also here the analysis was carried out for two soil types: a sand soil (S7) and a loam soil (TS-site).

The actual depth to the groundwater table and the groundwater fluctuation had both a great influence on the nitrogen leaching (Table 19 and Table 20). During the simulation period, the nitrogen leaching was found to increase with increasing depth to the groundwater table (Table

19). This trend is mainly a result of the decreased evapotranspiration and denitrification. In the study area the upward transport of water and nitrogen are predominant during large part of the year (see section 4.1.2.5.). Hence, lowering the groundwater table decreases evapotranspiration, the soil water content and also the denitrification, which only takes place at a high soil water contents (see Table 19).

Table 17. Estimated nitrogen (kg N/ha·y) and water balance (mm/y) for different soil profiles. Values are averaged over the three years period 1996-1998.

Soil Profile	*Site PR*	*Site BR*	*Site TS*	*Site BV*	*S5*	*S6*
Net mineralisation	*109*	*110*	*110*	*92*	*139*	*135*
Denitrification	*26*	*48*	*61*	*6*	*85*	*76*
Uptake	*226*	*208*	*193*	*224*	*201*	*198*
Leaching	*17*	*17*	*18*	*19*	*17*	*24*
Precipitation	*697*	*697*	*697*	*697*	*697*	*697*
Percolation	*161*	*213*	*195*	*200*	*246*	*203*
Evapotranspiration	*579*	*522*	*531*	*542*	*496*	*534*

Table 18. Estimated nitrogen (kg N/ha·y) and water balance (mm/y) for various depths to the sandy layer. Values are averaged over the three years period 1996-1998

Depth to sandy layer (m)	*80*	*110*	*130*	*140*
Net mineralisation	*109*	*110*	*109*	*109*
Denitrification	*61*	*58*	*59*	*59*
Uptake	*193*	*194*	*194*	*194*
Leaching	*17*	*14*	*12*	*11*
Precipitation	*696*	*696*	*696*	*696*
Percolation	*195*	*185*	*181*	*175*
Evapotranspiration	*532*	*531*	*531*	*531*

Table 19. Estimated nitrogen (kg N/ha·y) and water balance (mm/y) for various depths to the groundwater table. Values are averaged over the three years period 1996-1998.

Average ground water table (m)	*1.1*	*1.3*	*1.7*	*2.0*	*2.3*	*2.6*	*3.0*	*3.3*	*3.5*
					Silty Loam (TS-Site)				
Net mineralisation	*107*	*109*	*109*	*110*	*110*	*111*	*111*	*111*	*111*
Denitrification	*77*	*68*	*64*	*61*	*61*	*59*	*57*	*56*	*55*
Plant uptake	*181*	*190*	*191*	*192*	*193*	*194*	*195*	*196*	*196*
Leaching	*7*	*9*	*11*	*15*	*17*	*18*	*19*	*20*	*20*
Precipitation	*697*	*697*	*697*	*697*	*697*	*697*	*697*	*697*	*697*
Percolation	*154*	*158*	*169*	*189*	*195*	*189*	*178*	*168*	*171*
Evapotranspiration	*557*	*541*	*533*	*531*	*531*	*532*	*531*	*530*	*530*
					Sand (S7)				
Net mineralisation	*79*	*80*	*80*	*80*	*80*	*80*	*80*	*80*	*81*
Denitrification	*60*	*60*	*50*	*44*	*43*	*41*	*42*	*42*	*42*
Plant uptake	*189*	*189*	*187*	*187*	*187*	*188*	*188*	*187*	*187*
Leaching	*31*	*31*	*39*	*52*	*59*	*63*	*63*	*64*	*65*
Precipitation + Irrigation	*785*	*785*	*785*	*785*	*785*	*785*	*785*	*785*	*785*
Percolation	*254*	*254*	*270*	*287*	*293*	*285*	*272*	*264*	*260*
Evapotranspiration	*534*	*534*	*531*	*529*	*529*	*528*	*529*	*528*	*528*

Table 20. Estimated nitrogen (kg N/ha·y) and water balance (mm/y) for various groundwater fluctuations. Values are averaged over a three years period 1996-1998.

Ground water fluctuation	Low	Medium	High	Low	Medium	High	Low	Medium	High
Average groundwater table(m)	1.3	1.3	1.3	1.7	1.7	1.7	2.3	2.3	2.3
				Silty Loam (TS-Site)					
Net mineralisation	109	109	107	109	109	109	110	111	110
Denitrification	66	65	65	61	62	63	60	60	61
Uptake	191	190	189	192	192	191	194	194	193
Leaching	9	11	12	13	15	16	16	17	18
Precipitation	697	697	697	697	697	697	697	697	697
Percolation	163	190	217	172	207	241	171	194	222
Evapotranspiration	533	533	534	531	531	531	532	532	531
				Sand (S7)					
Net mineralisation	74	75	-	75	75	-	75	75	-
Denitrification	49	51	-	42	42	-	39	41	-
Uptake	187	187	-	186	187	-	187	186	-
Leaching	32	36	-	46	51	-	52	56	-
Precipitation irrigation	785	785	-	785	785	-	785	785	-
Percolation	257	278	-	270	295	-	277	290	-
Evapotranspiration	531	531	-	530	529	-	528	529	-

It is interesting to observe that when the groundwater table was lowered to approximately 2.5 meter percolation increased as a consequence of the lower evapotranspiration. Below 2.5 meter the groundwater table no longer had any influence on the evapotranspiration which remained constant, whereas percolation - despite of the constant evapotranspiration – decreased (see Table 19). This shift is further illustrated in Figure 35. showing the downward and upward water transport from the root zone. When the groundwater table was lowered below 2.5 meter the upward transport of water decreases (Figure 35). Along with the smaller capillary transport the soil water content in the root zone decreases, allowing smaller quantity of the infiltrating water to percolate through the root zone.

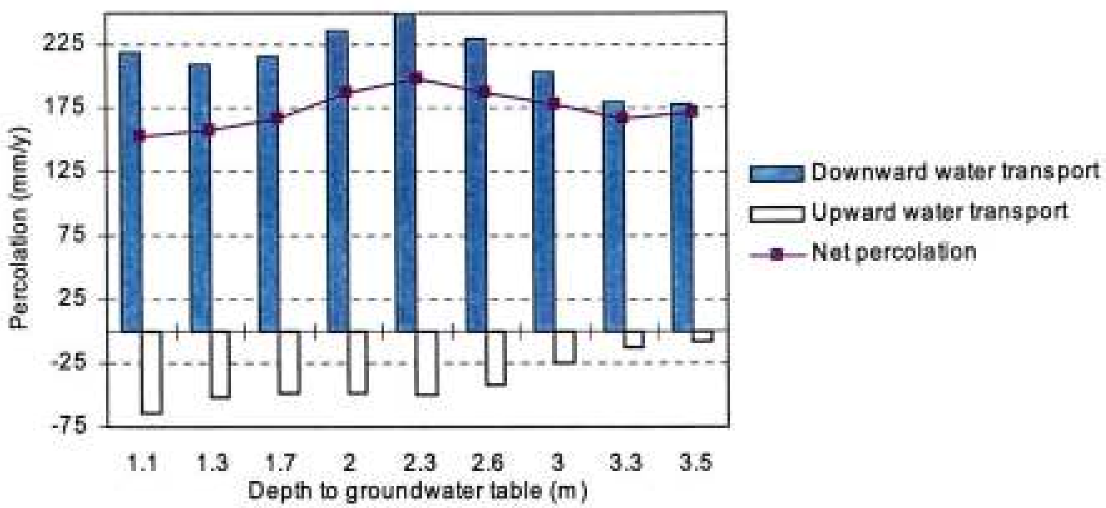

Figure 35. Estimated root zone percolation as a function of depth to the ground water table.

Based on these results it was decided to use five different interval of depth to the groundwater table as classification criteria's. A groundwater map was retrieved from the hydrological model MIKESHE, and the spatial distribution of the various groundwater table intervals was identified using the GIS system included in the MIKESHE package. Areas with groundwater table located within a range of 50 cm were lumped together into the five groundwater intervals indicated in Figure 36 – left. Various soil units - classified in terms of a combination of selected soil types and groundwater intervals - were then identified by combining the groundwater interval map with the soil class map (Figure 36). Subsequently, the groundwater dynamics was carefully analysed, in order to find time series of groundwater table that is most representative each of the identified soil unit in terms of groundwater level and fluctuation (Figure 37). Thus, for each soil unit, a single timeserie of groundwater table was retrieved from the MIKSHE simulation and applied as lower boundary conditions for the hydrological module in DAISY.

Within the GIS system the crop distribution of 1997 was when combined with the classified soil units, leading to a total number of 33 computational columns (Table 21). Subsequently, eight different cropping sequences were constructed and applied on the various soil units in order to give a realistic description of the temporal and spatial distribution of the cropping pattern (Table 22). The agricultural modelling included three crop types (maize, alfalfa and sugar beet) and the internal statistical distribution between these three crops was assumed to be representative for the arable land (Table 22). Moreover, it should be noted that rice fields (approximately 5% of the arable land) were not modelled due to model limitations in the simulation of the rice crop growth as well as to the different hydrology characterising a rice field.

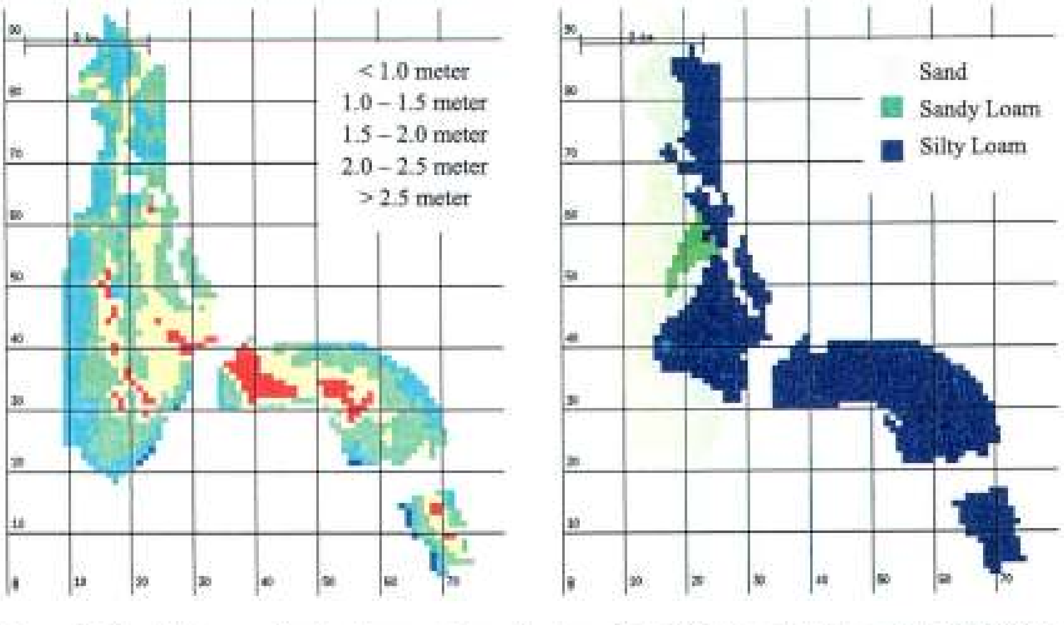

Figure 36. Simulation area for the large-scale application of the DAISY model. The spatial distribution of the various groundwater classes is illustrated on the left figure (legend indicate average depth to groundwater table), whereas the various soil classes are shown on the right figure.

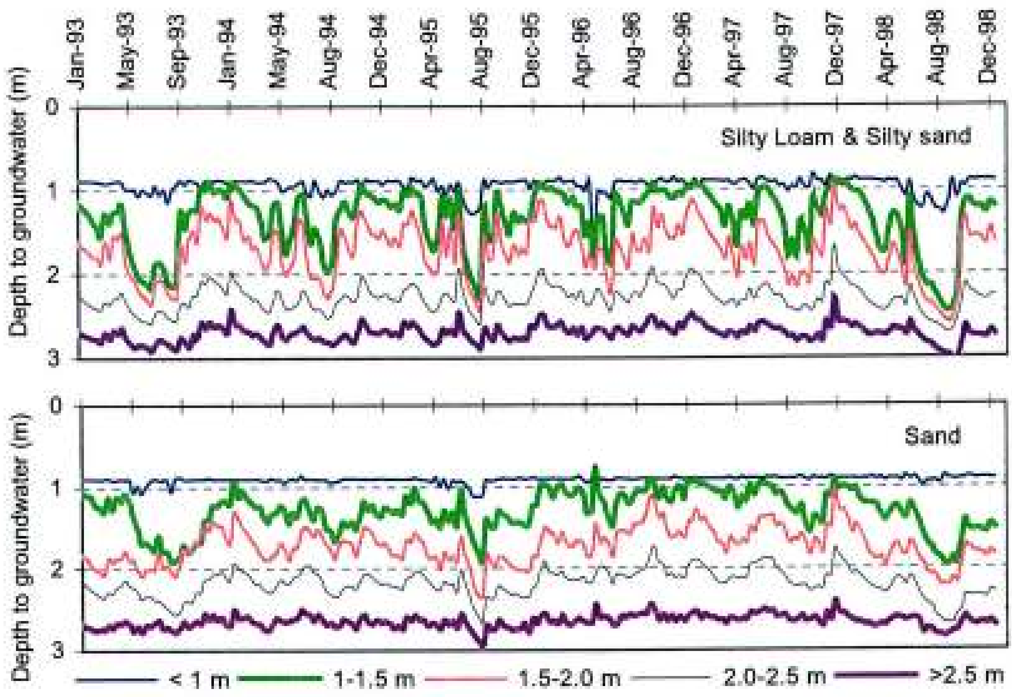

Figure 37. Timeseries of groundwater table for the various soil units.

Table 21. Computational columns for the large scale application of DAISY

Comp Column	Soil type	Depth to gwt (m)	Crop Seq.	Area (ha)	Comp. Column	Soil type	Depth to gwt (m)	Crop Seq.	Area (ha)
1	TS: Silty Loam	1.5-2.0	1	196	18	S6: Silty Sand	1.5-2.0	5	13
2	TS: Silty Loam	>2.5	1	74	19	S6: Silty Sand	1.5-2.0	7	3
3	TS: Silty Loam	2.0-2.5	2	220	20	S6: Silty Sand	2.0-2.5	7	10
4	TS: Silty Loam	< 1.0	3	1	21	S6: Silty Sand	1.0-1.5	8	2
5	TS: Silty Loam	1.0-1.5	3	22	22	S6: Silty sand	2.0-2.5	8	11
6	TS: Silty Loam	1.5-2.0	3	32	23	S7: Sand	< 1.0	2	4
7	TS: Silty Loam	2.0-2.5	3	30	24	S7: Sand	2.0-2.5	2	73
8	TS: Silty Loam	>2.5	3	8	25	S7: Sand	1.0-1.5	3	121
9	TS: Silty Loam	< 1.0	5	8	26	S7: Sand	1.5-2.0	3	45
10	TS: Silty Loam	1.5-2.0	6	199	27	S7: Sand	2.0-2.5	3	13
11	TS: Silty Loam	< 1.0	7	1	28	S7: Sand	>2.5	3	7
12	TS: Silty Loam	1.0-1.5	7	63	29	S7: Sand	1.0-1.5	4	152
13	TS: Silty Loam	2.0-2.5	7	46	30	S7: Sand	1.5-2.0	5	176
14	TS: Silty Loam	>2.5	7	16	31	S7: Sand	>2.5	5	11
15	TS: Silty Loam	1.0-1.5	8	66	32	S7: Sand	1.0-1.5	6	3
16	S6: Silty Sand	1.0-1.5	3	4	33	S7: Sand	1.5-2.0	7	26
17	S6: Silty sand	1.5-2.0	3	3				Total	1659

Each of the eight cropping sequences was characterised by a standard set of tillage and fertilisation operations, reflecting the general agricultural practices used in the area. The applied fertilisation rates for the various crop and soil types are illustrated in Table 23. The amount of nitrogen applied at the loamy soil represents the average fertilisation rate at the four calibration sites. Also the estimated volatilisation loss represents the average volatilisation loss of urea (see Table 12.). It should be noted that there is a large internal variation in the amount of nitrogen deriving from fertiliser input. E.g. at the four calibration sites the fertilisation rate for maize varied between 188 – 259 kg N/ha·y.

Table 22. Description of the eight cropping sequences included in the DAISY simulation

Crop Sequence	1993	1994	1995	1996	1997	1998
1	Sugar beet	Maize	Maize	Sugar beet	Maize	Maize
2	Alfalfa	Maize	Maize	Maize	Maize	Sugar beet
3	Maize	Sugar Beet	Maize	Maize	Sugar Beet	Maize
4	Maize	Alfalfa	Alfalfa	Alfalfa	Maize	Maize
5	Maize	Maize	Sugar beet	Maize	Maize	Alfalfa
6	Maize	Alfalfa	Alfalfa	Alfalfa	Alfalfa	Maize
7	Maize	Maize	Maize	Maize	Alfalfa	Alfalfa
8	Alfalfa	Maize	Sugar Beet	Maize	Maize	Maize
	Area distribution(% of simulation area)					
Maize	61%	61%	60%	62%	61%	60%
Alfalfa	22%	22%	22%	22%	22%	23%
Sugar Beet	17%	17%	18%	16%	17%	17%

Exact information about the fertilisation rate in the sandy area was not available. However, according to general agricultural practices (Barboni, 1999 personal communication) the application rate in the sandy areas was about 50% higher than in the loamy area. In the sandy area the volatilisation loss was estimated for each of the eight cropping sequences, by following the procedure described in Appendix II, and average losses are given in Table 23.

Table 23. Fertiliser application rates and estimated volatilisation losses for various soil and crop types. Alfalfa did not receive any nitrogen fertiliser.

	Type	NH_4– content (% of applied N)	Application rate (kg N/ha) Loam	Sand	Volatilisation loss (%) Loam	Sand
			Maize			
Before sowing	N-P	100	62	93	-	-
After sowing	Urea	100	153	230	20	30
			Sugar beet			
Before Sowing	N-P-K	100	48	72	-	
After sowing	Urea	100	71	107	20	30

After alfalfa harvesting a large amount of organic nitrogen is incorporated into the soil during plowing. This lead to a higher mineralisation with the consequence that larger amount of nitrogen is available for the following crop. The farmers generally compensate for the higher nitrogen availability by reducing the fertiliser application. According to general agricultural practice (Barboni, 1999 personal communication) the fertiliser application for the crop following alfalfa was reduced 50% in the model setup.

During the growing season irrigation water is taken from the Po di Goro and distributed throughout the area via the river system. The irrigation water is applied to the fields via channel release from a large number of irrigation ditches. In the eastern –loamy- part of the area this methods of "underneath irrigation" is predominating, whereas in the western -sandy- part of the area superficial distribution of the irrigation water is also used. In the loamy soil the irrigation water was therefore explicit included via the applied ground water table, whereas the irrigation water in the sandy area was included directly in the model setup via the DAISY irrigation module. Thus, in the sandy area irrigation was effectuated when the soil water pressure potential at 40 cm depth decreased below a predefined value of - 2.50 meter.

The simulation involved a six years simulation period from 1993 – 1998, with a preceding initialisation period of three year.

4.2.2. Results and discussion

The description of the large-scale nitrogen dynamics was based upon 33 computational columns representing 33 identified sub-areas. From an agricultural point of view (crop type, soil properties, depth to groundwater table, agricultural practices, etc.) these sub-areas were assumed to be homogeneous, and the model results from one computational column were therefore assumed to be representative for that specific area. The spatial distribution of the nitrogen leaching is illustrated in Figure 38; whereas simulated nitrogen balance for the three soil types is given in Table 24.

Table 24. Simulated inorganic nitrogen (kg N/ha·y) and water (mm/year) balances for the various soil types. The results are weighed averages over a six-year simulation period 1993-1998

	Sand	Sandy Loam	Silty Loam	All area
Simulation area (ha)	*631*	*46*	*982*	*1659*
Fertiliser input	*228*	*155*	*136*	*173*
Atmospheric input	*42*1)*	*40*	*40*	*41*
Net Mineralisation	*114*	*165*	*119*	*118*
Denitrification	*69*	*92*	*64*	*66*
Plant uptake	*245*	*236*	*204*	*221*
Leaching	*27*	*9*	*9*	*16*
Worse case leaching	*39*	*21*	*20*	*27*
Volatilisation	*47*	*22*	*19*	*29*
Changes in N-pool	*-4*	*1*	*-1*	*0*
Net precipitation	*205*2)*	*112*	*96*	*139*
Percolation (mm)	*208*	*123*	*104*	*145*
Changes in the root Zone	*-3*	*-11*	*-8*	*-6*

**1) Included nitrogen added with the irrigation water*
**2) Including a superficial irrigation input of 31 mm/year*

Common for all three soil types is that a large part of the nitrogen leaves the system through gaseous losses.

The estimated denitrification of 66 kg N/ha indicates that 56% of the mineralised nitrogen was lost from the system through denitrification. This is due the shallow groundwater table as well as to the strong capillary forces of the top soil, which maintain high soil water content in the root zone. Results from the calibration sites also showed, that the soil water content - during the larger part of the year - was above the 80% of saturation that permit denitrification to take place (Figure *23* - Figure 24). The volatilisation was also relatively high, even though estimations are affected by some uncertainties because the volatilisation model as applied here was not calibrated.

Compared to gaseous losses the leaching itself was found to be relatively small with an estimated average of 16 kg-N/ha. Especially the loam soils were characterised by a low leaching rate with an estimated average loss of 9 kg N/ha. Important factors for this limited leaching is the high denitrification. Moreover the combination of a high evapotraspiration, capillary forces of the soil, shallow water table as well as the applied irrigation method causes a large upward transport of water and nitrogen during the summer (see further discussion in section 4.1.2.5).

The '"worst case leaching" in Table 24 and Figure 38 consider only the downward transport of nitrogen. These estimates thereby represent a theoretical worse case scenario for the nitrogen leaching describing the maximum level of nitrogen leaching which can be found during wet winter periods with high net precipitation. Comparing the leaching with the "worst case" indicates that the upward nitrogen transport account for 11 kg N/ha·y.

Compared to the loamy soil the sandy area had a higher percolation leading to a larger nitrogen leaching. Thus, the average leaching from the sandy soil was in the order of 27 kg /ha and 8% of the sandy area had nitrogen leaching above 40 kg/ha. The difference in between the various soils type is clearly illustrated in Figure 38. The higher leaching rates characterising the western part of the area corresponds to the sandy soils, whereas the lower leaching rates characterising the eastern part of the area corresponds to the loamy soils.

It should be noted that the model performance was not evaluated at the sandy soils. Compared to the loamy soils the water balance at the sandy soils is characterised by different hydraulic soil properties as well as a superficial application of irrigation water. (See Section 4.2.1.). The model description regarding these differences has not been evaluated in this work, and some uncertainties may therefore be associated with model result deriving from the sand soil.

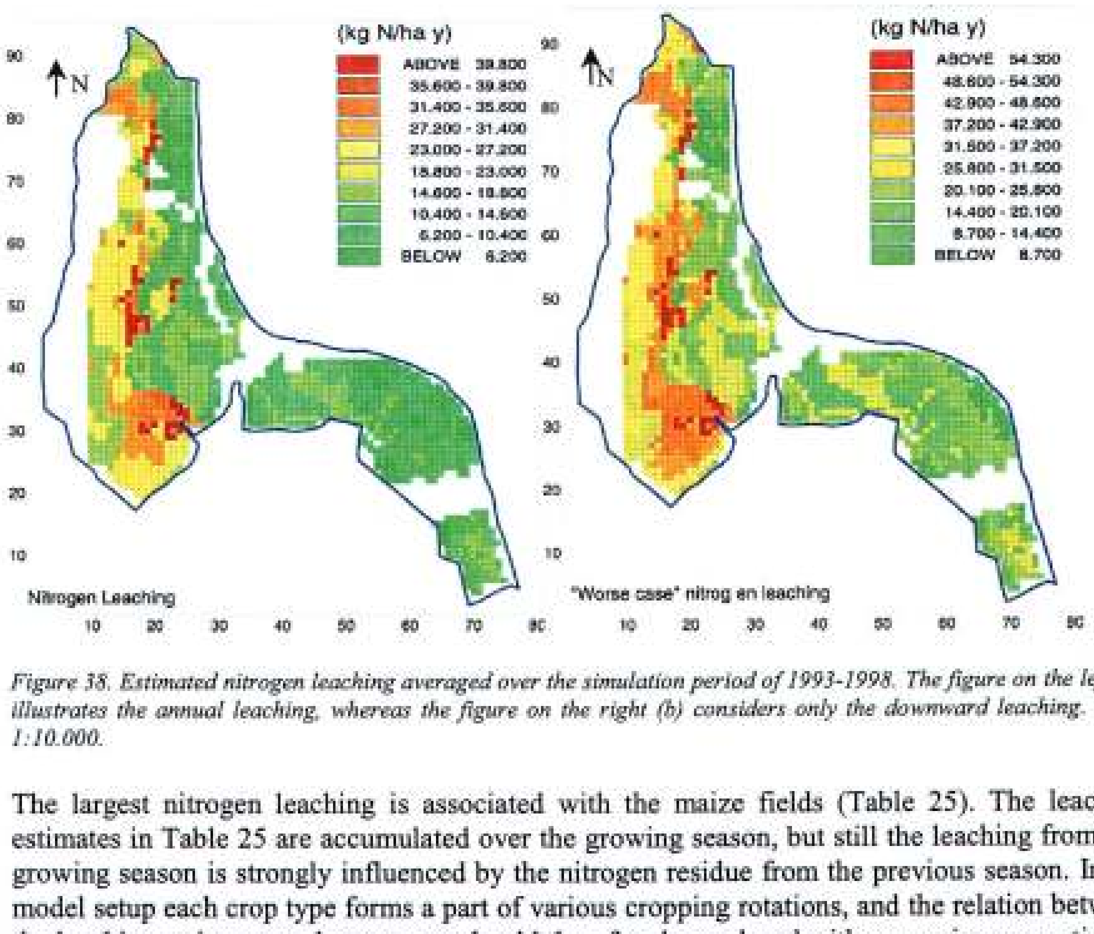

Figure 38. Estimated nitrogen leaching averaged over the simulation period of 1993-1998. The figure on the left (A) illustrates the annual leaching, whereas the figure on the right (b) considers only the downward leaching. Scale 1:10.000.

The largest nitrogen leaching is associated with the maize fields (Table 25). The leaching estimates in Table 25 are accumulated over the growing season, but still the leaching from one growing season is strongly influenced by the nitrogen residue from the previous season. In the model setup each crop type forms a part of various cropping rotations, and the relation between the leaching estimates and crop types should therefore be analysed with some circumspection.

Table 25. Nitrogen leaching for the various crop types (kg N/ha·y). The values are weighed averages over the six years simulation period 1993-1998. The standard deviation is indicated in the bracket.

	Area (% of simulation area)	*Sand*	*Loam*
Maize	*60%*	*30 (5)*	*9 (2)*
Alfalfa	*22%*	*19 (3)*	*5 (2)*
Sugar beet	*17%*	*15 (1)*	*4 (1)*

Alfalfa has a much longer growing season (typically 3 – 5 year) than the other crop types. The predominating nitrogen source for alfalfa derived from fixation of atmospheric nitrogen, and no fertiliser is therefore applied to the fields. During the growing season, the nitrogen leaching is generally low due the combined effect of a low soil nitrogen concentration and low percolation, caused by the permanent plant cover. After harvesting a large amount of organic nitrogen is incorporated into the soil with the plant residue, and the mineralisation, soil nitrogen concentration as well as the nitrogen leaching risk increased substantially (Figure 39). Alfalfa is currently growing at the calibration sites BV and TS. Calibration data for evaluating the model description of the nitrogen dynamics during the leaching season following harvesting and ploughing of alfalfa is therefore not available yet. Hence, the leaching estimates associated with the alfalfa fields may therefore be subject to some uncertainties.

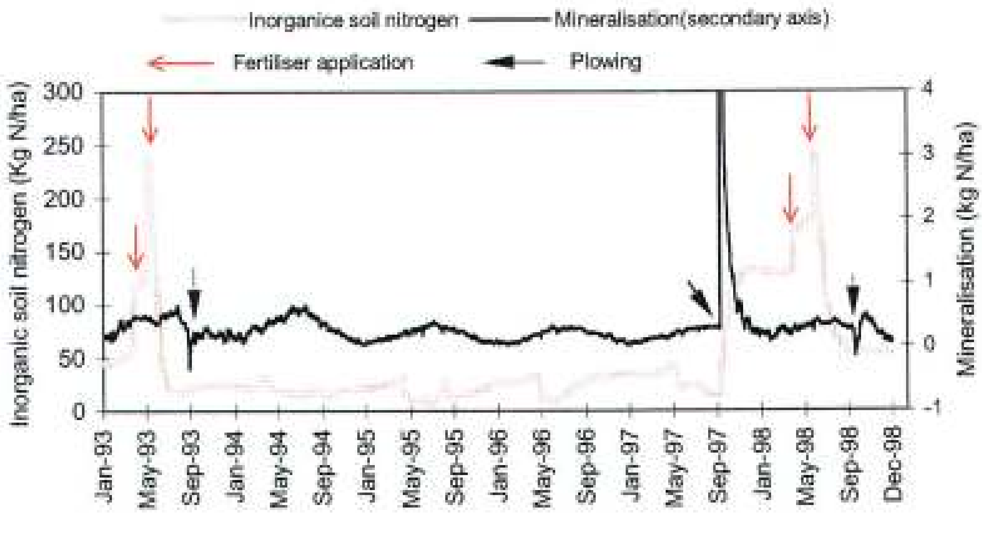

Figure 39. Inorganic soil nitrogen and mineralisation during the growth of alfalfa. The results derive from a selected computational column at the loamy soil with the following crop rotation: maize 1993, alfalfa 1994 – 1997, maize 1998.

In accordance with the results from the initial sensitivity analysis (Table 19) the nitrogen leaching was found to increase with increasing depth to the groundwater table (see Figure 40). The leaching estimates in Figure 40A are averaged over different soil units and cover different cropping sequences. Thus, apart from the groundwater table the leaching may also be influenced by differences deriving from crop rotation. The estimates in Figure 40B results from selected computational columns, representing the same cropping rotation, applied on different soil units. The increased nitrogen leaching in Figure 40B is therefore caused by the groundwater dynamics alone.

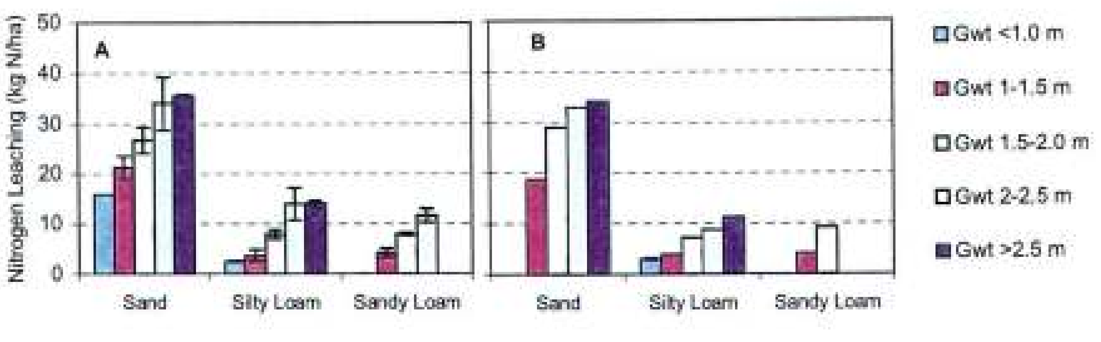

Figure 40. Nitrogen leaching averaged over the six years period 1993-1998 (kg N/ha·y). In figure A the leaching is averaged over the various soil units classified in terms of combination of soil type and groundwater classes. In figure B the leaching estimates derive from selected computational columns representing the same crop rotation.

Results from the hydrological modelling indicates that 17% and 21% of the total water input to the system derives from boundary infiltration and irrigation water, respectively (see Table 7 & Table 8). They also indicate, that the actual percolation only accounts for 43% of the channel discharge, whereas the remaining 57% derive from irrigation tail loss as well as boundary infiltration (see Table 8). The relative importance of nitrogen fluxes brought into the study area by irrigation water and boundary infiltration was evaluated by setting up a total nitrogen mass balance for the study area.

Figure 41 illustrates the various nitrogen fluxes entering the groundwater and the channel system. The leaching estimate was based upon average leaching rates as calculated by DAISY. The nitrogen fluxes from irrigation and boundary infiltration were estimated by multiplying measured nitrogen concentrations with the water fluxes retrieved from the MIKESHE model. The "net nitrogen input" was then estimated by summing up the various nitrogen fluxes, whereas "discharged nitrogen" is the measured amount of nitrogen pumped out into the Sacca di Goro. A detailed description of the data foundation and the applied calculations are given in Appendix III.

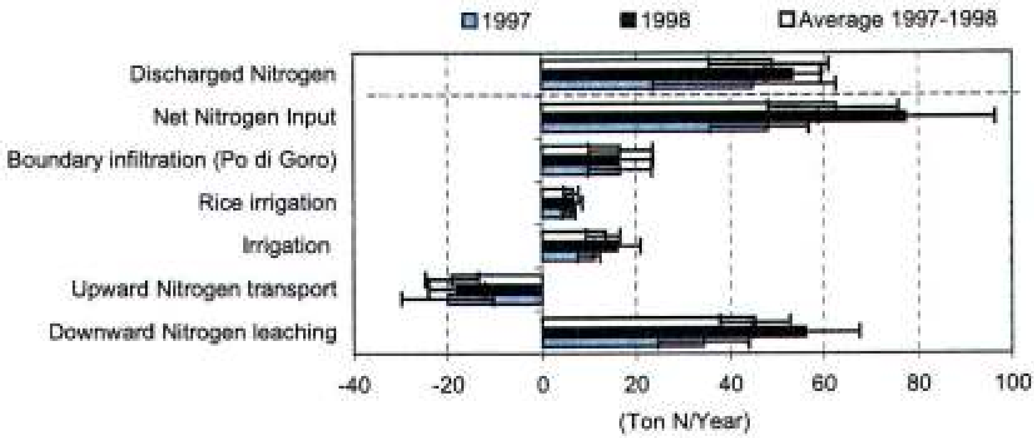

Figure 41. Nitrogen mass balance for the groundwater and channels system. Error bars for the leaching estimates represents standard deviation, whereas error bars for the other estimates represents minimum and maximum values.

The irrigation water and boundary infiltration did bring quite a bit of nitrogen into the system (see Figure 41). The results indicated that 24% and 20% of the total amount of nitrogen entering the groundwater and channel system derive from the irrigation water and boundary infiltration, respectively. The rice fields were characterised by a large irrigation demand, and 33% of the nitrogen entering the system with the irrigation water was therefore allocated especially to these fields.

Moreover, the balance was characterised by a large upward transport of nitrogen (negative leaching). It is quite interesting to see that nitrogen, which during the summer is entering the system with the irrigation water, is more or less counterbalanced by this upward nitrogen transport.

The estimated amount of nitrogen entering the groundwater and channel system (net nitrogen input) was found to be somewhat higher that the measured amount of nitrogen leaving the system through channel discharge (discharged nitrogen). The applied modelling system describes the nitrogen transformation processes in the root zone, while possible transformation processes occurring in the channels and saturated zone are not taken into account. Hence, the observed difference between measured and estimated fluxes is presumably due to various nitrogen transformation processes, such as denitrification, occurring elsewhere than the root zone. It should also be noted that two main uncertainties are associated with the obtained estimates. First, the nitrogen concentration in the channels was assessed from a very limited number of measurements (see appendix III). Second, the leaching estimates were based upon a model setup including only three crop type only, whereas the actual cropping pattern was much more diverse. Indeed the applied modelling system did not consider transformation processes occurring in the rice field, and any leaching input from these field was not included in the total leaching.

4.2.3. Summary and concluding remarks

The estimated amount of nitrogen entering the study area was 22% higher than the measured amount of nitrogen leaving the system through channel discharge. Taking the model limitation and uncertainties into account this is a satisfactory outcome, indicating that the applied methodology for the large-scale application of DAISY was appropriate. It should be noted that this result was based upon an overall nitrogen balance with respect to the nitrogen transformation processes in the root zone only. It would be recommended that further work is carried out with respect to quantifying the groundwater and in-stream transformation processes, as this would increase the confidence level of the applied methodology.

The predominant part of the area was characterised by low nitrogen leaching rate, whereas higher rates were restricted a relatively small part of the study area. The loamy soils were characterised by average leaching rates of 9 kg N/ha. The average nitrogen leaching from the sandy soil was instead found to be 26 kg /ha, with only 8% of the sandy area showing nitrogen leaching rates above 40 kg/ha. Important factors that contributed to the limited nitrogen leaching were the high denitrification. The groundwater dynamic was also found to have a great impact on the nitrogen leaching rates, particulary the nitrogen leaching was found to increase with increasing depth to the groundwater table.

The nitrogen balance was affected by large gaseous losses. The estimated denitrification was about 66 kg N/ha·y accounting for 56% of the mineralised nitrogen. The cause of this large denitrification was the shallow groundwater table. Indeed strong soil capillary forces maintain the high soil water content in the root zone that allows denitrification to take place.

Additionally, the volatilisation loss of 29 kg N/ha accounted for 16% of the fertiliser input. Part of the volatised nitrogen may re-enter the agricultural areas and the Sacca di Goro though atmospheric deposition. Thus, besides nitrogen leaching volatilisation can also be an important pathway for the nitrogen export to the coastal lagoon. However, the lagoon covers an area of 25.6 Km^2 whereas the inland drainage basin covers a much larger area of 880 km^2. The amount of nitrogen entering the Sacca through atmospheric deposition is therefore very small compared to the input deriving from nitrogen leaching from the agricultural land. According to Leip (2000) the atmospheric deposition of nitrogen accounts for only 5% of the total nitrogen input to Sacca di Goro.

The study area was characterised also by a large amount of nitrogen entering the system through the boundaries (boundary infiltration) and together with irrigation water. The results indicated that 24% and 20% of the total amount of nitrogen entering the groundwater and channel system derives from irrigation water and boundary infiltration, respectively.

4.3. Modelling of nitrogen dynamics - Scenarios Analysis

The regional authorities in Emilia-Romania are currently implementing the Council Regulation 1257/1999 (EC, 1999). The regulation is funded by the European Commission through the European Agricultural Guidance and Guarantee Fund (EAGGF), which gives incentives for reducing the use of fertilisers/plant protection products, for the extensification of crop and livestock production, and for the long-term set aside of arable land. The program works on a voluntary basis and farmers are subsidised to participate in the various action programs. For instance the program provide the farmers with subsidies through a five-year period for adjusting their fertiliser application according to an established nitrogen balance. Hence, soil nitrogen concentration is measured during spring and a nitrogen balance is prepared in order to determine the fertiliser requirement for the various crops. Regardless of the estimated nitrogen requirement the yearly fertilisation rate must not exceed the established maximum limit of e.g. 200 kg N/ha for maize and 100 kg N/ha for sugar beet (Regione Emilia-Romagna, 2000). The implementation of this action program is expected to reduce the overall fertiliser input in the order of 30% (Andreotti, 1998)

By using the modelling system calibrated and validated in this present work, the impact of possible action program was estimated for the Bonello catchment through scenario analyses. The objective of such analysis was to analyse the catchment response to reduced nitrogen input, and to evaluate which environmental improvement could be obtained by implementing alternative management practices. Hence, the model application for scenario analysis comprised the following:

- ***Basic model setup:*** A basic model setup representing the actual situation in the study area.
- ***Scenario I:*** Fertiliser reduction for all crops receiving nitrogen fertiliser e.g. maize and sugar beet. The fertiliser input for the loamy and sandy soil were reduced in the order of 10%- 40% and of 10% - 50% respectively (see Table *26*.).
- ***Scenario II:*** Fertiliser was only reduced in the maize field, whereas the fertiliser level for Sugar beet was kept constant at the original level in the basic model setup. The fertiliser in the maize field was reduced as described in Scenarios I.
- ***Scenario III:*** Reducing the fertiliser input to the maximum limit set out by the EC regulation 1257/1999.Thus, the applied fertiliser rate was 200 kg N/ha for maize and 100 kg N/ha for Sugar Beet (see Table *26*).
- ***Scenario IV:*** Decreasing the groundwater table depth in the arable land.
- ***Scenario V:*** Combination of Scenario III and Scenario IV. Thus, the groundwater table depth was decreased and the fertiliser input was reduced to the maximum limit set out by the EC regulation 1257/1999.

Table 26. Fertiliser application rate for the various scenarios (kg N/ha·y)

	Basic Model-setup	*Scenarios I & II*					*Scenarios III Max. fertiliser rate*
	Traditional fertiliser level	*-10%*	*-20%*	*-30%*	*-40%*	*-50%*	*EC reg.1257/1999*
Maize -Sand	*323*	*291*	*258*	*226*	*194*	*162*	*200*
Maize-Loam	*215*	*194*	*172*	*150*	*129*	*-*	*200*
Sugar Beet-Sand	*179*	*161*	*143*	*125*	*107*	*90*	*100*
Sugar beet-Loam	*119*	*107*	*95*	*83*	*71*	*-*	*100*

The previous results indicated that the groundwater table had a huge impact on the nitrogen leaching. The impact of groundwater regulation on nitrogen leaching and harvest yield therefore provided an interesting analysis, which was further evaluated in *scenario IV and V.*

A very dense channel network characterises the study area, and a pumping station situated at the channel outlet pumps all drainage water out. The pumping station begins to operate when the water level in the main channel leading to the pumping station exceeds a specified level. Thus, the level of the groundwater table can be regulated both in terms of the specified maximum level at the channel outlet, as well as though the large number of internal channels regulator, which are used during the growing period. For scenarios IV and V a situation with a higher level of groundwater table was first simulated with the hydrological model MIKSHE. Here the higher level of groundwater table was simulated by increasing the field irrigation input with an extra 100 mm/y. Subsequently, time series of groundwater table, were retrieved from the model output and applied as lower boundary condition for the hydrological module of DAISY (See Figure 42.).

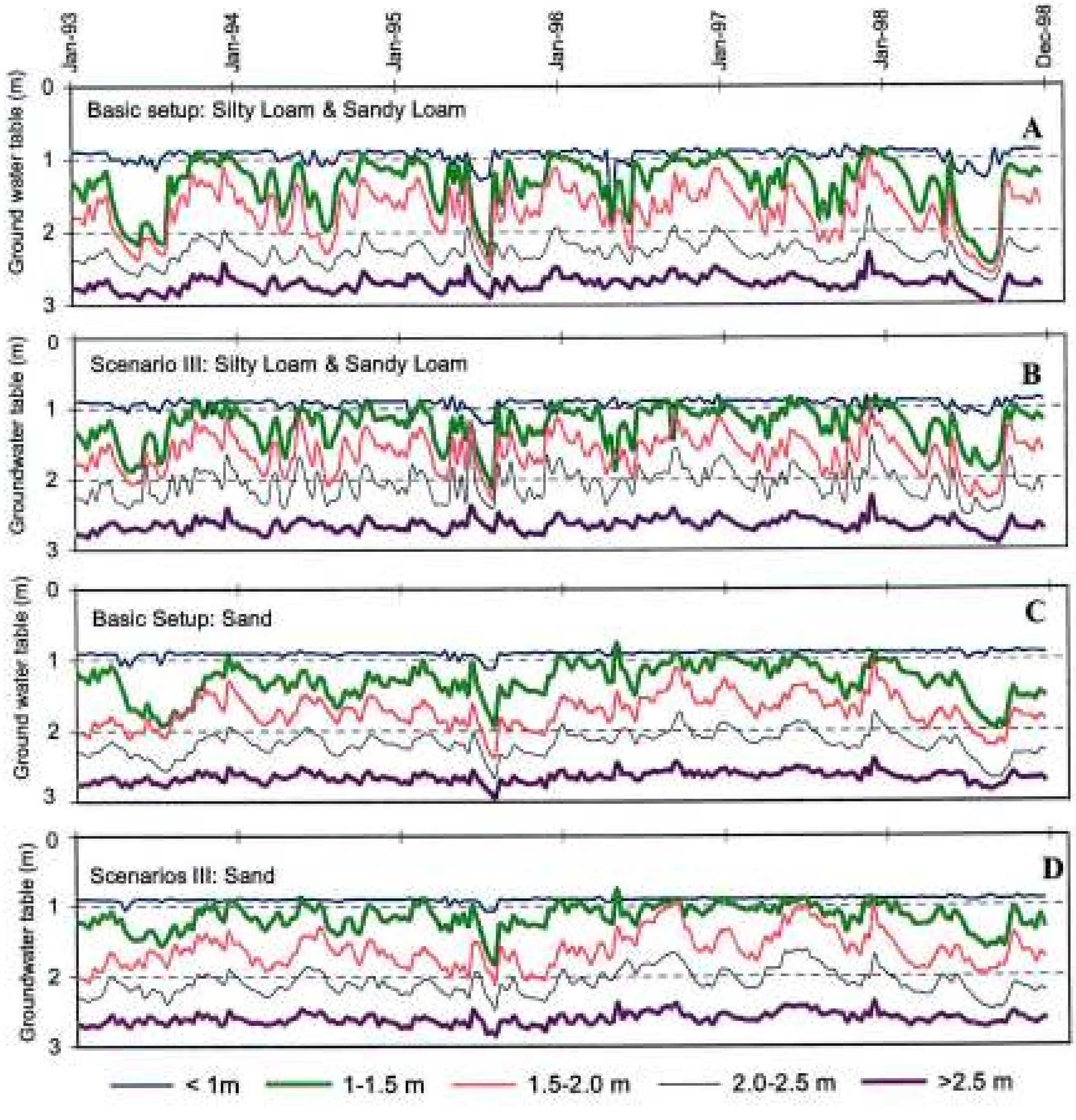

Figure 42. Time series of groundwater table depth applied in the basic model setup (graph A and C) as well as scenario IV and V (graph B and D).

4.3.1. Result and discussion

4.3.1.1. Catchment response to reduced nitrogen input

The catchment response to reduced nitrogen input was evaluated by comparing the estimated leaching reduction with the corresponding yield decrease. Thus, the results from the Scenarios I is compared with the results from the original basic model setup, and the relative (relative to the level in the basic setup) leaching and harvest yield are illustrated in Figure 43B. Moreover, the actual leaching rates are illustrated in Figure 43A.

The estimated leaching response to reduced nitrogen input varied considerably among the different soil types. The loamy soil was characterised by low leaching rate (9 kg N/ha·y) and a fertiliser nitrogen application that – more or less – was matched by the crop uptake. Thus, reducing the fertiliser input effected the harvest yield, but had only minor effect on the nitrogen leaching. The leaching reduction was only 1 and 2 kg N/ha for the silty loam and sandy loam respectively (Figure 43A). On the other hand the sandy soil was characterised by a somewhat higher leaching rate (26 kg N/ha·y) and an excess application of nitrogen fertiliser. Reducing the fertiliser input therefore had a huge impact on the nitrogen leaching, and a total leaching reduction of 31% (~ 9 kg N/ha·y) could be obtained with a yield decrease of 15%(Figure 43A B).

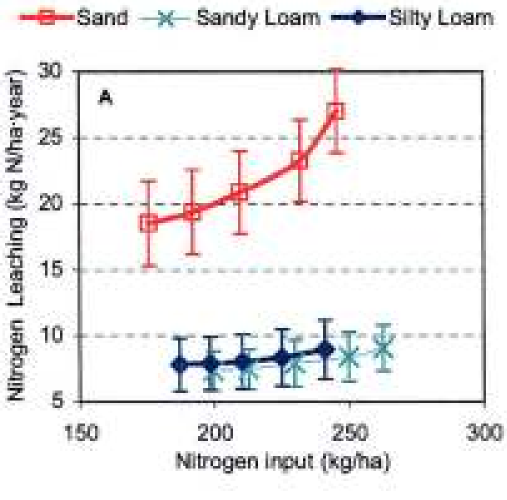

Figure 43A. Nitrogen leaching as a function of nitrogen input (scenario I). The nitrogen input comprises fertiliser application, atmospheric deposition and nitrogen fixation. The values are averaged over the various soil types and error bars indicate one standard deviation.

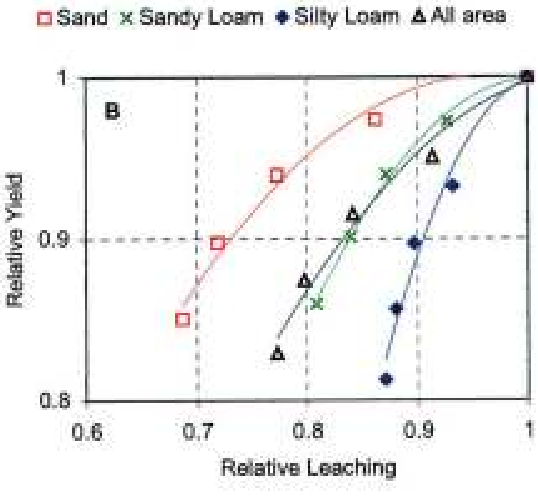

Figure 43B. Relative harvest yield and nitrogen leaching for scenarios I. The values are averaged over the various soil types and all values are expressed relatively to the level in the basic model setup

The harvest yield as well as the yield response of maize varied remarkably among years (Figure 44A & B). For instance 1996 was characterised by a low yield irrespectively of the fertiliser input, whereas high yield and large yield response for fertiliser reduction was characterising the growing season of 1995. Nevertheless, the modelling results did indicate, that the traditional fertiliser level could be reduced to the maximum limit of 200 kg N/ha without substantial yield reduction (See Figure 44C). A further reduction below the 200 kg N/ha did however influence the harvest yield. This effect is especially pronounced at the sandy soil, where the fertilisation rate could be reduced 30% with a minimum yield reduction of only 6% but with a considerable leaching reduction of 27%.

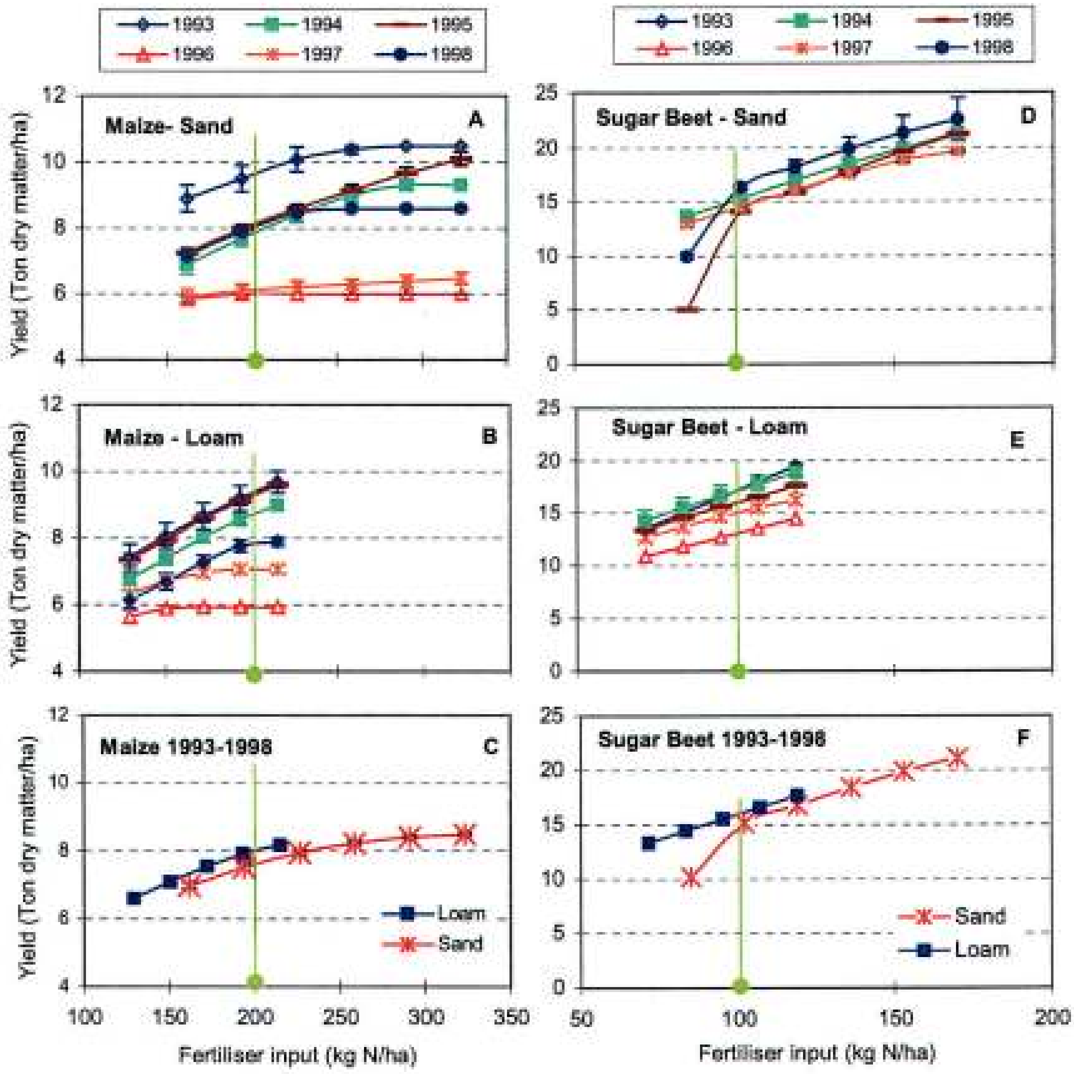

Figure 44. Yield of maize(left) and sugar beet (right) as a function of fertiliser input. Error bars indicate one standard deviation, and green lines indicate the maximum fertiliser level according to EC regulation 1257/1999. The yield response for each of the simulation years is illustrated in the two upper graphs (A,B,D,E) whereas the integrated effect for the six years simulation period is illustrated in the two lower graph.

The yield of sugar beet was found to be more sensitive to fertiliser reduction than maize. Even a small reduction in the fertiliser input reduced the yield of sugar beet, indicating that nitrogen is the limiting factor for the crop growth (Figure 44D and Figure 44E).

The modelling result also indicated that reducing the fertiliser input for sugar beet had no impact on nitrogen leaching, but only influenced the harvest yield. When the fertiliser reduction comprised both maize and sugar beet a 23% reduction of the nitrogen leaching could be obtained with a decreased yield of 17% (Figure 45A). The same 23% leaching reduction could however be obtained with a decreased yield of only 10% by concentrating the fertiliser reduction in the maize field only(Figure 45B).

It should however be noted that the model output in terms of harvest yield for sugar beet refers to the dry matter beet production and not to the amount of extractable sugar, which is the prior harvest product for sugar beet. For sugar beet the extractable amount of sugar is not inevitable linked to high N uptake and beet growth (Ulrich and Hills, 1973; Koch and Wendenburg, 1996). Consequently, the observed decrease in dry matter yield is not necessarily associated with a decreased sugar yield.

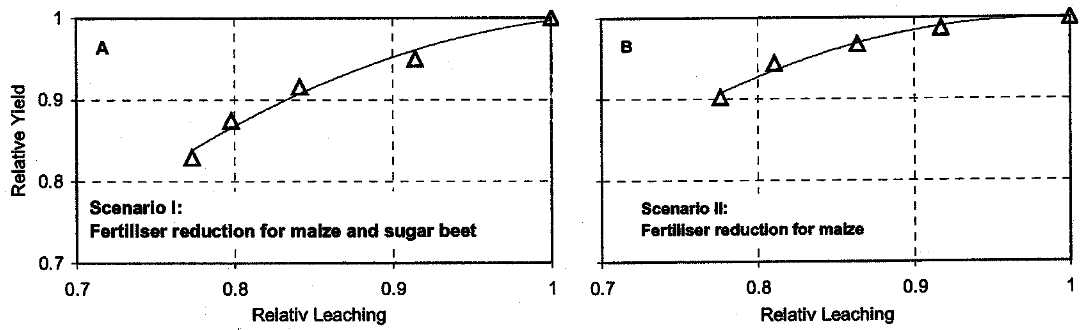

Figure 45A. Relative yield and nitrogen leaching for Scenarios I, where fertiliser reduction comprised both maize and sugar beet. All values are expressed relatively to the level in the basic model setup.

Figure 45B. Relative yield and nitrogen leaching for Scenarios II, where fertiliser reduction comprised only the maize fields. All values are expressed as relatively to the level in the basic model setup.

The yield and leaching response to fertiliser treatment is highly site specific, due to its dependence of the soil nitrogen concentration at the onset of the growing season, climate, nitrogen input from mineralisation influence, etc. etc. Nevertheless, the yield response for maize was somewhat in accordance with the results of Spallacci et. al. (1997) and Ceotto et. al. (1996), who investigated the yield response of maize to various fertiliser treatment on a silt-clay soil in the Low Po Valley. In Spallacci et. al. (1997) maize fields were treated with various combination of pig slurry and inorganic fertiliser. Although yield response varied remarkably among years and fertiliser type Spallacci et. al. (1997) found that increasing the fertiliser above 225 kg N/ha had no large impact on the yield (see Figure 46.).

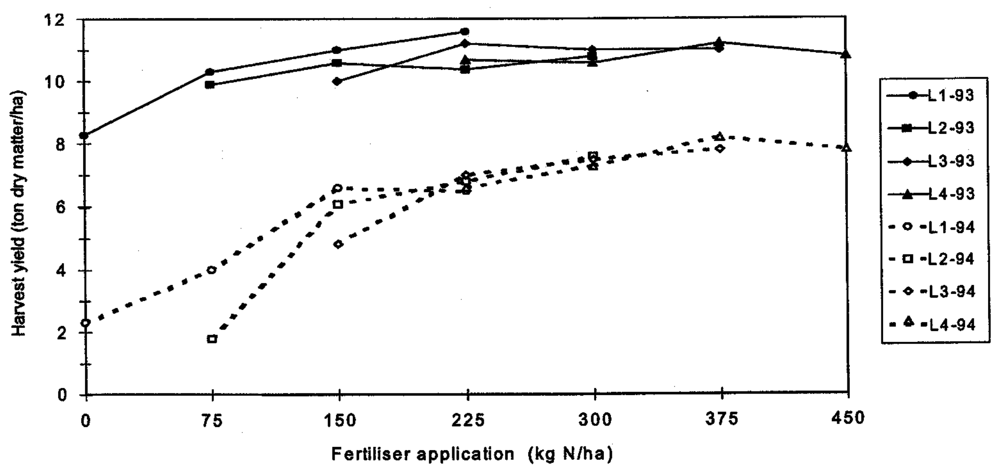

Figure 46. Yield of maize as a function of increased application of nitrogen from various combinations of pig slurry and inorganic fertiliser. The amount of pig slurry in L1, L2, L3 and L4 was 0, 75, 150, 225 kg N/ha respectively. The results derived from the cropping season 1993 and 1994 at S. Prospero (MO) in northern Italy. (Reproduced with permission from Spallacci et. al. 1997)

Ceotto et. al. (1996) applied three different fertiliser treatments (A,B and C) to five different cropping sequences containing soya, maize, sugar beet, barley and wheat (Table 27.). Reducing the fertiliser input from the traditional level of 370 kg N/ha to the reduced/minimum level of 325 kg N/ha had no impact on the harvest yield of maize, whereas a further reduction to 80 kg N/ha caused significant yield decrease. On the other hand the fertiliser treatment had no significant impact on the yield of Sugar beet (Ceotto et. al. 1996). These latter results differ from this analysis, where the fertiliser input did influence the yield of sugar beet.

Table 27. Average yield for maize and Sugar Beet for three different fertiliser treatment. The fertiliser treatment A, B and C characterised the traditional, reduced/alternative and minimum fertiliser application respectively (Reproduced with permission from Ceotto et. al. 1996).

	Maize			*Sugar Beet*		
	A	*B*	*C*	*A*	*B*	*C*
Pig Slurry (kg N/ha)	250	250	0	0	0	0
Inorganic fertiliser (kg N/ha)	120	75	80	80	40	0
Total nitrogen (kg N/ha)	370	325	80	80	40	0
Harvest yield (ton dry matter/ha)	9.15	9.17	7.64	15.67	15.71	16.40

Finally, Vachaud et. al. (1996) concluded that the traditional fertiliser input of 250 kg N/ha·y could be reduced nearly 30% without any substantial yield reduction, but with considerable reduction of nitrate leaching. His findings were based upon a field experiments with ^{15}N- on irrigated maize crops in Southern France (Grenoble).

4.3.2.3. Implementation of EC regulation 1257/1999 and groundwater regulation

The results indicated that the traditional fertiliser level could be reduced to the maximum limit according to EC regulation 1257/1999 with an average yield decrease of 9%, but with a average leaching reduction of 18% (~ 3 kg N/ha) (See Figure 47A.). By far the largest part of the leaching reduction (85%) derived from the sandy soil, and was due to a considerable excess application of nitrogen fertiliser for maize (see Figure 44). At the sandy soils a considerable leaching reduction of 30% (~ 8 kg N/ha) could be obtained with a corresponding yield decrease of 13%. On the other hand the fertiliser level at the loamy soil was only slightly higher than the maximum limit. Reducing the level to the maximum limit did not have an added effect on either the nitrogen leaching nor the yield (Figure 47A).

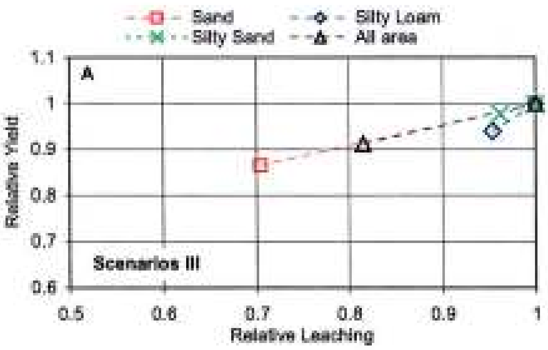

Figure 47A. Relative yield and nitrogen leaching for scenarios III, where the fertiliser level was reduced according to the EC regulation 1257/1999. Fertiliser rates of 100 kg N/ha and 200 kg N/ha were applied to sugar beet and maize respectively. All values are expressed relatively to the level in the basic model setup.

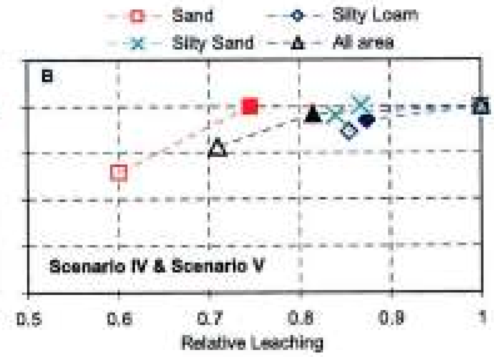

Figure 47B. Relative yield and nitrogen leaching for scenarios VI and V. Closed point illustrate the results from scenarios IV (raised groundwater table), whereas the open points illustrate the results from scenarios V (combination of fertiliser reduction according to EC regulation 1257/1999 and a raised groundwater table).

This difference between the various soil types is moreover illustrated with a comparison between Figure 48A and Figure 48B. A high leaching reduction is observed in the sandy soil covering the western part of the study area. Whereas no notable leaching reduction is observed in the loamy soil covering the eastern part of the study area.

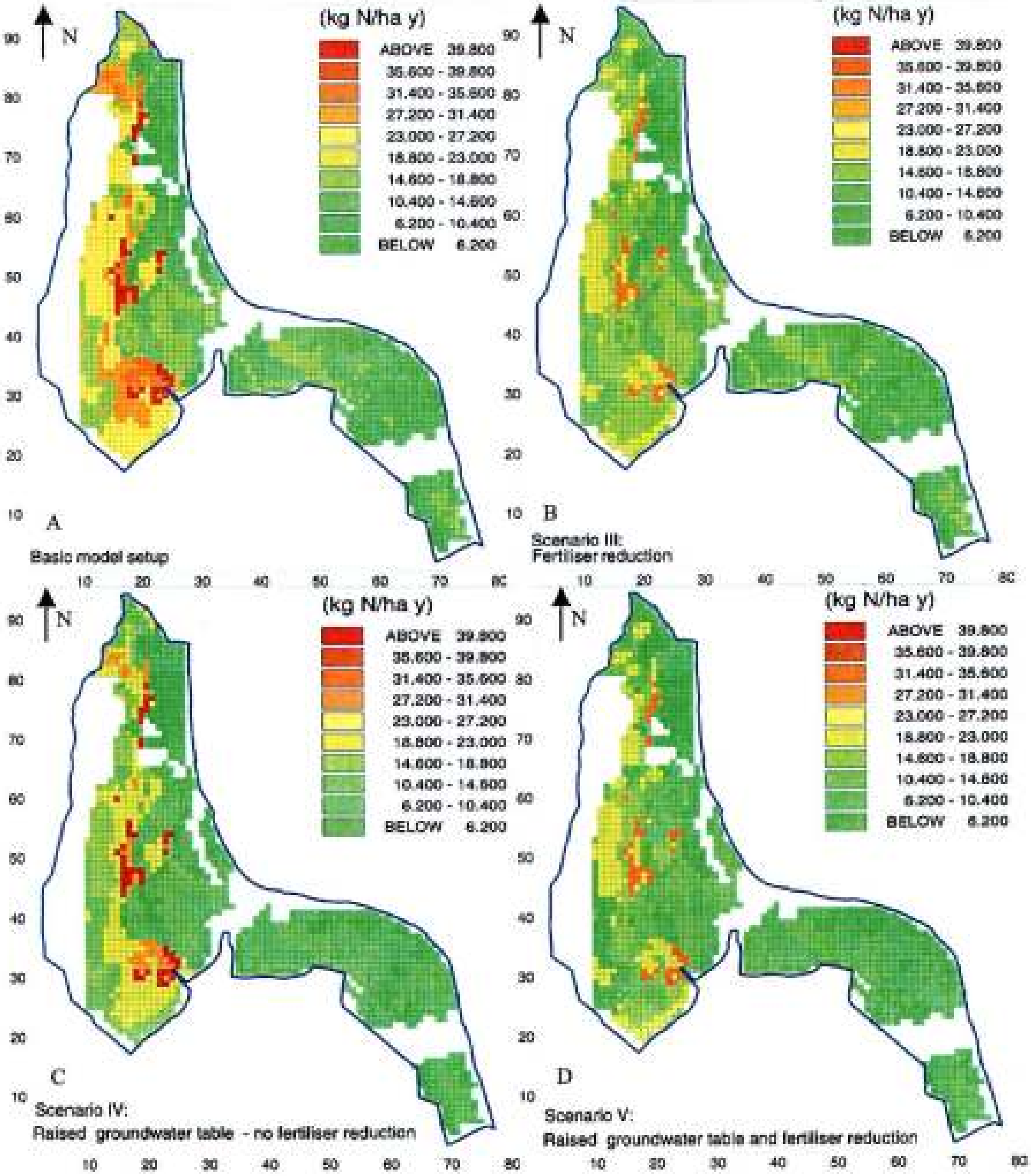

Figure 48. Estimated nitrogen leaching averaged over the simulation period of 1993-1998 (kg N/ha·y). Basic model setup describes the actual situation in the study area. Scenario III: Fertiliser reduction according to EC regulation 1257/1999. Scenarios IV: Decreased depth to the groundwater table. Scenario V: Combination of Scenario III and Scenario IV. Decreased depth to groundwater table and fertiliser reduced to the maximum limit according to EC regulation 1257/1999. Scale: 1:10.000.

Raising the groundwater table causes a higher evapotranspiration, lower percolation as well as a higher denitrification (See Table 28.). All these factors lead to a decrease of the nitrogen leaching. The modelling results moreover indicated that an average leaching reduction of 19% could be obtained by without any yield decrease (Figure 47B. & Table 28.). The leaching response to a raised groundwater table varied among the different soil types. The highest effect was found at the sandy soil in terms of a leaching reduction of 25% (~7 kg N/ha), whereas a minor leaching reduction of 10-12% (~ 1-2 kg N/ha) was found at the loamy soils (See Figure 48C)

Table 28. Nitrogen balance for the simulation area. The results derive from the basic model setup as well as scenarios III, IV and V. Values are averaged over the six years simulation period 1993-1998.

	Basic setup	*Scenario III*	*Scenario IV*	*Scenario V*
Fertiliser input (kg N/ha·y)	*172*	*136*	*173*	*136*
Atmospheric input (kg N/ha·y)	*41*	*41*	*41*	*41*
Net Mineralisation (kg N/ha·y)	*118*	*114*	*118*	*114*
Denitrification (kg N/ha·y)	*66*	*60*	*69*	*63*
Plant Uptake (kg N/ha·y)	*221*	*200*	*220*	*200*
Leaching (kg N/ha·y)	*16*	*13*	*13*	*11*
Worse case Leaching (kg N/ha·y)	*27*	*23*	*24*	*21*
Volatilisation (kg N/ha·y)	*29*	*22*	*29*	*22*
Changes in N-pool(kg N/ha·y)	*0*	*-5*	*2*	*-4*
Average Yield (ton dry matter/ha·y)	*10.5*	*9.6*	*10.5*	*9.6*
Percolation (mm/y)	*144*	*145*	*131*	*131*
Total leaching (ton N/y)	*26*	*21*	*21*	*18*

Scenario III: Fertiliser reduction according to EC regulation 1257/1999
Scenarios IV: Decreased depth to the groundwater table.
Scenario V: Combination of Scenario IV and Scenario IIIA. Decreased depth to groundwater table and fertiliser reduced to the maximum limit according to EC regulation 1257/1999.

The observed leaching reduction could be further enhanced if the groundwater regulation was combined with a fertiliser reduction. For all the simulation area, the nitrogen leaching could be reduced by 29% (~ 3 kg N/ha), with a minimum yield decrease of only 7% (See Table 3. Figure 47B.). Again this effect is especially pronounced at the sandy soil where a leaching reduction of 40% (~ 11 kg N/ha) could be obtained with a yield decrease of 13%.

4.3.3 Summary and concluding remarks

The impact of alternative management practices on nitrogen leaching and harvest yield varied considerable among the different soil types within the study area.

At the sandy soils a substantial leaching reduction could be obtained by the implementing the maximum fertiliser application rates set out by the EC regulation 1257/1999. Hence, a leaching reduction of 31% (~ 9 kg N/ha·y) could be obtained with a corresponding yield decrease of 15%. A leaching reduction associated with a considerable excess application of nitrogen fertiliser at the maize field. The modelling result indicated that the traditional fertilisation rate at the maize field can be reduced 30% with an average yield reduction of only 6% but with a considerable leaching reduction of 27%.

At the loamy soils a fertiliser reduction was found not to be worthwhile, as it affected only the yield without any notable leaching reduction. Hence, the modelling results indicated that an appropriate fertiliser level was applied at the loamy soils. The application rate was only slightly higher than the maximum limit and the applied nitrogen was more or less matched by the crop uptake.

By raising the groundwater table in the arable land an overall leaching reduction of 19% could be obtained without any yield decrease. If the groundwater regulation was combined with a fertiliser reduction the leaching reduction of 29% could be obtained with a minimum yield decrease of only 7%. Again this effect is especially pronounced at the sandy soil where a leaching reduction of 40% (~ 11 kg N/ha) could be obtained with a yield decrease of 13%.

5. Conclusions

The MIKESHE/DAISY modelling system was successfully calibrated and validated for the Bonello catchment, providing a satisfactory description of the overall hydrological processes and of the nitrogen transformation in the root zone.

The present work has contributed toward quantifying the extent of nitrogen leaching associated with current agricultural practices in the Bonello catchment. The estimated nitrogen leaching were relatively low ranging from 9 kg N/ha·y at the loamy soils to 26 kg N/ha·y at the sandy soils. An important factor that contributed toward the limited leaching is the high denitrification characterising the study area. Indeed, the estimated denitrification was 66 kg N/ha·y accounting for 56% of the mineralised nitrogen. The high denitrification loss is due to the shallow groundwater table, which together with the strong soil capillary forces maintains the high soil water content in the root zone that allows denitrification to take place.

The impact of alternative management practices on nitrogen leaching and harvest yield was also estimated through scenario analysis. For the loamy soils, a fertiliser reduction was not worthwhile, as it affected only the yield without any notable leaching reduction. A fertiliser reduction is, however, advisable for the sandy soils. By implementing the maximum fertiliser application rates set out by the EC regulation 1257/1999 (EC, 1999) a leaching reduction of 31% could be obtained with a corresponding yield decrease of 15%. A leaching reduction associated with a considerable excess application of nitrogen fertiliser at the maize field. Local authorities are currently implementing an action program (EC, 1999) aiming at reducing pollution deriving from agricultural sources. A number of additional scenarios, related to such an action programme, could be studied using the modelling system developed here. Of relevance would be an analysis of the impact of long-term set aside of arable land and the introduction of winter cover crop.

It has been shown that groundwater regulation has a positive impact on the nitrogen leaching. By raising the groundwater table in the arable land an overall leaching reduction of 19% could be obtained without any yield decrease. These results focus particularly on the impact of groundwater regulation on nitrogen leaching. An issue worthwhile taking into account when discussing the impact of alternative management practices on nitrogen leaching. However, groundwater regulation in the area has to make a balance between the needs of a groundwater table high enough for agricultural purposes and low enough to ensure a minimum risk of flooding. Additional analysis focusing on the impact of groundwater regulation on flooding risk would therefore be of relevance.

Model limitations were also identified for the DAISY model. These were mainly related to an inadequate description of the urea hydrolysis as well as of the water and nitrogen transport through macro pores. In this context it has to be mentioned that during the time of this project a new release allowing preferential flow to take place has been developed.

Finally, this work has proposed an upscaling procedure for the one-dimensional model DAISY to represent condition at the large catchment scale. In this context, model uncertainties were identified and the influence of selected catchment characteristics on nitrogen leaching was evaluated. This provides the basis for a further upscaling to the drainage basin of the Sacca di Goro lagoon. Indeed, many of the characteristics (soil type, hydrological conditions, agricultural and irrigation management practices) of the investigated sub-catchment are representative for the whole drainage basin. With an increase of scale, the required input data on agricultural practices have often to be based upon best estimates of general management practices as well as statistical data. Similarly, hydraulic and chemical properties of soils have to be assessed from existing databases if further refinement through in-situ campaigns is not possible. In general, uncertainties associated with these input data increase with increasing scale of model application. A thorough sensitivity analysis evaluating the influence of these uncertainties on model output would then be important to attach confidence to impact estimates suggested by scenario analyses.

6. References

Abbott, M. B., Bathurst, J. C., Cunge, J. A., O'Connell, P. E., and Rasmussen, J. (1986): An introduction to the European hydrological system - Systeme Hydrologique Europeen, "SHE",: Structure of a physically-based, distributed modelling system, *Journal of Hydrology; 87; 61-77.*

Andreotti, M. (1998): Le Produzioni Ecocompatilili in Provincia di Ferrara, *Lo stato dell'ambiente nella Provenzia di Ferrara Anno 1997, n. 1/Aprile 1998, Servizio Ambiente, Provencia di Ferrara, 91-101.*

Angulo-Jaramillo, R., Peugeot, C., and Gaudet, J. (1996): Experimental and numerical study on three-dimensional (3D) transport of water and solute in unsaturated soil media, Rapport No.2: Development of experimental studies concerning laboratory soil monolith facility (Cube), *Collaboration contract UJF-Grenoble and Environment Institute, JRC-Ispra, No 10356-94-07 F2ED ISP F; Internal report, 18 p, Grenoble 1996.*

Bartoli, M., Castaldelli, G., Nizzoli, D., Benetti, M., Pigoni, M., and Viaroli, P. (1997): Interazioni tra cominità microbiche e vegetali nella regolazione dei flussi dell'azoto in quattro ambienti costieri, *Società Italiana di Ecologia - Atti, 18, 505-508.*

Bartoli, M., Cattadori, M., Giordani, G., and Viaroli, P. (1996): Benthic oxygen respiration, ammonium and phosphorus regeneration in surficial sediments of the Sacca di Goro (Northen Italy) and two french coastal lagoons:a comparative study, *Hydrobiologia, 329, 143-159.*

Bencivelli, S. (1990) L'attivita della Provincia per il risanamento della Sacco di Goro, *In Bencivelli, S. and Castaldi, N. (eds.): Il piano di risanimento delle acque del bacino idrografico Burana-Po di Volano, 96-106, Franco Angeli Libir s.r.l.*

Bencievelli, S. (1998): La Sacca di Goro: La Situatione di Emergenza dell'estate 1997, *Lo stato dell'ambiente nella Provenzia di Ferrara Anno 1997, n. 1/Aprile 1998, Servizio Ambiente, Provencia di Ferrara, 61-66.*

Beven , K. J. (1991): "Scale Considerations", in Bolwer, D. S. and O'Connell, P. E. (Eds.): Recent Advances in the Modeling of Hydrologic Systems, Kluwer, Dordecht, 357-371.

Beven K. (1995): Linking parameter across scales: Subgrid parameterization and scale dependent hydrological models, *in: Kalma, J. D. and Sivapalan, M (eds.): Scale issues in hydrological modelling, John Wiley & Son, West Sussex, England.*

Beven , K. J. (1996): A discussion of distributed hydrological modelling, *in Abbott, M. B. & Refsgaard, J. C. (eds.): Distributed hydrological modelling, Kluwer Academic Publishers,255-278.*

Beven, K. J. and Fisher, J. (1996): Remote sensing and scaling in hydrology, *in Stewart, J. B., Engman, E. T., Feddes, R. A. and Kerr, Y. (eds.): Scaling up in Hydrology using Remote Sensing, John Wiley & Sons, 1-18.*

Blöschl, G. and Sivapalan (1995): Scale issues in Hydrological Modelling: A review, *in Kalma, J. D. and Sivpalan, M. (Eds.): Scale Issues in Hydrological Modelling, 9-48 John Wiley & Sons.*

Brodile, K. W. (1980): A reveiw of methods for curve and function drawings, *in: Butland, J. (ed): Mathematical Methods in Computer Graphics and Design, Academic Press, London.*

Brooks, R. H. and Corey, A. T. (1964): Hydraulic properties of porous media, *Hydrology Paper 3. Colorado State University, Fort Collins, 27 pp.*

Ceotto, E., Donatelli, M., Spallacci, P. and Casteli, F (1994): CROPSYST validation for soil content of water and nitrate at two location of Northen Italy, Proceedings of the third ESA congress, Abano-Padova 1994.

Ceotto, E., Donatelli, M., Marchetti and Spallacci, P. (1996): Confronto tra sistemi cerealico-industriali fertilizzati con concimi minerali e liquami suini nella bassa Pianura Padana, *Acricultura Ricerca, numero 164-165-166, 201-208.*

Ceotto, E., Donatelli, M. (1997) Simulazioni di Sistemi Colturali nella bassa Pianura Padana con il modello EPIC, *Acricultura Ricerca, 171, 59-66.*

Christian, R. R., Naldi, M. and Viaroli, P. (1998): Construction and Analysis of Static, Structured Models of Nitrogen Cycling in Coastal Ecosystems, *In Koch, A. L. (ed): Mathematical modelling in microbial ecology, Chapman & Hall, New York 1998.*

Clay, D. E., Malzer, G. L. and Anderson, J. L. (1990): Ammonia Volatilization from Urea as influenced by soil Temperature, Soil Water Content and Nitrification and Hydrolysis Inhibitors, *Soil Sci. Soc. Am. J., 54, 1, 263-266.*

de Vries, D. A. (1963): Thermal Properties of Soils, *Physics of Plant Environment, Published by North Holland Publishing, Amsterdam 1963.*

DeCoursey, D. G. Ahuja, L. R., Hanson, J., Shaffer, M., Nash, R., Rojas, K.W., Hebson, C., Hodges, T., Johnson, K. E., Ma. Q. and Ghidey, F. (1992): Root Zone water quality model (version 1.0), Technical Documentation, *United States Department of Agriculture, Agricultural Research Service, Great Plains Systems Research Units, Fort Collins, Colorado, USA.*

Denmead, O. T. (1983): Micrometeological Methods for Measuring Gaseous Losses of Nitrogen in the Field, *in Freney, J. R. and Simpson, J. R. (eds): Gaseous Loss of Nitrogen from Plant-Soil Systems, Published by Martinus Nijhoff/ Junk, W., The Hague 1983, 131-157.*

de Willigen, P. (1991): Nitrogen turnover in the soil-crop-systems: Comparision of fourteen simulation models, *Fertiliser Research, 27, 141- 149.*

DHI (1993): MIKESHE user guide (Release 1.0), *Danish Hydraulic Institute, August 199.*

Dyhr-Nielsen, M., Hansen, M., Holter, E. M., Krag-Andersen, K., and Iversen, T. M. (1991): Kvælstof og fosfor i jord og vand. Transport, omsætning og effect. *NPO-forskning fra Milj¢styrelsen, Samplerapport, Milj¢styrelsen, Copenhagen.*

Diez, J. A., Roman, R., Caballero, R., and Caballero, A. (1997): Nitrate leaching from soils under a maize-wheat-maize sequence, two irrigation schedules and three types of fertilisers, *Agriculture, Ecosystems and Environment, 65, 189-199.*

Diekkrüger, B., Sondergerath, D., Kersebaum, K. C., and McVoy, C. W. (1995): Validation of agroecosystem models A comparision of results of different models applied to the same data set, *Ecological Modelling, 81, 3-29.*

Djurhuus, J. (1992): N-transformation and N-transport in a Sandy Loam and a Coarse Sandy Soil Cropped with Spring Barley, *Danish Journal of Plant and Soil Science, Beretning nr. 2214-1992, 137-152.*

Djurhuus, J., Hansen, S., Schelde, K., and Jacobsen, O. H. (1999): Modelling mean nitrate leaching from spatially variable fields using effective hydraulic parameters, *Geoderma, 87, 261-279.*

EC (1991): Concerning the Protection of Water against Pollution Caused by Nitrates from Agricultural Sources, *Council Directive of 31 December 1991, 91/676/EEC, Off. J. Eur. Commun, 375, 1-8.*

EC (1996): Commission proposal for and Action programme for Integrated Groundwater Protection and Management, *Community preparatory acts, Document 596PC0315.*

EC (1997): The implementation of Council Directive 91/676/ concerning the Protection of Waters against Pollution caused by Nitrates from Agricultural Sources, *Report COM(97) 473, 12 p.*

EC (1999): Commission Regulation (EC) No 1750/1999 of 23 July Laying down detailed rules for the application of Council Regulation (EC) No 1257/1999 on support for Rural development from the European Agricultural Guidance and Guarantee Fund (EAGGF), *Off. J. Eur. Commun, 214, 31- 35.*

EEA (1995): Europe's Environment, The Dobriss Assessment, *Eds: Stanners, D. and Bourdeus, P., European Environment Agency (EEA), Copenhagen.*

EEA (1997): Water stress in Europe - Can the challenge be met?, Environmental Information, *European Environment Agency (EEA), Copenhagen, also available on www.themes.eea.eu.it/toc.php/state/water.*

EEA (1998): Europe's Environment: The Second Assesment, *eds: Weiringa, K, European Environment Agency (EEA), Copenhagen.*

EEA (1999): Groundwater quality and quantity in Europe, Data and basic information, *Techical report No 20, European Environment Agency (EEA), Copenhagen.*

EEA (2000): Environmental signals 2000, European Environment Agency Regular indicator report, *Environmental assessment report No 6., European Environment Agency (EEA), Copenhagen, also available on www.themes.eea.eu.it/toc.php/state/water.*

Filippi, N. and Sbarbati, L. (1994): I Suoli dell'Emilia-Romangna, *Servizio Cartografico – Ufficio Pedologico Regione Emilia Romagna.*

Fox, R. H., Piekielek, W. P., and Macneal, K. E. (1996): Estimating Ammonia Volatilization Losses from Urea Fertilizers Using a Simplified Micrometeorological Sampler, *Soil Science Society of America Journal, 60, 596-601.*

Franco, U., Oelschlägel, B. and Schenk, S. (1995) Simulation of temperature, water and nitrogen dynamics using the model CANDY, *Ecological Modelling, 8, 213-222.*

Freijer, H. and Van der Linden (1996): SOTRAS report 5 PESTRAS 3.1. A one-dimensional model for the assessment of the fate of pesticides in soil, *Report n° 715501007 RIVM Bilthoven Netherlands.*

Freney, J. R., Simpson, J. R., and Denmead, O. T. (1981): Ammonia Volatilization, *in: Clark, F.E. and Rosswall, T. (eds): Terrestrial Nitrogen Cycles,Ecol. Bull. (Stockholm), 33, 391-302.*

Gezgin, S., and Bayrakl, F. (1995): Ammonia Volatilization from Ammonium Sulphate, Ammonium Nitrate and Urea Surface Applied to Winter Wheat on Calcarous Soil, *Journal of Plant Nutrition, 18, 11, 2483-2494.*

Grasshoff, K., and Johansen, H. (1972): A new sensitive and direct method for determination of ammonia in sea water, *J.Cons.Perm, int. Explor. Mer., 34, 516-521.*

Hansen, S., Jensen, H. E., Nielsen, N. E., and Svendsen, H. (1990): DAISY A Soil Plant Atmosphere System Model, *NPO Report A10, The National Agency for Environmental Protection, Copenhagen, Denmark.*

Hansen, S., Jensen, H. E., Nielsen, N. E., and Svendsen, H. (1993): Description of the Soil Plant System Model DAISY, Basic Principles and Modelling Approach, *Jordbrugsforlaget, The Royal Veterinary and Agricultural University Copenhagen 1993.*

Hansen, S., Nielsen, N. E., and Svendsen, H. (1991): Simulation of nitrogen dynamics and biomass production in winter wheat using the Danish simulation model DAISY, *Fertilizer Research, 27, 245-259.*

Hargrove, W. L. (1988): Evaluation of Ammonia Volatilization in the Field, *J. Prod. Agric, 1, 2, 104-111.*

Heidmann, T., Hutchings, N. J., Jensen, N. H., and Østergaard, H. S. (1999): Effect of Input Data Quality on Simulation Results from the DAISY Model, *in Feyen, J. and Kiyo, K. (eds.): Modelling the transport processes in soils at various scales in time and space, International workshop of EuAgEng's field of interest on soil and water, 24-26 Nov. 1999, Leuven, 553 – 562.*

Heng, H. H. and Nikolaidis, N. P. (1998): Modeling of Nonpoint Source Pollution of Nitrogen at the Watershed Scale, Journal of the American Water Resources Association, 34, 2, 259-374.

Jayatilaka, C. J., Storm, B., and Mudgway, L. B. (1998): Simulation of water flow on irrigation bay scale with MIKESHE, *Journal of Hydrology, 208, 108-130.*

Jensen, C., Stougaard, B., and Olsen, P. (1994): Simulation of water and nitrogen dynamics at three Danish locations by use of the DAISY model, *Acta.Agric. Scand., Sect. B., Soil and Plant Sci., 44, 75-83.*

Jensen, C., Stougaard, B., and ¥stergaard, H. S. (1994a): Simulation of nitrogen dynamics in farmland areas of Denmark, *Soil Use and Managment, 10, 3, 111-118.*

Jensen, L. S., Mueller, T., Nielsen, N. E., Hansen, S., Grocker, G. J., Grace, P. R., Klìr, J., Körschens, M., and Poulton, P. R. (1997): Simulating trends in soil organic carbon in long-term experiments using the soil-plant-atmosphere model DAISY, *Geoderma, 81, 5-28.*

Kaiser, C., Zalidis, G., and Gerakis, A. (1997): Hydrological modeling of the Karla watershed to analyse groundwater depletion and restore pre-existing wetland functions, *in Refgaard, J. C. andKaralis, E. A. (eds.): Operational Water Management, Proceedings of the European Water Resources Association Conference, September 1997, Copenhagen, Publ. by Balkema, Rotterdam, 325-332.*

Kersebaum, K. C. (1995) Application of a simple management model to simulate water and nitrogen dynamics, *Ecological Modelling, 81, 145-156.*

Koch, H. J. and Wendenburg, C. (1996) How little nitrogen need sugar beet – plant requirement as a basis of fertilizer recommendation, *Transactions of the 9th Nitrogen Workshop, Techenische Universität Braunschweig, September 1996, 91-94.*

Kristensen, K. J., and Jensen, S. E. (1975): A model for Estimating Actual Evapotranspiration from Potential Evapotranspiration, *Nordic Hydrology, 6, 70-88.*

Kunze, R. J., Uehara, G., and Graham, K. (1986): Factors Important in the Calculation of Hydraulic Conductivity, *Soil Sci. Soc. Amer.Proc, 32, 760-765.*

Leip (2000): Nitrous oxide (N_2O) emissions from a coastal catchment in the delta of the Po river: Measurement and modelling of fluxes from a Mediterranean Lagoon and Agricultural soils, *Ph.D. thesis, Joint Research Centre, European Commission, EUR 19052 EN.*

Madsen, H. B. (1983): Himmerlands Jordbundsforhold: Et Regionalt studie omhandlende jordbundsudvikling, -klassifikation, afgrøders rodudvikling og jordens plantetilgængeligt vandmængde, *Folia Geographica Danica. TOM. XVI, C.A. Reitzels Forlag, Rosendahls Bogtrykkeri, K¢benhavn 1983 (in Danish).*

Massellani, M. (1984): Studio Geologica del Comune di Goro (1979-80), *Piano regolatore generale, Geotecnica, Documnetazione di Sintesi, Comune di Goro 1984.*

McInnes, K. J., Fergusosn, R. B., Kissel, D. E., and Kanemasu, E. T. (1986): Field measurements of ammonia loss from surface application of urea to bare soil, *Agron. J., 78, 192-196.*

Miljøstyrelsen (1987): Handlingsplan mod fourening af det danske vandmiljo med næringsalte, *Miljøstyrelsen (In Danish).*

Miljøstyrelsen (1998): Afdtale vedrørende Vanmiljoplan II, *available on* **www.mem.dk/publikationer/vandplan/aftale 2.htm**. *(In Danish).*

Ministerio, and delle Risorse Agriculture, A. e. F. (1994): Metodi Ufficiali di Analisi Chemica del Soulo, *Osservatorio Nationale Pedologico e per la Qualita del Suolo, Ministerio delle Risorse Agriculture, Alimentari e Forestali, Roma.*

Mosello, R. (1991): Situazione Degli Studi Sulla Chimica della Deposizioni Atmosferiche nel 1998 in Italia, *Rete Italiano per lo Studio delle Deposizioni Atmosferiche (RIDEP), Consiglio Italiano di Idrobiologia, Pallanza (Italy), 4.*

Mosello, R. (1993): Rapporto sull'attivita della rete ridep nel quinquennio 1998-1992, Rete Italiana per lo stidio delle deposizioni atmosferiche (RIDEP) *Consiglio Italiano di Idrobiologia, Pallanza (Italy), Pubblicazione n. 13.*

Nach, I. E. and Sutcliffe, J. V. (1970): River flow forecasting through conceptual models, *Journal of Hydrology, 10, 282-290.*

Neitsch, S. L., Arnold, J. G. Williams, J. R. (1999): Soil and Water Assessment Tool, User's Manual, version 98.1, *Grassland, Soil and Water Research Laboratory, Agricultural Research Service, 808 East Blackland Road, Temple, Texas.*

Page, A. L. (1982): Methods of Soil Analysis,Part 2, Chemical and Microbiological Properties, Second Edition, *Madison, Wisconsin USA.*

Piccapietra, L., Bidoglio, G., Elorza, F. J., and Rindone, B. (1998): Valutazione dell'influenza dell'uso di pesticidi sulla qualita di acque e di suoli in un bacino idrografico costiero, *EUR 18052 IT, Environment Institute, Joint Research Centre, Ispra Italy.*

Rachhpal-Singh, and Nye, P. H. (1988): Processes Controlling Ammonia Losses from Fertilizer Urea, *in Jenkinson, D. S. and Smith, K. A. (Eds): Nitrogen Efficiency in Agricultural Soils, Proceedings of the Seminar "Nitrogen Efficiency in Agricultural Soils and the Efficient Use of Fertilizer Nitrogen", Edingburgh, 16-18 Sep 1987, Published by Elsevier Applied Science, London & New York, 246-255.*

Ragab, R., Cooper, D. M., Harris, G. L., and Catt, J. A. (1996): Simulating nitrate leaching under winter wheat grown on a structured clay soil considering bypass flow, *Journal of Hydrology, 182, 157-173.*

Rauschkob, R. S. and Hornsby, A. (1994): Nitrogen Management in Irrigated Agriculture, *Oxford University Press, New York – Oxford.*

Rawls, W. J., and Brakensiek, D. L. (1989): Estimation of Soil Water Retention and Hydraulic Properties, *in Morel-Seytoux, H. J. (ed): Unsaturated Flow in Hydrological Modeling Theory and Practice, NATO ASI Series, Series C: Mathematical and Physical Sciences, 275, 275-300.*

Refsgaard, J. C., and Storm, B. (1995): MIKESHE, *In: V. P. Singh (Ed) Computer Models of Watershed Hydrology, Water Resources Publications, Colorado, USA, 809-846.*

Refsgaard, J. C., Storm, B and Abbott, M. B. (1996): Comment on "a Discussion of distributed hydrological modelling " by K. Beven, *in Abbott, M. B. and Refsgaard, J. C. (eds.): Distributed hydrological modelling, Kluwer Academic Publishers, 41-54.*

Refsgaard, J. C. (1997): Parameterisation, calibration and validation of distributed hydrological models, *Journal of Hydrology, 198, 69-97.*

Refsgaard, J. C., Sørensen, H. R., Mucha, I., Rodak, D., Hlavaty, Z.,Bansky, L., Klucovska, J., Takac, J., Kosc, V.,Enggrob, H. G.,Engesgaard, P., Jensen, J. K., Fiselier, J., Griffioen, J. and Hansen, S. (1998): An integrated Model for the Danubian Lowland – Methodology and Applications, Water Resources Management, 12, 433-465.

Refsgaard, J. C., Thorsen, M., Jensen, J. B., Kleeschulte, S. and Hansen, S. (1999): Large scale modelling of groundwater contamination from nitrate leaching, *Journal of Hydrology, 221, 117-140.*

Regione Emilia-Romagna (2000): Piano Regionale di Sviluppo Rurale 2000-2006, Regione Emilia-Romagna, Reg. (CEE) 1257/99, *Disciplinari di Produzione, Dell'azione Produzione Integrata (Azione 1 – Misure 2F del Sottoasse Agroambiente – Asse 2 del PRSR 2000-2006, in press.*

Rinaldi, M. and Ventrella, D. (1997): Uso del modelli Epic e CROPSYST in stistemi colturali del Sud Italia, *Agricultura Ricerca, 171, 47-58.*

Ryden, J. C. (1984): The Flow of Nitrogen in Grassland, *Proc. Fert. Soc., 229, Fert. Soc, London 1984.*

Schachtschabel, P., Blume, H. P., Brümmer G., Hartge, K. H. and Schwertmann, U. (1992): Lehrbuch der Bodenkunde, *Ferdinand Enke Verlag, Stuttgart, 1992.*

Shepard, F. P. (1954): Nomenclature based on sand-silt-clay ratios, *Journal of Sedimentary Petrology, 24, 3, 151-158.*

Singh, R., Refgaard, J. C., Yde, L., J¢rgensen, G. H., and Thorsen, M. (1997): Hydraulic-hydrological Simulation of Canal-command for Irrigation Water Management, *Irrigation and Drainage System, 11, 185-213.*

Smith, R.E. and Diekrükker, B.(1996): Effective soil water characteristic and ensembled soil water profiles in hetrogeneous soils, *J. Geophys. Res., 32, 1993-2002.*

Spallacci, P., Marchetti, R., Papini, R. and Donatelli, M. (1997): Bilancio dell'azoto per la coltura di mais trattata con liquami suini e urea di sintesi, *Agricultura Ricerca, 168, 57-68.*

Spallacci, P, Ceotto, E., Donatelli, M: Papini, R: (1994): EPIC and CROPSYST evaluation in predicting corn growth, and soil water and nitrate content at two levels of N-fertilization, *Proceedings of the Third ESA congress, Abano-Padova, 1994, 412- 413.*

Stålnacke, P. (1996): Nutrient Loads to the Baltic Sea, *Ph.D Thesis, Department of Water and Environmental Studies, Linköping University, Sweden.*

Stockle, C. O., Martin, A. A. and Cambell, G. S. (1992): A model to assess environmental Impact of Cropping System, *Trans. of ASAE, Charlotte, North Carolina, USA, June 21-24, 1992.*

Stockmarr, J. (1997): Revised Groundwater Quality Monitoring Programme in Denmark 1998-2003, *In Refsgaard, J. C. and Karalis, E. A. (Eds): Operational Water Management, Proceedings of the European Water Resources Association Conference, Copenhagen, Denmark 3-6 Sep 1997, Published by A.A.Balkema Rotterdam Brookfield 1997, 49-55.*

Styczen, M., and Storm, B. (1993a): Modelling of N-movements on catchment scale - a tool for analysis and decision making. 1. Model description., *Fertilizer Research, 36, 1-6.*

Styczen, M., and Storm, B. (1993b): Modelling of N-movements on catchment scale - a tool for analysis and decision making. 2. A case study, *Fertilizer Research, 36, 7-17.*

Svendsen, H., Hansen, S., and Jensen, H. E. (1995): Simulation of crop production, water and nitrogen balances in two German agro-ecosystems using DAISY model, *Ecological Modelling, 81, 197-212.*

Swensen, B. and Singh, B. R. (1997): Transport Processes and Transformations of Urea and its derivatives through a mineral subsoil, *J. Environ. Qual., 26, 1516-1523*

Thorsen, M., Feyen, J. and Styczen, M. (1996): Agricultural Modelling, *in Abbott, M. B. and Refsgaard, J. C. (Eds.): Distributed hydrological Modelling, Kluwer Academic Publishers, 121-141.*

Tonderski, A. (1996): Control of Nutrient Fluxes in Large River Basins, *Ph.D. Thesis, Department of Water and Environmental Studies, Linköping University, Sweden.*

Thomasson, T., Bourma, J., and Leith, H. (1991): Soil and Groundwater Research Report II, Nitrate in Soil, *Contract nos EV4-0098-NL andEV4V-00107-C-AM, Commision of the European Communities.*

Ulrich, A. and Hills, F.J. (1973): Plant analysis as an aid in fertilizing crops: Part 1. Sugar Beets, *In Dinauer, R. C. (eds): Soil Testing and Plant Analysis, Soil, Sci.Soc.Amer., Madison, WI, 271-288.*

Vachaud, G., Mormand, B., Recous, S., Kegni, L., and Garino, B. (1996): In-situ Nitrogen Balance of a Irrigated Maize by Continuous Monitoring plus Sampling at Harvest, *Soil Sci. Doc. of Americal Journal, 1-30.*

Vanclooster, M., Viane, P., Christians, K. 1994: WAWE – a mathematical model for simulation agrochemicals in the soil and vadose environment, *Reference and user's manual (release 2.0) Institute for Land and Water Management, Katholieke University Leuven, Belgium.*

Vanclooster, M. Viane, P. Diehls, J. and Feyen, J. (1995): A deterministic evaluation analysis to an integrated soil-crop model, *Ecological Modelling, 81, 197-212.*

Van Genuchten, M. (1980): A Closed-form Relationship for Predicting the Hydraulic Conductivity of Unsaturated Soils, *Soil Sci. Soc. Am. J., 44, 892-898.*

Vereecken, H., Jansen, E. J., H., B., Swerts, M., Engelke, R., Fabrewitz, S., and Hansen, S. (1991): Comparision of simulation results of five nitrogen models using different data sets, *Soil and Ground Water Research Report II, Nitrate in Soil, Final Report of Contracts EV4V-0098-NL and EV4V-00107-C, Commission of the European Communities, 321-338.*

Viaroli, P. (1992): Eutrophication of the Po delta Lagoons: Evolution and prospects for restoration, *In Finlayson, M. (ed): Managing mediteranean wetlands and their birds, IWRB Special Publication No. 20.*

Viaroli, P., Bartoli, M., Naldi, C., and Ferrari, I. (1993): Macroalgae growth and decomposition and nutrient cycling in the Sacca di Goro (Po river Delta), *In Rodriguez-Velera, F. and Caumette, P. (eds): C.L.E.A.N. Coastal Lagoon Eutrophication and Anaerobic processes, Progress Report - 1993 - Part II. Proceedings of the CLEAN Meeting Alicante, 3-5 December 1993.*

Viaroli, P., Pugnetti, A., Naldi, M. and Zaccaria, V. (1994): Richeche sul filoplancton e su crescita e decomposizione di Ulva Ridiga nella Sacca di Goro, *In Bencivelli, S. Castaldi, N. and Finessi, D (eds.): Sacca di Goro: Studio intergrato sull'ecologia, 109-130.*

Viaroli P., Naldi M., Bondavalli C., and Bencivelli S. (1996) Growth of the seaweed Ulva rigida C. Agardh in relation to biomass densities, internal nutrient pools and external nutrient supply in the Sacca di Goro lagoon (Northern Italy), *Hydrobiologia 329, 93-103.*

Whitmore, A. P. (1995): Modelling the mineralisation and leaching of nitrogen from crop residues during three successive growing season, *Ecological Modelling, 81, 197-212.*

Williams, J. R., Jones, C. A., and Dyke, P. T. (1984): The EPIC Model and its Application, *in Proc. ICRISAT-IBSNAT-SYSS Symp on Minimum Data Sets for Agrotecnology Transfer, 1983, Hyderabad, India, 111-121.*

Evaporation method for analysing water retention curve

The following describes the evaporation method, which was used for measuring the water retention curve. The methods is based on the simultaneous measurement of capillary pressure and water content, as a function of time. The method characterises only the desaturation part of the retention curve, whereas the hysteresis is not considered.

Description of the methodology

Undisturbed soil samples were taken in the field in each horizon using 98 cm^3-metal cylinders (5 cm high x 5 cm in diameter). The soil sample was carefully saturated from the bottom in the laboratory and a tensiometer probe was then inserted into the soil. The system (soil sample and tensiometer probe) was subsequently placed on a weight and the soil sample was left for evaporation to take place. The weight was connected to a PC as well as the tensiometer probe, allowing a continuous and simultaneous monitoring of capillary pressure and water with time. The experiment was stopped when the maximal capillary pressure measurable with the probe was reached (around -875 cm). At the end of the measurements the soil samples were dried at 105°C for 48 hours to determine the mass of dry material. Then, dividing the actual mass of water in the sample at different time intervals by the sample volume led to the determination of water content. The retention curve is build by plotting capillary pressure versus water content measured at each time step. The tensiometer probe used allowed pressure measurement down to -875 cm, and the water content for lower capillary pressure was subsequently estimated by fitting the measured data with pedotranfer functions.

Description of the tensiometer probes

For measuring the capillary pressure, tensiometer probes were produced following a design developed at Laboratoire d'etude des Transferts en Hydrologie at Environnement (LTHE), Grenoble - France. The arrangement is briefly described in the following section, whereas further information can be found in Angulo-Jaramillo et. al. (1996).

The tensiometer probe (figure A1.) includes three main parts:

- *Porous ceramic probe*: A hollow cylindrical ceramic probe providing hydrostatic continuity between soil medium and pressure sensor.
- *Pressure transducers* (Model AB, Data instruments) registering pressure as a electrical signal
- *Plexiglas holder*, supporting and sealing the above two components

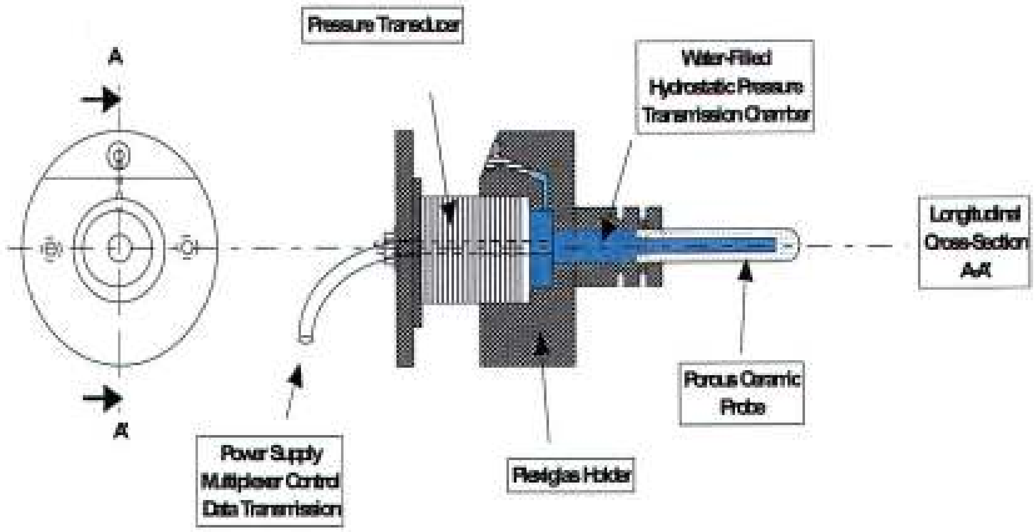

Figure A1 Tensiometer probe

The porous ceramic probe was glued to the holder with a resistant epoxy resin and submerged under water. The cylinder was saturated with water, extracting residual entrapped air by suction with a syringe. With the pressure transducer in place, the whole assembly was kept intact by two screws. Tightening of the screws ensures that the tensiometer is air-tight.

The tensiometer probe delivers electrical voltage proportional to pressure. The sensors were connected to a data acquisition card installed on PC, which made analogue-to-digital conversion of the voltage signal for storage on file. Capillary pressure were derived subsequently from respective calibration of the tension probe.

Description of the weight

The weight was a Mettler PM 4600 connected to a PC through a serial interface.

Measurement protocol

Automated measurements of both pressure and the total weight were only retained when the difference between two successive values exceeded a predetermined threshold, within a given time threshold. A computer code written in Visual-Basic was used to monitor the experiment.

Estimation of ammonia volatilisation

Equations for estimating the ammonia volatilisation was adopted from the EPIC model (Williams et. al. 1984). EPIC uses first-order kinetics for estimating the volatilisation loss (Eq. 18). For the surface applied ammonium the rate coefficient (AKV) was estimated as a function of temperature and wind speed following Eq. 19. When the fertiliser was incorporated into the soil the AKV was instead calculated according to Eq. 22 taking into account incorporation depth, CEC as well as soil temperature.

At the calibration sites the fertiliser was incorporated into the soil with an incorporation depth of circa 2-3 cm. Hence, the volatilisation was assumed to take place only from the upper 3 cm of the soil.

$$NVi = WNH3 \cdot (1 - e^{-AKVi}) \qquad \text{Eq. 18}$$

$$AKV = TFi \cdot WNF \qquad \text{Eq. 19}$$

$$\mathrm{WNF} = 0.335 + 0.16 \cdot \ln(\mathrm{V}) \qquad \text{Eq. 20}$$

$$TF_i = \frac{0.41 \cdot (T_i - 5)}{10} \cdots\cdots\cdots t_i > 5^\circ C \qquad \text{Eq. 21}$$

$$AKV_i = TF_i \cdot CECF_i \cdot DPF_i \qquad \text{Eq. 22}$$

$$CECF_i = 1 - 0.038 \cdot CEC_i \, CECF_i \geq 0 \qquad \text{Eq. 23}$$

$$DPF_i = 1 - \frac{ZZ}{ZZ + exp(4.706 - 0.0305 \cdot ZZ)} \qquad \text{Eq. 24}$$

Where:

NV = Volatilization Loss (kg N/ha·d)
WNH_3= Ammonia concentration (kg NH_3/ha)
T= Soil temperature (·C)
V = Mean wind speed (m/sec)
CECi = Cation Exchange Capacity(meq/100 g soil)
ZZ = Depth to the middle of the soil layer (mm)

After fertiliser application the ratio of ammonia and ammonium concentration in the soil are determined by soil pH and temperature (Eq. 25-Eq. 27). The ammonia concentration needed in Eq. 18 was calculated from Eq. 28 taken into account the soil pH, water and ammonia dissociation constants. Values for pKw as well as pKb were taken from Freney et. al. (1981) whereas the ammonium concentration was retrieved from an initial DAISY simulation, which did not take volatilisation into account.

Soil pH and ammonium concentration as a result of urea hydrolyse. Subsequently, the equilibrium between NH4 and NH3 shift towards NH3 favouring the evolution of NH3. According to Rachhpal-Singh and Nye (1988) each mole of hydrolysed Urea therefore produces one mole of ammonia. Thus, in the case of urea application the soil ammonia concentration needed in Eq. 18 could be retrieved directly from an initial DAISY simulation, which did not take volatilisation loss into consideration.

$$NH_4^+ \Leftrightarrow NH_3 + H^+ \qquad \text{Eq. 25}$$

$$H_2O \Leftrightarrow H^+ + OH^- \qquad \text{Eq. 26}$$

$$Log\left(\frac{[NH^+{}_4]}{[NH_3]}\right) = (pK_b - pK_v) - pH \qquad \text{Eq. 27}$$

$$[NH_3] = \frac{[NH_4^+]}{10^{(pK_w - pK_b) - pH}} \qquad \text{Eq. 28}$$

Where:
K_b *=Ammonia dissociation constant*

K_W *=Water ionization constant*

The calculations indicate that volatilisation occurred four to five days after fertiliser application. However, in the model setup it was assumed that volatilisation occurred right after application, and the volatilisation loss was included by reducing the fertiliser input according to the estimated ammonium loss.

CEC measurements were unavailable from the sandy soils and values for CEC was therefore estimated according to Eq. 29.

$$CEC = C_{Clay} \cdot (0.044 \cdot pH + 0.3) + C_{C-org} \cdot (0.51 \cdot pH - 0.59) \qquad \text{Eq. 29} \quad \textit{(Schachtschabel, 1992).}$$

Where:
CEC = Cation exchange capacity (meq/100 g soil)
C_{Clay} *= Clay concentration in the soil (%)*
$C_{C\text{-}org}$ *= Organic carbon concentration in the soil (%)*

Estimation of total nitrogen mass balance

A total Nitrogen mass balance indicating the nitrogen fluxes entering the saturated zone and channel network was estimated for the study area (Table A1.). The applied calculation as well as the data foundation for this estimated balance is described in the following appendix.

Table A1. Estimated nitrogen fluxes entering the groundwater and channel network (ton N/year)

	1997		1998		1997-1998	
	Mean	**(Max-min)**	**Mean**	**(Min-max)**	**Mean**	**(Min-max)**
Downwards Nitrogen leaching	34	(25-44)	56	(45-67)	45	(38-53)
Upwards Nitrogen" leaching"	-20	(-10-30)	-18	(-12 -24)	-19	(-13-25)
Irrigation	11	(7-12)	16	(11-21)	14	(9-17)
Rice irrigation	6	(4-7)	7	(5-9)	7	(5-8)
Boundary infiltration (Po di Goro)	17	(10-23)	16	(10-24)	17	(10-24)
Boundary infiltration (Sacca di Goro)	<0.01	-	<0.01	-	<0.01	-
Net Nitrogen input	48	(36-57)	77	(59-97)	63	(48-76)
Discharged Nitrogen	43	(24-63)	53	(48-59)	48	(36-61)

Average leaching rates for loam and sand soils were retrieved directly from DAISY, and the total leaching input (L_{each}) were estimated according to Eq. 30. The applied modelling system did not take into account the transformation processes in the rice fields, and leaching input from these fields were not included in the total leaching input.

$$N_{Leach} = L_{Sand} \cdot A_{Sand} + L_{Silty\,loam} \cdot (A_{Silty\,loam} - A_{Rice}) + L_{Sandy\,loam} \cdot A_{Sandy\,Loam} \quad \text{(ton N/year)} \qquad \textit{Eq. 30}$$

Where:

L_i = *Leaching rates for sand, Silty loam and sandy loam soils (ton N/ha·year)*
A_{Sand} = *Area of the arable sand soil (ha)*
$A_{Silty\,loam}$ = *Area of the arable silty loam soil (ha)*
$A_{Sandy\,Loam}$ = *Area of the arable sandy loam soil (ha)*
A_{Rice} = *Area of the rice fields (ha)*

The nitrogen fluxes from boundary infiltration and the amount of nitrogen discharged into the Sacca di Goro (N_i) were calculated according to *Eq. 31*.

$$N_i = Q1_i \cdot C1_i + Q2_i \cdot C2_i + Q3_i \cdot C3_i \quad \text{(ton N/year)} \qquad \textit{Eq. 31}$$

Where:

Q1, Q2 and Q3 = The accumulated water fluxes for the seasons Jan-Apr, May-Sep and Oct-Dec (m3)
C1, C2 and C3= The average nitrogen (NH4-N and NO3-N) concentration in Q1, Q2 and Q3 (ton N/m3)

The water fluxes infiltrating from Po di Goro and Sacca di Goro were retrieved from the MIKESHE model, whereas discharges water fluxes derived from the measured data obtained by Consorzio di Bonifica (Table A2). The ammonium and nitrate concentrations were measured at the channel outlet, Po di Goro and the Canale Bianche. Samples position and the measured concentration are given in Figure A2 and Table A3 respectively. For the calculation the year was divided into three seasons (Jan-Apr, May-Sep, Oct-Dec).

The average seasonal nitrogen concentration in the channels was estimated from the measured data (Table A3), whereas the nitrogen concentration in the Sacca di Goro (Table 4.) was assessed from measurement carried out by Leip (2000).

It should be noted that the nitrogen contribution from the Sacca di Goro considers only nitrate, as measurement of ammonium was unavailable. Moreover, the nitrogen concentration was not measured in Po di Goro during the first winter period of 1997 (Table A3). The average nitrogen concentration (Q_1) required for Eq. 31. was therefore assumed to be equal to the concentration measured 16 of May 1997.

The irrigation water derived from two external sources: Po di Goro supplying the area south of Goro and Canale Bianche supplying the area north of Goro. The amount of nitrogen entering the system with the irrigation water was estimated following Eq. 32. The nitrogen input from irrigation was calculated both for the rice field ($N_{Rice\text{-}irr}$) and for the remaining part of the arable land (N_{irr}).

$$N_{irr} = (I_{Loam} \cdot (A_{Sloam} - A_{Rice}) \cdot C2_{Po} + I_{Loam} \cdot A_{Nloam} \cdot C2_{CB} + I_{Sand} \cdot A_{Sand} \cdot C2_{CB}) \cdot 10^{-9} \quad \text{(t/y)} \qquad \text{Eq. 32}$$

$$N_{Rice-irr} = I_{Rice} \cdot A_{Rice} \cdot C2_{Po} \cdot 10^{-9} \quad \text{(t/y)} \qquad \text{Eq. 33}$$

Where:

$C2_{Po}$ = *Average Nitrogen* ($NH_4 - N + NO_3 - N$) *summer conc in Po di Goro (mg/)*
$C2_{CB}$ = *Average Nitrogen* ($NH_4 - N + NO_3 - N$) *summer conc in Canale Bianche (mg/)*
A_{Sand} = *Area of the sandy arable soil* (m^2)
A_{Sloam} = *Area of the loamy arable soil south of Goro* (m^2)
A_{Nloam} = *Area of the loamy arable soil north of Goro* (m^2)
A_{Rice} = *Area of the rice field* (m^2)
I_{Sand} = *Irrigation rate for sand (mm/y)*
I_{Loam} = *Irrigation rate for loam (mm/y)*
I_{Rice} = *Irrigation rate for Rice (mm/y)*

For the leaching estimates the range of variation were given in terms of standard deviation. The range of variation for the remaining estimates was illustrated in terms of minimum and maximum values calculated by using the lowest and highest nitrogen concentration within each season.

Table A2. Water fluxes for the various season

		Measured channel discharge	*Simulated boundary infiltration (m3)*	
			Po di Goro	*Sacca di Goro*
Jan-Apr	***1996***	*4.76E+06*	*7.28E+05*	*5.62E+05*
May-Sep	***1996***	*6.01E+06*	*1.73E+06*	*1.31E+06*
Oct-Dec	***1996***	*5.66E+06*	*5.29E+05*	*3.91E+05*
Jan-Apr	***1997***	*3.19E+06*	*7.28E+05*	*5.98E+05*
May-Sep	***1997***	*5.13E+06*	*1.05E+06*	*8.59E+05*
Oct-Dec	***1997***	*3.83E+06*	*5.52E+05*	*4.73E+05*
Jan-Apr	***1998***	*4.30E+06*	*7.30E+05*	*6.57E+05*
May-Sep	***1998***	*5.61E+06*	*9.89E+05*	*8.50E+05*
Oct-Dec	***1998***	*4.96E+06*	*5.60E+05*	*4.49E+05*

Figure A2. Sample position for water analysis

Table A2. Measured NH4-N and NO3- N concentration in surface water

	Idrovora		Po di Goro		Canale Bianche	
	(mg/l) NO3-N	(mg/l) NH4-N	(mg/l) NO3-N	(mg/l) NH4-N	(mg/l) NO3-N	(mg/l) NH4-N
14-Nov-96	13.5	1.0	-	-	-	-
4-Dec-96	7.9	1.3	-	-	-	-
21-Feb-97	4.0	2.3	-	-	-	-
16-May-97	11.8	< 0.1	7.4	0.20	-	-
13-Jun-97	0.10	0.35	3.6	0.54	0.12	0.18
11-Jul-97	1.3	1.1	6.9	< 0.01	0.18	0.25
6-Mar-98	1.2	1.3	3.1	5.0	3.0	0.30
12-Jun-98	0.50	0.07	4.5	< 0.006	2.5	< 0.006
10-Jul-98	6.1	0.01	7.7	< 0.009	2.7	< 0.009
16-Oct-98	7.8	< 0.009	8.0	< 0.009	2.7	< 0.009

Table A3. Measured NO3-N concentration in the Sacca di Goro (Leip, 2000)

	NO3-N (mg/l) Hide tide		NO3-N (mg/l) Low tide		NO3-N (mg/l)
	Surface	Bottom	Surface	Bottom	Average
26-Oct-96	5.46E-03	5.63E-03	5.52E-03	5.05E-03	5.42E-03
21-Mar-97	2.23E-03	0.00E+00	3.08E-03	0.00E+00	1.33E-03
8-May-97	7.01E-04	5.39E-04	8.49E-04	1.75E-03	9.59E-04
9-May-97	7.69E-04	7.38E-04	3.08E-03	6.20E-04	1.30E-03
20-Jun-97	1.49E-04	9.92E-05	2.48E-04	2.29E-04	1.81E-04
24-Jul-97	1.43E-03	5.08E-04	2.91E-04	1.98E-04	6.08E-04
13-Nov-97	1.85E-03	1.87E-03	2.61E-03	2.56E-03	2.22E-03
23-Jan-97	2.10E-03	2.48E-03	3.24E-03	2.27E-03	2.52E-03

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